Chapter 13. Wetlands

Lead Author: Scott D. Bridgham[1]

Contributing Authors: J. Patrick Megonigal,[2] Jason K. Keller,[2] Norman B. Bliss[3], and Carl Trettin[4]

[1]Center for Ecology and Evolutionary Biology, University of Oregon, [2]Smithsonian Environmental Research Center, [3]Science Applications International Corporation, USGS Center for Earth Resources Observation and Science, [4]Center for Forested Wetland Research, USDA Forest Service

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KEY FINDINGS

- North America is home to approximately 41% of the global wetland area, encompassing about 2.5 million km^2 with a carbon pool of approximately 220 Gt, mostly in peatland soils.
- North American wetlands currently are a CO_2 sink of approximately 70 Mt C yr^{-1}, but that estimate has an uncertainty of greater than 100%. North American wetlands are also a source of approximately 26 Mt yr^{-1} of methane, a more potent atmospheric heat-trapping gas. The uncertainty in that flux is also greater than 100%.
- Historically, the destruction of North American wetlands through land-use change has reduced carbon storage in wetlands by 43 Mt C yr^{-1}, primarily through the oxidation of carbon in peatland soils as they are drained and a more general reduction in carbon sequestration capacity of wetlands converted to other land uses. Methane emissions have also declined with the loss of wetland area.
- Projections of future carbon storage and methane emissions of North American wetlands are highly uncertain and complex, but the large carbon pools in peatlands may be at risk for oxidation and release to the atmosphere as CO_2 if they become substantially warmer and drier. Methane emissions may increase with warming, but the response will likely vary with wetland type and with changes in precipitation.
- Because of the potentially significant role of North American wetlands in methane production, the activities associated with the restoration, creation and protection of wetlands are likely to focus on the ecosystem services that wetlands provide, such as filtering of toxics, coastal erosion protection, wildlife habitat, and havens of biodiversity, rather than on carbon sequestration per se.
- Research needs to reduce the uncertainties in carbon storage and fluxes in wetlands to provide information about management options in terms of carbon sequestration and trace gas fluxes.

INTRODUCTION

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

Peatlands occupy about 3% of the terrestrial global surface, yet they contain 16–33% of the total soil carbon pool (Gorham, 1991; Maltby and Immirzi, 1993). Most peatlands occur between 50 and 70º N, although significant areas occur at lower latitudes (Matthews and Fung, 1987; Aselmann and Crutzen, 1989; Maltby and Immirzi, 1993). Large areas of peatlands exist in Alaska, Canada, and in the northern midwestern, northeastern, and southeastern United States (Bridgham *et al.*, 2000). Because this peat formed over thousands of years, these areas represent a large carbon pool but with relatively slow rates of accumulation. By comparison, estuarine wetlands and some freshwater mineral-soil wetlands rapidly sequester carbon as soil organic matter due to rapid burial in sediments. Large areas of wetlands have been converted to other land uses globally and in North America (Dugan, 1993; OECD, 1996), which may have resulted in a net flux of carbon to the atmosphere (Armentano and Menges, 1986; Maltby and Immirzi, 1993). Additionally, wetlands emit 92–237 Mt methane (CH_4) yr^{-1}, which is a large fraction of the total annual global flux of about 600 Mt CH_4 yr^{-1} (Ehhalt *et al.*, 2001). This is important because methane is a potent greenhouse gas, second in importance to only carbon dioxide (Ehhalt *et al.*, 2001). A number of previous studies have examined the role of peatlands in the global carbon balance (reviewed in Mitra *et al.*, 2005). Roulet (2000) focused on the role of Canadian peatlands in the Kyoto process. Here we augment these previous studies by considering all types of wetlands (not just peatlands) and integrate new data to examine the carbon balance in the wetlands of Canada, the United States, and Mexico. We also briefly compare these values to those from global wetlands.

Given that many undisturbed wetlands are a natural sink for carbon dioxide and a source of methane, a note of caution in interpretation of our data is important. Using the International Panel on Climate Change (IPCC) terminology, a radiative forcing denotes “an externally imposed perturbation in the radiative energy budget of the Earth's climate system" (Ramaswamy *et al.*, 2001). Thus, it is the change from a baseline condition in greenhouse gas fluxes in wetlands that constitute a radiative forcing that will impact climate change, and carbon fluxes in unperturbed wetlands are important only in establishing a baseline condition. For example, historical steady state rates of methane emissions from wetlands have zero net radiative forcing, but an increase in methane emissions due to climatic warming would constitute a positive radiative forcing. Similarly, steady state rates of soil carbon sequestration in wetlands have zero net radiative forcing, but the lost sequestration capacity and the oxidation of the extant soil carbon pool in drained wetlands are both positive radiative forcings. Here we consider changes from a historical baseline of about 1800 A.D. to present and future emissions of greenhouse gas fluxes in North American wetlands.

INVENTORIES

Current Wetland Area and Rates of Loss

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

The conterminous United States has 312,000 km^2 of FWMS wetlands, 93,000 km^2 of peatlands, and 23,000 km^2 of estuarine wetlands, which encompass 5.5% of the land area (Table 13-1). This represents just 48% of the original wetland area in the conterminous United States (Table 13A-1 in Appendix 13A). However, wetland losses in the United States have declined from 1,855 km^2 yr^{-1} in the 1950s–1970s to 237 km^2 yr^{-1} in the 1980s–1990s (Dahl, 2000). Such data mask large differences in loss rates among wetland classes and conversion of wetlands to other classes, with potentially large effects on carbon stocks and fluxes (Dahl, 2000). For example, the majority of wetland losses in the United States have occurred in FWMS wetlands. As of the early 1980s, 84% of U.S. peatlands were unaltered (Armentano and Menges, 1986; Maltby and Immirzi, 1993; Rubec, 1996), and, given the current regulatory environment in the United States, recent rates of loss are likely small.

Table 13-1. The area, carbon pool, net carbon balance, and methane flux from wetlands in North America and the world.

Canada has 1,301,000 km^2 of wetlands, covering 14% of its land area, of which 87% are peatlands (Table 13-1). Canada has lost about 14% of its wetlands, mainly due to agricultural development of

FWMS wetlands (Rubec, 1996), although the ability to estimate wetland losses in Canada is limited by the lack of a regular wetland inventory.

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

Problems with inadequate wetland inventories are even more prevalent in lesser developed countries (Finlayson *et al.*, 1999). We estimate a global wetland area of 6.0×10^6 km^2 (Table 13-1); thus, North America currently has about 43% of the global wetland area. It has been estimated that about 50% of the world's historical wetlands have been converted to other uses (Moser *et al.*, 1996).

Carbon Pools

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

Soil Carbon Fluxes

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

Very little attention has been given to the role of FWMS wetlands in North American or global carbon balance estimates, with the exception of methane emissions. Carbon sequestration associated with sediment deposition is a potentially large, but poorly quantified, flux in wetlands (Stallard, 1998). Using a review by Johnston (1991), we calculate a substantial carbon accumulation rate in sedimentation in FWMS wetlands of -129 g C m^{-2} yr^{-1} (see Appendix 13A). However, it is unlikely that the actual sequestration rate is this high. Researchers may have preferentially chosen wetlands with high sedimentation rates to study this process, providing a bias towards greater carbon sequestration. More fundamentally, it is important to distinguish between autochthonous carbon (derived from on-site plant production) and allochthonous carbon (imported from outside the wetland) in soil carbon storage. Almost all of the soil carbon stored in peatlands is of autochthonous origin and represents sequestration of atmospheric carbon dioxide at the landscape scale. In contrast, much of the soil carbon that is stored in FWMS wetlands is likely of allochthonous origin. At a landscape scale, redistribution of sediments from uplands to wetlands does not represent net carbon sequestration if the decomposition rate of carbon is the same in both environments. Carbon exported from upland source areas is likely to be relatively recalcitrant and physically protected from decomposers by association with mineral soil. Thus, despite the anaerobic conditions in wetlands, decomposition rates in deposited sediments may not be substantially lower than in the uplands from which those sediments were eroded. There are no data to our knowledge to evaluate these important caveats. Because of this reasoning, we somewhat arbitrarily assumed that sediment carbon sequestered in FWMS wetlands is of allochthonous origin and decomposed 25% slower than in the uplands from which the sediment was derived. Accordingly, we reduced our calculated rates of *landscape-level* carbon sequestration in FWMS wetlands by 75% to -34 g C m^{-2} yr^{-1} (Table 13A-2 in Appendix 13A). Nevertheless, this still represents a substantial carbon sink. For example, Stallard (1998) estimated that global wetlands are a large sediment sink, with a flux on the order of -1 Gt C yr^{-1}. However, this analysis was based on many assumptions and was acknowledged by the author to be a first guess at best.

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

We estimate that estuarine wetlands currently sequester -9.7 Mt C yr^{-1}, with a historical reduction in sequestration capacity of 1.6 Mt C yr^{-1} due to loss of area (Table 13-1). However, the reduction is almost certainly greater because our 'historical' area is only from the 1950s. Despite the relatively small area of estuarine wetlands, they currently contribute about 26% of total wetland carbon sequestration in the conterminous United States and about 14% of the North American total. Estuarine wetlands sequester carbon at a rate about 10 times higher on an area basis than other wetland ecosystems due to high sedimentation rates, high soil carbon content, and constant burial due to sea level rise. Estimates of sediment deposition rates in estuarine wetlands are robust, but it is unknown to what extent soil carbon sequestration is due to allochthonous versus autochthonous carbon. As with FWMS wetlands, the contribution of soil carbon sequestration in estuarine wetlands to the North American carbon budget is overestimated to the extent that allochthonous carbon simply represents redistribution of carbon in the landscape. There is also large uncertainty in the area and carbon content of mud flats, particularly in Canada and Mexico.

Overall, North American wetland soils appear to be a substantial carbon sink with a net flux of -70 Mt C yr^{-1} (with very large error bounds because of FWMS wetlands) (Table 13-1). The large-scale conversion of wetlands to upland uses has led to a reduction in the wetland soil carbon sequestration capacity of 25 Mt C yr^{-1} from the likely historical rate (Table 13-1), but this estimate is driven by large losses of FWMS wetlands with their highly uncertain sedimentation carbon sink. Adding in the current net oxidative flux of 18 Mt C yr^{-1} from conterminous U.S. peatlands, we estimate that North American wetlands currently sequester 43 Mt C yr^{-1} less than they did historically (Table 13A-2 in Appendix 13A). Furthermore, North American peatlands and FWMS wetlands have lost 2.6 Gt and 4.9 Gt of soil carbon, respectively, and collectively they have lost 2.4 Gt of plant carbon since approximately 1800. Very little data exist to estimate carbon fluxes for freshwater Mexican wetlands, but because of their small area, they will not likely have a large impact on the overall North American estimates.

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

Methane and Nitrous Oxide Emissions

We estimate that North American wetlands emit 26 Mt CH_4 yr^{-1} (Table 13-1), a value that is substantially higher than the previous estimate by Bartlett and Harriss (1993) (see Appendix 13A). A mechanistic methane model yielded similar rates of 3.8 and 7.1 Mt CH_4 yr^{-1} for Alaska and Canada, respectively (Zhuang *et al.*, 2004). For comparison, a regional inverse atmospheric modeling approach estimated total methane emissions (from all sources) of 16 and 54 Mt CH_4 yr^{-1} for boreal and temperate North America, respectively (Fletcher *et al.*, 2004b).

Methane emissions are currently about 24 Mt CH_4 yr^{-1} less than they were historically in North American wetlands (see Table 13A-4 in Appendix 13A) because of the loss of wetland area. We do not consider the effects of conversion of wetlands from one type to another (Dahl, 2000), which may have a significant impact on methane emissions. Similarly, we estimate that global methane emissions from natural wetlands are only about half of what they were historically due to loss of area (Table 13A-4 in Appendix 13A). However, this may be an overestimate because wetland losses have been higher in more developed countries than less developed countries (Moser *et al*., 1996), and wetlands at lower latitudes have higher emissions on average (Bartlett and Harriss, 1993).

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

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

Methane has GWPs of 1.9, 6.3, and 16.9 CO_2-carbon equivalents on a mass basis across 500-year, 100-year, and 20-year time frames, respectively (Ramaswamy *et al*., 2001)[1]. Depending upon the time frame and within the large confidence limits of many of our estimates in Table 13-1, the *net radiative balance* of North American wetlands as a whole currently are in a range between approximately neutral and a large source of net CO_2-carbon equivalents to the atmosphere (note that we discuss *net radiative forcing* in *Trends and Drivers of Wetland Carbon Fluxes*). It is likely that FWMS wetlands, with their high methane emissions, are a net source of CO_2-carbon equivalents to the atmosphere. In contrast, estuarine wetlands are a net sink for CO_2-carbon equivalents because they support both rapid rates of carbon sequestration and low methane emissions. However, caution should be exercised in using GWPs to draw conclusions about changes in the net flux of CO_2-carbon equivalents because GWPs are based upon a pulse of a gas into the atmosphere, whereas carbon sequestration is more or less continuous. For example, if one considers continuous methane emissions and carbon sequestration in peat over time, most peatlands are a net sink for CO_2-carbon equivalents because of the long lifetime of carbon dioxide sequestered as peat (Frolking *et al*., 2006).

[1]GWPs in Ramaswamy *et al.* (2001) were originally reported in CO_2-mass equivalents. We have converted them into CO_2-carbon equivalents so that the net carbon balance and methane flux columns in Table 13-1 can be directly compared by multiplying methane fluxes by the GWPs given here.

Plant Carbon Fluxes

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

TRENDS AND DRIVERS OF WETLAND CARBON FLUXES

While extensive research has been done on carbon cycling and pools in North American wetlands, to our knowledge, this is the first attempt at an overall carbon budget for all of the wetlands of North America, although others have examined the carbon budget for North American peatlands as part of global assessments (Armentano and Menges, 1986; Maltby and Immirzi, 1993; Joosten and Clarke, 2002). Historically, the destruction of wetlands through land-use changes has had the largest effect on the carbon fluxes and, consequently, the radiative forcing of North American wetlands. The primary effects have been a reduction in their ability to sequester carbon (a small to moderate increase in radiative forcing depending on carbon sequestration by sedimentation in FWMS and estuarine wetlands), oxidation of their soil carbon reserves upon drainage (a small increase in radiative forcing), and a reduction in the emission of methane to the atmosphere (a large decrease in radiative forcing) (Table 13A-1 and Appendix 13A). Globally, the disturbance of peatlands appears to have shifted them into a net source of carbon to the atmosphere. Any positive effect of wetland loss due to a reduction in their methane emissions, and hence radiative forcing, will be more than negated by the loss of the many ecosystem services they provide such as havens for biodiversity, recharge of groundwater, reduction in flooding, fish nurseries, etc. (Zedler and Kercher, 2005).

A majority of the effort in examining future global change impacts on wetlands has focused on northern peatlands because of their large soil carbon reserves, although under current climate conditions they have modest methane emissions (Moore and Roulet, 1995; Roulet, 2000; Joosten and Clarke, 2002, and references therein). The effects of global change on carbon sequestration in peatlands are probably of minor importance as a global flux because of the relatively low rate of peat accumulation. However, losses of soil carbon stocks in peatlands drained for agriculture and forestry (Table 13A-2 in Appendix 13A) attest to the possibility of large losses from the massive soil carbon deposits in northern peatlands if they become substantially drier in a future climate. Furthermore, Turetsky *et al.* (2004) estimated that up to 5.9 Mt C yr^{-1} are released from western Canadian peatlands by fire and predicted that increases in fire frequency may cause these systems to become net atmospheric carbon sources.

Our compilation shows that attention needs to be directed toward understanding climate change impacts to FWMS wetlands, which collectively emit over 3-times more methane than North American peatlands and potentially sequester an equivalent amount of carbon. The effects of changing water table depths are somewhat more tractable in FWMS wetlands than peatlands because FWMS wetlands have less potential for oxidation of soil organic matter. In forested FWMS wetlands, increased precipitation and runoff may increase radiative forcing by simultaneously decreasing wood production and increasing methanogenesis (Megonigal *et al*., 2005). The influence of changes in hydrology on methane emissions, plant productivity, soil carbon preservation, and sedimentation will need to be addressed in order to fully anticipate climate change impacts on radiative forcing in these systems.

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

One of the greatest concerns is how climate change will affect future methane emissions from wetlands because of their large GWP. Wetlands emit about 107 Mt CH_4 yr^{-1} (Table 4), or 20% of the global total. Increases in atmospheric methane concentrations over the past century have had the second largest radiative forcing (after carbon dioxide) in human-induced climate change (Ehhalt *et al*., 2001). Moreover, methane fluxes from wetlands have provided an important radiative feedback on climate over the geologic past (Chappellaz *et al*., 1993; Blunier *et al*., 1995; Petit *et al*., 1999). The large global warming observed since the 1990s may have resulted in increased methane emissions from wetlands (Fletcher *et al*., 2004a; Wang *et al*., 2004; Zhuang *et al*., 2004).

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

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

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

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

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

OPTIONS AND MEASURES

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

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

Regardless of their internal carbon balance, the area of restored wetlands is currently too small to form a significant carbon sink at the continental scale. Between 1986 and 1997, only 4,157 km^2 of uplands were converted into wetlands in the conterminous United States (Dahl, 2000). Using the soil carbon sequestration rate of 305 g C m^{-2} yr^{-1} found by Euliss *et al.* (2006) for restored prairie pothole wetlands[2], we estimate that wetland restoration in the U.S. would have sequestered 1.3 Tg C over this 11-year period. However, larger areas of wetland restoration may have a significant impact on carbon sequestration. A simulation model of planting 20,000 km^2 into bottomland hardwood trees as part of the Wetland Reserve Program in the United States showed a sequestration of 4 Mt C yr^{-1} through 2045 (Barker *et al*., 1996). Euliss *et al.* (2006) estimated that if all cropland on former prairie pothole wetlands in the U.S. and Canada (162,244 km^2) were restored that 378 Tg C would be sequestered over 10 years in soils and plants. However, neither study accounted for the GWP of increased methane emissions.

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

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

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

Canada lacks any regular inventory of its wetlands, and thus it is difficult to quantify land-use impacts upon their carbon fluxes and pools. While excellent scientific data exists on most aspects of carbon cycling in Canadian peatlands, Canadian FWMS and estuarine wetlands have been relatively poorly studied, despite having suffered large proportional losses to land-use change. Wetland data for Mexico is

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

almost entirely lacking. Thus, anything that can be done to improve upon this would be helpful. All wetland inventories should consider the area of estuarine mud flats, which have the potential to sequester considerable carbon, and are poorly understood with respect to carbon sequestration.

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

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

	Area[a] (km^2)		**Carbon Pool[b] (Gt C)**		**Net Carbon Balance[c] ($Mt\ C\ yr^{-1}$)**		**Historical Loss in Sequestration Capacity ($Mt\ C\ yr^{-1}$)**		**Methane Flux ($Mt\ CH_4\ yr^{-1}$)**	
Canada										
Peatland	1,135,608	****	149	****	-19	***	0.3	*	3.2	**
Freshwater Mineral	158,720	**	4.9	**	-5.1	*	6.5	*	5.7	*
Estuarine	6,400	***	0.1	***	-1.3	**	0.5	*	0.0	***
Total	1,300,728	****	154	****	-25	**	7.2	*	8.9	*
Alaska										
Peatland	132,196	****	15.9	**	-2.0	**	0.0	****	0.3	*
Freshwater Mineral	555,629	****	27.1	**	-18	*	0.0	****	1.4	*
Estuarine	8,400	****	0.1	***	-1.9	**	0.0	****	0.1	***
Total	696,224	*****	43.2	**	-22	*	0.0	****	1.8	*
Conterminous United States										
Peatland	93,477	****	14.4	***	4	*	2.1	*	3.4	**
Freshwater Mineral	312,193	*****	6.2	***	-18	*	15	*	11.2	**
Estuarine	23,000	*****	0.6	*****	-4.9	**	0.4	*	0.1	***
Total	428,670	*****	21.2	***	-19	*	17	*	14.7	**
U.S. Total	1,124,895	*****	64	**	-41	*	17	*	17	**
Mexico										
Peatland	10,000	*	1.5	*	-1.6	*	ND[d]	*	0.4	*
Freshwater Mineral	20,685	*	0.4	*	-0.7	*	ND	*	0.7	*
Estuarine	5,000	*	0.2	*	-1.6	*	0.5	*	0.0	*
Total	35,685	*	2.1	*	-3.9	*	ND	*	1.1	*

North America										
Peatland	1,371,281	****	180	****	-18	*	2.4	*	7	**
Freshwater Mineral	1,047,227	****	39	***	-42	*	21	*	19	*
Estuarine	42,800	***	1.0	***	-9.7	**	1.4	*	0.2	**
Total	2,461,308		220		-70	*	25	*	26	*
Global										
Peatland	3,443,000	***	460	***	150	**	16	*	37	**
Freshwater Mineral	2,315,000	***	46	***	-75	*	87	*	68	**
Estuarine	203,000	*	5.4	*	-43	*	13.2	*	1.5	**
Total	5,961,000	***	511	***	32	*	116	*	107	**

[a]Estuarine includes salt marsh, mangrove, and mudflat, except for Mexico and global for which no mudflat estimates were available.

[b]Includes soil C and plant C, but overall soil C is 98% of the total pool.

[c]Includes soil C sequestration, plant C sequestration, and loss of C due to drainage of wetlands. Plant C sequestration and soil oxidative flux due to drainage are either unknown or negligible for North American wetlands except for the conterminous United States (see Appendix 13A).

[d]No data.

The error categories are as follows:

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

Appendix 13A
Wetlands – Supplemental Material

INVENTORIES

Current Wetland Area and Rates of Loss

The ability to estimate soil carbon pools and fluxes in North American wetlands is constrained by the national inventories (or lack thereof) for Canada, the United States, and Mexico (Davidson *et al*., 1999). The National Wetland Inventory (NWI) program of the United States has repeatedly sampled several thousand wetland sites using aerial photographs and more limited field verification. The data are summarized in a series of reports detailing changes in wetland area in the conterminous United States for the periods of the mid-1950s to mid-1970s (Frayer *et al*., 1983), mid-1970s to mid-1980s (Dahl and Johnson, 1991), and 1986 to 1997 (Dahl, 2000). We used these relatively high-quality data sets extensively for estimating wetland area and loss rates in the conterminous United States, including mud flats. However, the usefulness of the NWI inventory reports for carbon budgeting is limited by the level of classification used to define wetland categories within the Cowardin *et al.* (1979) wetland classification system. At the level used in the national status and trend reports, vegetated freshwater wetlands are classified by dominant physiognomic vegetation type, and it is impossible to make the important distinction between wetlands with deep organic soils (i.e., peatlands) and wetlands with mineral soils. The data are not at an adequate spatial resolution to combine with U.S. Department of Agriculture (USDA) National Resources Conservation Service (NRCS) soil maps to discriminate between the two types of wetlands (T. Dahl, personal comm.). Because of these data limitations, we used the NRCS soil inventory of peatlands (i.e., Histosols and Histels, or peatlands with and without permafrost, respectively) to estimate historical peatland area (Bridgham *et al*., 2000) and combined these data with regional estimates of loss (Armentano and Menges, 1986) to estimate current peatland area in the conterminous United States. We calculated the current area of freshwater mineral-soil (FWMS) wetlands in the conterminous United States by subtracting peatland area from total wetland area (Dahl, 2000). This approach was limited by the Armentano and Menges peatland area data being current only up to the early 1980s, although large losses of peatlands since then are unlikely due to the institution of wetland protection laws.

We used a similar approach for Alaskan peatlands: peatland area was determined by the NRCS soil inventory [N. Bliss, query of the NRCS State Soil Geographic (STATSGO) database, February 2006] and overall wetland inventory was determined by standard NWI methods (Hall *et al*., 1994). However, our peatland estimate of 132,000 km^2 (Table 13A-1) is 22% of the often cited value by Kivinen and Pakarinen (1981) of 596,000 km^2.

Table 13A-1. Current and historical area of wetlands in North America and the world ($\times 10^3$ km^2).

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

The area of salt marsh in the conterminous U.S. and Alaska were taken from Alexander *et al*. (1986) and Hall (1994), respectively, as reported in Mendelssohn and McKee (2000). Because these estimates include brackish tidal marshes, they cannot be compared directly to the area of Canadian salt marsh. The historical area of tidal wetlands in the conterminous U.S. was based on the NWI (Dahl, 2000), but 'historical' here only refers to the 1950s as we could not find earlier estimates. It is almost certain that historical salt marsh area in the conterminous U.S. was larger than our estimate. We made the reasonable assumption that the historical area of Alaskan tidal wetlands was similar to the current area. The area of freshwater tidal marshes was not included.

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

No national wetland inventories have been done for Mexico. Current freshwater wetland estimates for Mexico were taken from Davidson *et al*. (1999) and Spiers (1999), who used inventories of discrete wetland regions performed by a variety of organizations. Thus, freshwater wetland area estimates for Mexico are highly unreliable and are possibly a large underestimate. For mangrove area in Mexico, we used the estimates compiled by Mendelssohn and McKee (2000), which are similar to estimates reported in Davidson *et al.* (1999) and Spalding *et al.* (1997). We could find no estimates of tidal marsh or mud flat area for Mexico. Since most vegetated Mexican tidal wetlands are dominated by mangroves (Olmsted, 1993; Mendelssohn and McKee, 2000), the omission of Mexican tidal marshes should not significantly affect our carbon budget. However, there may be large areas of mud flat that would significantly increase our estimate of carbon pools and sequestration in this country. We arbitrarily estimated that 25% of the mangrove area was lost since the late 1800s, which is less than the rough worldwide estimate of 50% wetland loss that is often cited (see Zedler and Kercher, 2005). A lower estimate is reasonable because wetland losses are lower in coastal systems than freshwater systems (Zedler and Kercher, 2005).

CARBON POOLS

Freshwater Mineral-Soil (Gleysol) Carbon Pools

Gleysol is a soil classification used by the Food and Agriculture Organization (FAO) and many countries that denotes mineral soils formed under waterlogged conditions (FAO-UNESCO, 1974). Tarnocai (1998) reported a soil carbon density of 200 Mg C ha^{-1} for Canadian Gleysols but did not indicate to what depth this extended. Batjes (1996) determined soil carbon content globally from the *Soil Map of the World* (FAO, 1991) and a large database of soil pedons. He gave a very similar average value for soil carbon density of 199 Mg C ha^{-1} (CV[3] = 212%, n = 14 pedons) for Gleysols of the world to 2-m depth; to 1-m depth, he reported a soil carbon density of 131 Mg C ha^{-1} (CV = 109%, n =142 pedons).

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

Some caution is necessary regarding the use of Gleysol or 'wet' mineral soil carbon densities because apparently they include large areas of seasonally wet soils that are not considered wetlands by the more conservative definition of wetlands used by the United States and many other countries and organizations.

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

For example, Eswaran *et al.* (1995) estimated that global wet mineral-soil area was 8,808,000 km^2, which is substantially higher than the commonly accepted mineral-soil wetland area estimated by Matthews and Fung (1987) of 2,289,000 km^2 and Aselmann and Crutzen (1989) of 2,341,000 km^2, even accounting for substantial global wetland loss. In our query of the NRCS STATSGO database for the United States, we found 1,258,000 km^2 of wet soils in the conterminous United States versus our estimate of 312,000 km^2 of FWMS wetlands currently and 762,000 km^2 historically (Table 13A-1). We assume that including these wet-but-not-wetland soils will decrease the estimated soil carbon density, but to what degree we do not know. However, just considering the differences in area will give large differences in the soil carbon pool. For example, Eswaran *et al.* (1995) estimated that wet mineral soils globally contain 108 Gt C to 1-m depth, whereas our estimate is 46 Gt C to 2-m depth (Table 13A-2).

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

Peatland Soil Carbon Pools

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

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

Table 13A-2. Soil carbon pools (Gt) and fluxes (Mt yr^{-1}) of wetlands in North America and the world.

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

Estuarine Soil Carbon Pools

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

Plant Carbon Pools

While extensive data on plant biomass in individual wetlands have been published, no systematic inventory of wetland plant biomass has been undertaken in North America. Nationally, the forest carbon biomass pool (including aboveground and belowground biomass) has been estimated to be 5.49 kg C m^{-2} (Birdsey, 1992), which we used for forested wetlands in the United States and Canada. This approach assumes that wetland forests do not have substantially different biomass carbon densities from upland forests. There is one regional assessment of forested wetlands in the southeastern United States, which comprise approximately 35% of the total forested wetland area in the conterminous United States. We utilized the southeastern U.S. regional inventory to evaluate this assumption; aboveground tree biomass averaged 125.2 m^3 ha^{-1} for softwood stands and 116.1 m^3 ha^{-1} for hardwood stands. Using an average wood density and carbon content, the carbon density for these forests would be 3.3 kg C m^{-2} for softwood stands and 4.2 kg C m^{-2} for hardwood stands. However, these estimates do not include understory vegetation, belowground biomass, or dead trees, which account for 49% of the total forest biomass (Birdsey, 1992). Using that factor to make an adjustment for total forest biomass, the range would be 4.9 to 6.6 kg C m^{-2} for the softwood and hardwood stands, respectively. Accordingly, the assumption of using 5.49 kg C m^{-2} seems reasonable for a national-level estimate.

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

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

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

(23.5°). Total biomass was estimated using a root-to-shoot ratio of 0.82 and a carbon-mass-to-biomass ratio of 0.45, both from Twilley *et al.* (1992).

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

Table 13A-3. Plant carbon pools (Gt) and fluxes (Mt yr^{-1}) of wetlands in North America and the world.

CARBON FLUXES

Peatland Soil Carbon Accumulation Rates

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

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

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

Freshwater Mineral-Soil Wetland Carbon Accumulation Rates

Many studies have estimated sediment deposition rates in FWMS wetlands, with an average rate of 1,680 g m^{-2} yr^{-1} (range 0 to 7,840) in a review by Johnston (1991). Assuming 7.7% carbon for FWMS wetlands (Batjes, 1996), this gives a substantial accumulation rate of -129 g C m^{-2} yr^{-1}. Johnston (1991) found many more studies that just reported vertical sediment accumulation rates, with an average of 0.69 cm yr^{-1} (range -0.6 to 2.6). If we assume a bulk density of 1.38 g cm^{-3} for FWMS wetlands (Batjes, 1996), this converts into an impressive accumulation rate of -733 g C m^{-2} yr^{-1}. For reasons discussed in the main chapter, we assumed a lower carbon sequestration rate in FWMS wetlands of -34 g C m^{-2} yr^{-1}.

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

Estuarine Carbon Accumulation Rates

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

Extractive Uses of Peat

Use of peat for energy production is, and always has been, negligible in North America, as opposed to other parts of the world (WEC, 2001). However, Canada produces a greater volume of horticultural and agricultural peat than any other country in the world (WEC, 2001). Currently, 124 km^2 of Canadian peatlands have been under extraction now or in the past (Cleary *et al*., 2005). A life-cycle analysis by these authors estimated that as of 1990 Canada emitted 0.9 Mt yr^{-1} of CO_2-C equivalents through peat extraction. The U.S. production of horticultural peat is about 19% of Canada's (Joosten and Clarke, 2002), which assuming a similar life-cycle as for Canada, suggests that the United States produces 0.2 Mt of CO_2-C equivalents through peat extraction.

Methane Fluxes

Moore *et al.* (1995) reported a range of methane fluxes from 0 to 130 g CH_4 m^{-2} yr^{-1} from 120 peatland sites in Canada, with the majority <10 g CH_4 m^{-2} yr^{-1}. They estimated a low average flux rate of 2 to 3 g CH_4 m^{-2} yr^{-1}, which equaled an emission of 2–3 Mt CH_4 yr^{-1} from Canadian peatlands. We used an estimate of 2.5 g CH_4 m^{-2} yr^{-1} for Canadian peatlands and Alaskan freshwater wetlands (Table 13A-4).

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

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

Table 13A-5. Methane fluxes measured in the conterminous United States.

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

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

Using the conversion factors for freshwater and estuarine wetlands, we estimated average annual fluxes from the average daily fluxes. For freshwater wetlands, the calculated average annual flux rate was

38.6 g CH_4 m^{-2} yr^{-1} (n = 74), which is slightly larger than the average actual measured flux rate of 32.1 g CH_4 m^{-2} yr^{-1} (n = 32). For estuarine wetlands, the average calculated annual flux rate was 9.8 g CH_4 m^{-2} yr^{-1} (n = 25), which is smaller than the average measured flux rate of 16.9 g CH_4 m^{-2} yr^{-1} (n = 13). However, if we remove one outlier, the average measured flux rate is 10.2 g CH_4 m^{-2} yr^{-1}.

Finally, we combined both approaches. In cases where a paper presented an annual value, we used that number. In cases where only an average daily number was presented, we used that value corrected with the appropriate conversion factor. For conterminous U.S. wetlands, FWMS Canadian wetlands, and Mexican wetlands, we used an average flux of 36 g CH_4 m^{-2} yr^{-1}, and for estuarine wetlands, we used an average flux of 10.3 g CH_4 m^{-2} yr^{-1}.

Plant Carbon Fluxes

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

Tree biomass carbon sequestration averages -140 g C m^2 yr^{-1} in U.S. forests across all forest types (Birdsey, 1992). Using the tree growth estimates from the southeastern U.S. regional assessment of wetland forests (Brown *et al*., 2001) yields an even lower estimate of sequestration in aboveground tree biomass (approx. -50.2 g C m^2 yr^{-1}). We used this lower value and area estimates from Dahl (2000) to estimate that forested wetlands in the conterminous U.S. currently sequester -10.3 Mt C yr^{-1}.

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

	Permafrost peatlands	Non-permafrost peatlands	Mineral-soil freshwater	Salt marsh	Mangrove	Mudflat	**Total**
Canada							
Current	422[a]	714[a]	159[b]	0.4[c]	0	6[d]	1301
Historical	424[e]	726[f]	359[g]	1.3[b]	0	7[h]	1517
Alaska							
Current	89[i]	43[i]	556[j]	1.4[c]	0	7[k]	696
Historical	89	43	556	1.4	0	7	696
Conterminous United States							
Current	0	93[l]	312[m]	18[c]	3[c]	2[n]	428
Historical	0	111[i]	762[o]	20[p]	4[n]	3[n]	899
Mexico							
Current	0	10[p]	21[p]	0	5[c]	ND[q]	36
Historical	0	45[p]		0	7[h]	ND	52
North America							
Current	511	861	1,047	20	8	15	2,461
Historical	513	894[r]	1,706[r]	23	11	17	3,164
Global							
Current	3,443[s]		2,315[t]	22[u]	181[v]	ND	~6,000
Historical	4,000[w]		5,000[x]	26[y]	ND	ND	~9,000[x]

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

[b]National Wetlands Working Group (National Wetlands Working Group, 1988).

[c]Mendelssohn and McKee (2000).

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

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

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

[g]Rubec (1996).

[h]Assumed same loss rate as the conterminous United States since 1954 (Dahl, 2000).

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

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

[k]Hall (1994).

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

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

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

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

[p]Spiers (1999).

[q]ND indicates that no data are available.

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

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

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

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

[v]Spalding (1997).

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

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

[y]Assumed.

Table 13A-2. Soil carbon pools (Gt) and fluxes (Mt yr^{-1}) of wetlands in North America and the world. "Sequestration in current wetlands" refers to carbon sequestration in extant wetlands; "oxidation in former wetlands" refers to emissions from wetlands that have been converted to non-wetland uses or conversion among wetland types due to human influence; "historical loss in sequestration capacity" refers to the loss in the carbon sequestration function of wetlands that have been converted to non-wetland uses; "change in flux from wetland conversions" is the sum of the two previous fluxes. Positive flux numbers indicate a net flux into the atmosphere, whereas negative numbers indicate a net flux into the ecosystem**.**

	Permafrost peatlands	Non-permafrost peatlands	Mineral-soil freshwater	Salt marsh	Mangrove	Mudflat	**Total**
Canada							
Pool Size in Current Wetlands	44.2[a]	102.9[a]	4.6[b]	0.0[c]	0.0	0.1[d]	151.8
Sequestration in Current Wetlands	-5.5[e]	-13.6[e]	-5.1[f]	-0.1	0.0	-1.2[d]	-25.5
Oxidation in Former Wetlands	0.2[g]		0.0[h]	0.0[i]	0.0	0.0	0.2
Historical Loss in Sequestration Capacity	0.0[e]	0.2[e]	6.5[f]	0.2	0.0	0.3	7.2
Change in Flux From Wetland Conversions	0.4		6.5	0.2	0.0	0.3	7.4
Alaska							
Pool Size in Current Wetlands	9.3[j]	6.2[j]	26.0[k]	0.0	0.0	0.1	41.7
Sequestration in Current Wetlands	-1.1[e]	-0.8[e]	-18.0[f]	-0.3	0.0	-1.6	-21.9
Oxidation in Former Wetlands	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Historical Loss in Sequestration Capacity	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Change in Flux From Wetland Conversions	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Conterminous United States							
Pool Size in Current Wetlands	0	14.0[l]	5.1[k]	0.4	0.1	0.1	19.7
Sequestration in Current Wetlands	0	-11.6[m]	-10.1[f]	-3.9	-0.5	-0.5	-26.6
Oxidation in Former Wetlands	0	18.0[n]	0.0[h]	0.0	0.0	0.0	18.0
Historical Loss in Sequestration Capacity	0	2.1[m]	14.5[f]	0.3	0.0	0.1	17.1
Change in Flux from Wetland Conversions	0	20.1	14.6	0.3	0.0	0.1	35.2
Mexico							
Pool Size in Current Wetlands	0.0	1.5[l]	0.3[k]	0.0	0.1	ND*	1.9
Sequestration in Current Wetlands	0	-1.6[o]	-0.7[f]	0.0	-1.6	ND	-3.9

Oxidation in Former Wetlands	0	ND	ND	0.0	0.0	0.0	ND
Historical Loss in Sequestration Capacity	0	ND	ND	0.0	0.5	ND	0.5
Change in Flux from Wetland Conversions	0	ND	ND	0.0	0.5	ND	0.5
North America							
Pool Size in Current Wetlands	53.5	124.6	36.0	0.4	0.2	0.3	215.1
Sequestration in Current Wetlands	-6.6	-27.6	-33.9	-4.3	-2.1	-3.3	-77.8
Oxidation in Former Wetlands	18.2		0.0	0.0	0.0	0.0	18.2
Historical Loss in Sequestration Capacity	0	2.3	21.0	0.5	0.5	0.5	24.8
Change in Flux from Wetland Conversions	20.5		21.1	0.5	0.5	0.5	43.1
Global							
Pool Size in Current Wetlands	462[p]		46[q]	0.4[r]	5.0[r]	ND	513
Sequestration in Current Wetlands	-55[s]		-75[f]	-4.6[r]	-38.0[r]	ND	-173
Oxidation in Former Wetlands	205[t]		ND	0	0	0	205
Historical Loss in Sequestration Capacity	16[t]		87[f]	0.8[u]	12.7[v]	ND	116
Change in Flux From Wetland Conversions	221[t]		> 87[w]	0.8	12.7	ND	321

*ND indicates that no data are available.

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

[b]Tarnocai (1998).

[c]Rates calculated from Chimura *et al.* (2003); areas from Mendelssohn and McKee (2000).

[d]Assumed the same carbon density and accumulation rates as the adjacent vegetated wetland ecosystem (mangrove data for Mexico and salt marsh data elsewhere).

[e]Assumed carbon accumulation rate of 0.13 Mg C ha^{-1} yr^{-1} for permafrost peatlands and 0.19 Mg C ha^{-1} yr^{-1} non-permafrost peatlands. Reported range of long-term apparent accumulation rates from 0.05-0.35 (Ovenden, 1990; Maltby and Immirzi, 1993; Trumbore and Harden, 1997; Vitt *et al*., 2000; Turunen *et al*., 2004).

[f]Potential rate calculated as the average sediment accumulation rate of 1680 g m^{-2} yr^{-1} (range 0–7840) from Johnston (1991) times 7.7% C (CV = 109) (Batjes, 1996). We assumed that all sequestered soil C was of allochthonous origin and decomposition was 25% slower in wetlands than in the uplands from which the sediment was eroded (see text).

[g]Sum of -0.24 Mt C yr^{-1} from horticulture removal of peat (Cleary *et al.*, 2005) and 0.10 Mt C yr^{-1} from increased peat sequestration due to permafrost melting (Turetsky *et al.*, 2002).

[h]Assumed that the net oxidation of 8.6% of the soil carbon pool (Euliss *et al*., 2006) over 50 yr after conversion to non-wetland use.

[i]Assumed that conversion of tidal systems is caused by fill and results in burial and preservation of SOM define SOM rather than oxidation.

[j]Soil carbon densities of 1,441 Mg C ha^{-1} for Histosols and 1,048 Mg C ha^{-1} for Histels (Tarnocai *et al.*, 2005).

[k]Soil carbon density of 162 Mg C ha^{-1} for the conterminous United States and Mexico and 468 Mg C ha^{-1} for Alaska based upon NRCS STATSGO database and soil pedon information.

[l]Assumed soil carbon density of 1,500 Mg C ha^{-1}.

[m]Webb and Webb (1988).

[n]Estimated loss rate as of early 1980s (Armentano and Menges,1986). Overall wetlands losses in the United States have declined dramatically since then (Dahl, 2000) and probably even more so for Histosols, so this number may still be representative.

[o]Using peat accumulation rate of 1.6 Mg C ha^{-1} (range 1.0–2.25) (Maltby and Immirzi, 1993).

[p]From Maltby and Immirzi (1993). Range of 234 to 679 Gt C (Gorham, 1991; Maltby and Immirzi, 1993; Eswaran *et al*., 1995; Batjes, 1996; Lappalainen, 1996; Joosten and Clarke, 2002).

[q]Soil carbon density of 199 Mg C ha^{-1} (Batjes, 1996).

[r]Chmura *et al.* (2003).

[s]Joosten and Clarke (2002) reported range of -40 to -70 Mt C yr^{-1} . Using the peatland estimate in Table 13A-1 and a C accumulation rate of 0.19 Mg C ha^{-1} yr^{-1}, we calculate a global flux of -65 Mt C yr^{-1} in peatlands.

[t]Current oxidative flux is the difference between the change in flux and the historical loss in sequestration capacity from this table. The change in flux is from Maltby and Immirzi (1993) (reported range 176 to 266 Mt C yr^{-1}) and the historical loss in sequestration capacity is from this table for North America, from Armentano and Menges (1986) for other northern peatlands, and from Maltby and Immirzi (1993) for tropical peatlands.

[u]Assumed that global rates approximate the North America rate because most salt marshes inventoried are in North America.

[v]Assumed 25% loss globally since the late 1800s.

[w]> sign indicates that this a minimal loss estimate.

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

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

*ND indicates that no data are available.

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

[b]Assumed 2000 g C m^{-2} in aboveground and belowground plant biomass (Gorham, 1991).

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

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

[e]50 g C m^{-2} yr^{-1} sequestration from forest growth from a southeastern U.S. regional assessment of wetland forest growth (Brown *et al*., 2001).
[f]Assumed that global pools approximate those from North America because most salt marshes inventoried are in North America.
[g]Twilley *et al*. (1992).

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

	Permafrost peatlands	Non-perma-frost peatlands	Mineral-soil freshwater	Salt marsh	Mangrove	Mudflat	**Total**
Canada							
CH_4 Flux in Current Wetlands	1.1[a]	2.1[b]	5.7	0.0	0.0	0.0[c]	8.9
Historical change in CH_4 Flux	0.0	0.3	-7.2	0.0	0.0	0.0	-6.9
Alaska							
CH_4 Flux in Current Wetlands	0.2	0.1	1.4	0.0	0.0	0.1	1.8
Historical change in CH_4 Flux	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Conterminous United States							
CH_4 Flux in Current Wetlands	0.0	3.4	11.2	0.1	0.0	0.0	14.7
Historical change in CH_4 Flux	0.0	-0.6	-16.2	0.0	0.0	0.0	-16.8
Mexico							
CH_4 Flux in Current Wetlands	0.0	0.4	0.7	0.0	0.0	ND*	1.1
Historical change in CH_4 Flux	0.0	-0.5		0.0	0.0	ND	-0.5
North America							
CH_4 Flux in Current Wetlands	1.3	5.9	19.1	0.1	0.1	0.1	26.5
Historical change in CH_4 Flux	0.0	-24.2		0.0	0.0	0.0	-24.2
Global							
CH_4 Flux in Current Wetlands	14.1[d]	22.5[d]	68.0[d]	0.1[e]	1.4	ND	164[f]
Historical change in CH_4 Flux	-3.6		-79	0.0[g]	-0.5	ND	-83

*ND indicates that no data are available.

[a]Used CH_4 flux of 2.5 g m^{-2} yr^{-1} (range 0 to 130, likely mean 2–3) (Moore and Roulet, 1995) for Canadian peatlands and all Alaskan freshwater wetlands. Used CH_4 flux of 36.0 g m^{-2} yr^{-1} for Canadian freshwater mineral-soil wetlands and all U.S. and Mexican freshwater wetlands and 10.3 g m^{-2} yr^{-1} for estuarine wetlands—from synthesis of published CH_4 fluxes for the United States (see Table 13A-5).

[b]Includes a 17-fold increase in CH_4 flux (Kelly *et al*., 1997) in the 9000 km^2 of reservoirs that have been formed on peatlands (Rubec, 1996) and an estimated CH_4 flux of 15 g m^{-2} yr^{-1} (Moore *et al*., 1998) from 2,630 km^2 of melted permafrost peatlands (Vitt *et al*., 1994).

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

[d]Bartlett and Harriss (1993).

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

[f]Ehhalt *et al.* (2001), range of 92 to 237 Mt yr^{-1}.

[g]Assumed a conservative 25% loss since the late 1800s.

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

Habitat	State	Method[a]	Salt/ Fresh	Daily Average Flux (mg CH_4 m^{-2} d^{-1})	Measured Annual Flux (g CH_4 m^{-2} yr^{-1})	Conversion Factor	Calculated Annual Flux (g CH_4 m^{-2} yr^{-1})	Used Annual Flux (g CH_4 m^{-2} yr^{-1})	Reference
Fens	CO	C	FW		40.7			40.7	Chimner and Cooper (2003)
Wet Alpine Meadow	CO	C	FW	0.1			0.0	0.0	Neff *et al.* (1994)
Lake - Average	CO	C	FW	25.4			9.2	9.2	Smith and Lewis (1992)
Wetland - Average	CO	C	FW	28.3			10.3	10.3	Smith and Lewis (1992)
Nuphar Bed	CO	C	FW	202.1			73.6	73.6	Smith and Lewis (1992)
Tundra - Carex Meadow	CO	C	FW	2.8			1.0	1.0	West *et al.* (1999)
Tundra - Acomastylis Meadow	CO	C	FW	-0.5			-0.2	-0.2	West *et al.* (1999)
Tundra - Kobresia Meadow	CO	C	FW	-0.8			-0.3	-0.3	West *et al.* (1999)
Moist Grassy	CO	C	FW	6.1	1.9	0.32	2.2	1.9	Wickland *et al.* (1999)
Moist Mossy	CO	C	FW	1.5	0.5	0.33	0.5	0.5	Wickland *et al.* (1999)
Wetland	CO	C	FW		41.7			41.7	Wickland *et al.* (1999)
Hardwood Hammock	FL	C	FW	0.0			0.0	0.0	Bartlett *et al.* (1989)
Dwarf Cypress / Sawgrass	FL	C	FW	7.5			2.7	2.7	Bartlett *et al.* (1989)
Spikerush	FL	C	FW	29.4			10.7	10.7	Bartlett *et al.* (1989)
Sawgrass < 1m	FL	C	FW	38.8			14.1	14.1	Bartlett *et al.* (1989)
Sawgrass/Spkerush/Periphyton	FL	C	FW	45.1			16.4	16.4	Bartlett *et al.* (1989)
Swamp Forest	FL	C	FW	68.9			25.1	25.1	Bartlett *et al.* (1989)
Sawgrass > 1m	FL	C	FW	71.9			26.2	26.2	Bartlett *et al.* (1989)
Sawgrass	FL	C	FW	107.0			38.9	38.9	Burke *et al.* (1988)
Pond Open Water	FL	C	FW	624.0			227.1	227.1	Burke *et al.* (1988)
Everglades - Cladium	FL	C	FW	45.4			16.5	16.5	Chanton *et al.* (1993)
Everglades - Typha	FL	C	FW	142.9			52.0	52.0	Chanton *et al.* (1993)
Wet Prairie (Marl)	FL	C	FW	87.0			31.6	31.6	Happell *et al.* (1993)
Wet Prairie (Marl)	FL	C	FW	27.4			10.0	10.0	Happell *et al.* (1993)
Marsh (Marl)	FL	C	FW	30.0			10.9	10.9	Happell *et al.* (1993)
Marsh (Marl)	FL	C	FW	49.6			18.0	18.0	Happell *et al.* (1993)
Marsh (Peat)	FL	C	FW	45.4			16.5	16.5	Happell *et al.* (1993)

Marsh (Peat)	FL	C	FW	13.0			4.7	4.7	Happell *et al.* (1993)
Marsh (Peat)	FL	C	FW	163.6			59.6	59.6	Happell *et al.* (1993)
Marsh (Peat)	FL	C	FW	20.4			7.4	7.4	Happell *et al.* (1993)
Wet Prairie / Sawgrass	FL	C	FW	61.0			22.2	22.2	Harriss *et al.* (1988)
Wetland Forest	FL	C	FW	59.0			21.5	21.5	Harriss *et al.* (1988)
Cypress Swamp - Flowing Water	FL	C	FW	67.0			24.4	24.4	Harriss and Sebacher (1981)
Open Water Swamp	FL	C	FW	480.0			174.7	174.7	Schipper and Reddy (1994)
Waterlily Slough	FL	C	FW	91.0			33.1	33.1	Schipper and Reddy (1994)
Cypress Swamp - Deep Water	GA	C	FW	92.3			33.6	33.6	Harriss and Sebacher (1981)
Bottotmand Hardwoods/ Swamps	GA	C	FW		23.0			23.0	Pulliam (1993)
Swamp Forest	LA	C	FW	146.0			53.1	53.1	Alford *et al.* (1997)
Freshwater Marsh	LA	C	FW	251.0			91.4	91.4	Alford *et al.* (1997)
Fresh	LA	C	FW	587.0	213.0	0.36	213.6	213.0	DeLaune *et al.* (1983)
Fresh	LA	C	FW	49.0	18.7	0.38	17.8	18.7	DeLaune *et al.* (1983)
Sphagnum Bog	MD	C	FW	-1.1			-0.4	-0.4	Yavitt et al. (1990)
Bog	MI	C	FW	193.0			70.2	70.2	Shannon and White (1994)
Bog	MI	C	FW	28.0			10.2	10.2	Shannon and White (1994)
Beaver Meadow	MN	C	FW		2.3			2.3	Bridgham *et al.* (1995)
Open Bogs	MN	C	FW		0.0			0.0	Bridgham *et al.* (1995)
Bog (Forested Hummock)	MN	C	FW	10.0	3.5	0.35	3.6	3.5	Dise (1993)
Bog (Forested Hollow)	MN	C	FW	38.0	13.8	0.36	13.8	13.8	Dise (1993)
Fen Lagg	MN	C	FW	35.0	12.6	0.36	12.7	12.6	Dise (1993)
Bog (Open Bog)	MN	C	FW	118.0	43.1	0.37	42.9	43.1	Dise (1993)
Fen (Open Poor Fen)	MN	C	FW	180.0	65.7	0.37	65.5	65.7	Dise (1993)
Poor Fen	MN	C	FW	242.0			88.1	88.1	Dise and Verry (2001)
Sedge Meadow	MN	C	FW		11.7			11.7	Naiman *et al.* ((1991)
Submergent	MN	C	FW		14.4			14.4	Naiman *et al.* (1991)
Deep Water	MN	C	FW		0.5			0.5	Naiman *et al.* (1991)
Poor Fen	MN	T	FW		14.6			14.6	Shurpali and Verma (1998)
Submerged Tidal	NC	C, E	FW	144.8			52.7	52.7	Kelly *et al.* (1995)
Banks Tidal	NC	C, E	FW	20.1			7.3	7.3	Kelly *et al.* (1995)
Tidal Marsh	NC	C	FW	3.0	1.0	0.34	1.1	1.0	Megonigal and Schlesinger (2002)
Tidal Marsh	NC	C	FW	3.5	2.3	0.65	1.3	2.3	Megonigal and Schlesinger (2002)
Prairie Marsh	NE	T	FW		64.0			64.0	Kim *et al.* (1998)
Poor Fen	NH	C	FW	503.3	110.6	0.22	183.2	110.6	Carroll and Crill (1997)
Poor Fen	NH	C	FW		69.3			69.3	Frolking and Crill (1994)

Forested Peatland	NY	C	FW	0.6	0.2	0.37	0.2	0.2	Coles and Yavitt (2004)
Pools Forested Swamp	NY	C	FW	224.6	69.0	0.31	81.7	69.0	Miller *et al.* (1999)
Typha Marsh - Mineral Soils	NY	C	FW	344.4			125.3	125.3	Yavitt (1997)
Typha Marsh - Peat Soils	NY	C	FW	65.1			23.7	23.7	Yavitt (1997)
Typha Marsh - All soils	NY	C	FW	204.8			74.5	74.5	Yavitt (1997)
Cypress Swamp - Floodplain	SC	C	FW	9.9			3.6	3.6	Harriss and Sebacher (1981)
Swamp	VA	C	FW	470.3			171.2	171.2	Chanton *et al.* (1992)
Maple/gum Forested Swamp	VA	C	FW		0.5			0.5	Harriss *et al.* (1982)
Emergent Tidal Freshwater Marsh	VA	C	FW		96.2			96.2	Neubauer *et al.* (2000)
Oak Swamp (Bank Site)	VA	C	FW	117.0	43.7	0.37	42.6	43.7	Wilson *et al.* (1989)
Emergent Macrophytes (Peltandra)	VA	C	FW	155.0			56.4	56.4	Wilson *et al.* (1989)
Emergent Macrophytes (Smartweed)	VA	C	FW	83.0			30.2	30.2	Wilson *et al.* (1989)
Ash Tree Swamp	VA	C	FW	152.0			55.3	55.3	Wilson *et al.* (1989)
Bog	WA	C	FW	73.0			26.6	26.6	Lansdown *et al.* (1992)
Lowland Shrub and Forested Wetland	WI	T	FW		12.4			12.4	Werner *et al.* (2003)
Sphagnum Eriophorum (Poor Fen)	WV	C	FW	6.6			2.4	2.4	Yavitt *et al.* (1990)
Sphagnum Shrub (Fen)	WV	C	FW	0.1			0.0	0.0	Yavitt *et al.* (1990)
Polytrichum Shrub (Fen)	WV	C	FW	-0.1			0.0	0.0	Yavitt *et al.* (1990)
Sphagnum Forest	WV	C	FW	9.6			3.5	3.5	Yavitt *et al.* (1990)
Sedge Meadow	WV	C	FW	1.5			0.5	0.5	Yavitt *et al.* (1990)
Beaver Pond	WV	C	FW	250.0			91.0	91.0	Yavitt *et al.* (1990)
Low Gradient Headwater Stream	WV	C	FW	300.0			109.2	109.2	Yavitt *et al.* (1990)
Sphagnum-Eriophorum	WV	C	FW	52.1	19.0	0.37	18.9	19.0	Yavitt *et al.* (1993)
Polytrichum	WV	C	FW	41.1	15.0	0.37	15.0	15.0	Yavitt *et al.* (1993)
Sphagnum-Shurub	WV	C	FW	4.4	1.6	0.37	1.6	1.6	Yavitt *et al.* (1993)
Salt Marsh	DE	C	SW	0.5			0.2	0.2	Bartlett *et al.* (1985)
Red Mangroves	FL	C	SW	4.2			1.4	1.4	Bartlett *et al.* (1989)
Dwarf Red Mangrove	FL	C	SW	81.9			27.9	27.9	Bartlett *et al.* (1989)
High Marsh	FL	C	SW	3.9			1.3	1.3	Bartlett *et al.* (1985)
Salt Marsh	FL	C	SW	0.6			0.2	0.2	Bartlett *et al.* (1985)
Salt Water Mangroves	FL	C	SW	4.0			1.4	1.4	Harriss *et al.*(1988)
Salt Marsh	GA	C	SW	13.4			4.6	4.6	Bartlett *et al.* (1985)

Short Spartina Marsh - High Marsh	GA	C	SW	145.2	53.1	0.37	49.5	53.1	King and Wiebe (1978)
Mid Marsh	GA	C	SW	15.8	5.8	0.37	5.4	5.8	King and Wiebe (1978)
Tall Spartina Marsh - Low Marsh	GA	C	SW	1.2	0.4	0.34	0.4	0.4	King and Wiebe (1978)
Intermediate Marsh	LA	C	SW	**912**[b]					Alford *et al.* (1997)
Salt Marsh	LA	C	SW	15.7	5.7	0.36	5.4	5.7	DeLaune *et al.* (1983)
Brackish	LA	C	SW	267.0	97.0		91.1	97.0	DeLaune *et al.* (1983)
Salt Marsh	LA	C	SW	4.8	1.7	0.35	1.6	1.7	DeLaune *et al.* (1983)
Brackish	LA	C	SW	17.0	6.4	0.38	5.8	6.4	DeLaune *et al.* (1983)
Cypress Swamp - Floodplain	SC	C	SW	1.5			0.5	0.5	Bartlett *et al.* (1985)
Salt Marsh	SC	C	SW	0.4			0.1	0.1	Bartlett *et al.* (1985)
Salt Marsh	VA	C	SW	3.0	1.3	0.43	1.0	1.3	Bartlett *et al.* (1985)
Salt Marsh	VA	C	SW	5.0	1.2	0.24	1.7	1.2	Bartlett *et al.* (1985)
Salt Meadow	VA	C	SW	2.0	0.4	0.22	0.7	0.4	Bartlett *et al.* (1985)
Salt Marsh	VA	C	SW	-0.8			-0.3	-0.3	Bartlett *et al.* (1985)
Salt Marsh	VA	C	SW	1.5			0.5	0.5	Bartlett *et al.* (1985)
Salt Meadow	VA	C	SW	-1.9			-0.6	-0.6	Bartlett *et al.* (1985)
Tidal Salt Marsh	VA	C	SW	16.0	5.6	0.35	5.5	5.6	Bartlett *et al.* (1987)
Tidal Brackish Marsh	VA	C	SW	64.6	22.4	0.35	22.0	22.4	Bartlett *et al.* (1987)
Tidal Brackish/Fresh Marsh	VA	C	SW	53.5	18.2	0.34	18.2	18.2	Bartlett *et al.* (1987)
				FW Average =	32.1	0.36	38.6	36.0	
				FW n =	32	18	74	88	
				FW StError=	7.9	0.02	6.0	5.0	
				SW Average =	16.9	0.34	9.8	10.3	
				SW n =	13	12	25	25	

SW StError=	7.8	0.02	4.1	4.4

[a]C = chamber, T = tower, eddy covariance, E = ebulition measured separately.
[b]Outlier that was removed from further analysis.

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