Wetlands

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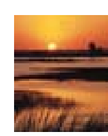
KEY FINDINGS

- North America is home to approximately 40% of the global wetland area, encompassing about 2.5 million square kilometers (965,000 square miles) with a carbon pool of approximately 223 billion tons, mostly in peatland soils.
- North American wetlands currently are a carbon dioxide sink of approximately 49 million tons of carbon per year, but that estimate has an uncertainty of greater than 100%. North American wetlands are also a source of approximately 9 million tons 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 15 million tons of carbon per year, primarily through the oxidation of carbon in peatland soils as they are drained and a more general reduction in carbon uptake and storage 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 carbon dioxide 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 biological diversity, 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 uptake and storage and trace gas fluxes.

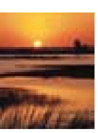


13.1 INTRODUCTION

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 greater than 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 F). 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)[1]. 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 million tons of methane (Mt CH_4) per year[1], which is a large fraction of the total annual global flux of about 600 Mt CH_4 per year (Ehhalt *et al.*, 2001). This is important because CH_4 is a potent greenhouse gas (GHG), second in importance only to carbon dioxide (CO_2) (Ehhalt *et al.*, 2001).

5.5% of the land area of the contiguous United States is wetlands. This represents just 48% of the original wetland area in the conterminous United States.



[1] The uncertainties for the numerical values cited in this chapter are presented and explained in Table 13.1 and Appendix F.

A number of previous studies have examined the role of peatlands in the global carbon balance (reviewed in Mitra *et al.*, 2005), and 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. We limit this review to those components of the carbon budget that result in a net gaseous exchange with the atmosphere on an interannual basis and do not consider other internal carbon fluxes. We do not consider dissolved organic carbon (DOC) fluxes from wetlands, although they may be substantial (Moore, 1997), because the oxidation of the DOC would be counted as atmospheric carbon emissions in the receiving ecosystems downstream and we do not want to double-count fluxes.

Given that many undisturbed wetlands are a natural sink for CO_2 and a source of CH_4, a note of caution in interpretation of our data is important. Using the Intergovernmental 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 GHG 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 CH_4 emissions from wetlands have zero net radiative forcing, but an increase in CH_4 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 soil carbon pool in drained wetlands are both positive radiative forcings.

13.2 INVENTORIES

13.2.1 Current Wetland Area and Rates of Loss

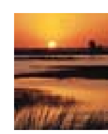
The current and original wetland area and rates of loss are the basis for all further estimates of pools and fluxes in this chapter. The loss of wetlands has caused the oxidation of their soil carbon, particularly in peatlands, reduced their ability to sequester carbon, and reduced their emissions of CH_4. The strengths and weakness of the wetland inventories of Canada, the United States, and Mexico are discussed in Appendix F.

The conterminous United States has 312,000 km² of FWMS wetlands, 93,000 km² of peatlands, and 25,000 km² 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 F.1 in Appendix F).

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 F (see Box 13.1 for uncertainty conventions).

	Area[a] (km^2)	Carbon Pool[b] (Gt C)	Net Carbon Balance[c] (Mt C per year)	Historical Loss in Sequestration Capacity (Mt C per year)	Methane Flux (Mt CH_4 per year)
Canada					
Peatland	1,135,608****	152****	-19***	0.3*	3.2**
Freshwater Mineral	158,720**	4.9**	-2.7*	3.4*	1.2*
Estuarine	6,400***	0.1***	-1.3**	0.5*	0.0***
Total	1,300,728****	157****	-23**	4.2*	4.4*
Alaska					
Peatland	132,196****	15.9**	-2.0**	0.0****	0.3*
Freshwater Mineral	555,629****	27.1**	-9.4*	0.0****	1.4*
Estuarine	8,400****	0.1***	-1.9**	0.0****	0.0***
Total	696,224*****	43.2**	-13*	0.0****	1.7*
Conterminous United States					
Peatland	93,477****	14.4***	5.7*	1.2*	0.7**
Freshwater Mineral	312,193*****	6.2***	-9.8*	7.6*	2.4**
Estuarine	25,000*****	0.6*****	-5.4**	0.5*	0.0***
Total	430,670*****	21.2***	-9.5*	9.4*	3.1**
U.S. Total	1,126,895*****	64.3**	-23*	9.4*	4.8**
Mexico					
Peatland	10,000*	1.5*	-1.6*	ND*	0.1*
Freshwater Mineral	20,685*	0.4*	-0.4*	ND*	0.2*
Estuarine	5,000*	0.2*	-1.6*	1.0*	0.0*
Total	35,685*	2.0*	-3.6*	ND*	0.2*
North America					
Peatland	1,371,281****	184****	-17*	1.5*	4.3**
Freshwater Mineral	1,047,227****	39***	-22*	11*	5.1*
Estuarine	44,800***	0.9***	-10**	2.0*	0.1**
Total	2,463,308****	223****	-49*	15*	9.4*
Global					
Peatland	3,443,000***	462***	150**	16*	37**
Freshwater Mineral	2,315,000***	46***	-39*	45*	68**
Estuarine	203,000*	5.4*	-43*	21*	0.2**
Total	5,961,000***	513***	68*	82*	105**

[a] Estuarine includes salt marsh, mangrove, and mudflat, except for Mexico and global for which no mudflat estimates were available.
[b] Includes soil carbon and plant carbon, but overall soil carbon is 98% of the total pool.
[c] Includes soil carbon sequestration, plant carbon sequestration, and loss of carbon due to drainage of wetlands. Plant carbon 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 F).
ND indicates that no data are available.



BOX 13.1: CCSP SAP 2.2 Uncertainty Conventions

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

However, wetland losses in the United States have declined from 1855 km^2 per year in the 1950s–1970s to 237 km^2 per year in the 1980s–1990s (Dahl, 2000). Such data mask large differences in loss rates among wetland classes and conversion of wetlands to other classes (Dahl, 2000), with potentially large effects on carbon stocks and fluxes. For example, the majority of wetland losses in the United States have occurred in FWMS wetlands. As of the early 1980s, 84% of United States' 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.

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 F.1 in Appendix F). However, given the lack of a nationwide wetland inventory and a general paucity of data, this number is highly uncertain.

North America currently has about 43% of the global wetland area.

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 original wetlands have been converted to other uses (Moser *et al.*, 1996).

13.2.2 Carbon Pools

We estimate that North American wetlands have a current soil and plant carbon pool of 223 billion tons (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 F).

13.2.3 Soil Carbon Fluxes

North American peatlands currently have a net carbon balance of about -17 million metric tons of carbon (Mt C) per year (Table 13.1), but several large fluxes are incorporated into this estimate. (Negative numbers indicate net fluxes into the ecosystem, whereas positive numbers indicate net fluxes into the atmosphere). Peatlands sequester -29 Mt C per year (Table F.2 in Appendix F). However, this carbon sink is partially offset by a net oxidative flux of 18 Mt C per year as of the early 1980s in peatlands in the conterminous United States that have been drained for agriculture and forestry (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 1.5 Mt C per year (Table 13.1), so the overall soil carbon sink of North American peatlands is about 20 Mt C per year 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 CH_4 emissions. Carbon sequestration associated with sediment deposition is a potentially large, but poorly quantified, flux in wetlands (Stallard, 1998; Smith *et al.*, 2001). We estimate that North American FWMS wetlands sequester -18 Mt C per year in sedimentation (Table F.2 in Appendix F). However, as discussed in Appendix F, wetland sedimentation rates are extremely variable. Moreover, almost no studies have placed sediment carbon sequestration in FWMS wetlands in a landscape context, considering allochthonous-derived (from on-site plant production) versus autochthonous-derived (imported from outside the wetland) carbon, replacement of carbon in terrestrial source areas, and differences in decomposition rates between sink and source areas (Stallard, 1998; Harden *et al.*, 1999; Smith *et al.*, 2001). However, it is clear that sedimentation in FWMS wetlands

is a potentially substantial carbon sink and an important unknown in carbon budgets. For example, agriculture typically increases sedimentation rates by 10- to 100-fold and 90% of sediments are stored within the watershed, amounting to about -40 Mt C per year in the conterminous United States (Stallard, 1998; Smith *et al*., 2001). Our estimate of sediment carbon sequestration in FWMS wetlands seems quite reasonable in comparison to within-watershed sediment storage in North America. Moreover, Stallard (1998) and Smith *et al*. (2001) estimated a global sediment sink on the order of -1 Gt C per year.

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 F.2 in Appendix F). However, due to the historical loss of FWMS wetland area, we estimate that they currently sequester 11 Mt C per year 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 extant FWMS wetlands.

We estimate that estuarine wetlands currently sequester -10.2 Mt C per year (Table F.2 in Appendix F), with a historical reduction in sequestration capacity of 2.0 Mt C per year due to loss of area (Table 13.1). However, the reduction is almost certainly greater because our “original” area is only from the 1950s. Despite the relatively small area of estuarine wetlands, they currently contribute about 31% of total wetland carbon sequestration in the conterminous United States and about 18% 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 reasonably robust, but the same ‘landscape’ issues of allochthonous versus autochthonous inputs of carbon, replenishment of carbon in source area soils, and differences in decomposition rates between sink and source areas exist as for FWMS wetlands. Another large uncertainty in the estuarine carbon budget is 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 -49 Mt C per year (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 15 Mt C per year 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 per year from conterminous United States’ peatlands, we estimate that North American wetlands currently sequester 33 Mt C per year less than they did historically (Table F.2 in Appendix F). Furthermore, North American peatlands and FWMS wetlands have lost 2.6 Gt and 0.8 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.

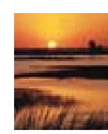
The global wetland soil carbon balance has only been examined in peatlands, which currently are a moderate source of atmospheric carbon of about 150 Mt C per year (Table 13.1), largely due to the oxidation of peat drained for agriculture and forestry and secondarily due to peat combustion for fuel (Armentano and Menges, 1986; Maltby and Immirzi, 1993). The cumulative historical shift in soil carbon stocks has been estimated to be 5.5 to 7.1 Gt C (Maltby and Immirzi, 1993). Although we are aware of no previous evaluation of the carbon balance of global FWMS and estuarine wetlands, using the assumption noted above, we estimate that they are a sink of approximately -39 and -43 Mt per year, respectively.

13.2.4 Methane and Nitrous Oxide Emissions

We estimate that North American wetlands emit 9.4 Mt CH_4 per year (Table 13.1). For comparison, a mechanistic CH_4 model yielded emissions of 3.8 and 7.1 Mt CH_4 per year for Alaska and Canada, respectively (Zhuang *et al*., 2004). A regional inverse atmospheric modeling approach estimated total CH_4 emissions (from all sources) of 16 and 54 Mt CH_4 per year for boreal and temperate North America, respectively (Fletcher *et al*., 2004b).

Despite the relatively small area of estuarine wetlands, they currently contribute about 31% of total wetland carbon sequestration in the conterminous United States and about 18% of the North American total.

Methane emissions are currently about 5 Mt CH_4 per year less than they were historically in North American wetlands (see Table F.4 in Appendix F) 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 CH_4 emissions. Similarly, we estimate that global CH_4 emissions from natural wetlands are only about half of what they were historically due to loss of area (Table F.4 in Appendix F). 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).

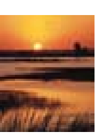


When we multiplied the very low published estimates of nitrous oxide emissions from natural and disturbed wetlands (Joosten and Clarke, 2002) by North American wetland area, the flux was insignificant (data not shown). However, nitrous oxide emissions have been measured in few wetlands, particularly 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 CO_2 and CH_4 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 CO_2 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 CH_4 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.

Historically, the destruction of wetlands through land-use changes has had the largest effect on the carbon fluxes.

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)[2]. 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 approximately neutral in terms of net CO_2-carbon equivalents to the atmosphere (note that we discuss *net radiative forcing* in *Trends and Drivers of Wetland Carbon Fluxes,* Section 13.3). The exception is estuarine wetlands, which are a net sink for CO_2-carbon equivalents because they support both rapid rates of carbon sequestration and low CH_4 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 CH_4 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 CO_2 sequestered as peat (Frolking *et al.*, 2006).



Wetland ecosystems provide havens for biodiversity, recharge of groundwater, and reduction in flooding and fish nurseries.

[2] 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 CH_4 flux columns in Table 13.1 can be directly compared by multiplying CH_4 fluxes by the GWPs given here.

13.2.5 Plant Carbon Fluxes

We estimate that wetland forests in the conterminous United States currently sequester -10.3 Mt C per year as increased plant biomass (see Table F.3 in Appendix F). 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.

13.3 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 CH_4 emissions (a small to large decrease in radiative forcing depending on actual emissions) (Table F.1 and Appendix F). 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 CH_4 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).

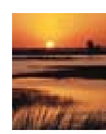
A majority of the effort in examining future global change impacts on wetlands has focused on northern peatlands because of their large soil carbon reserves, although under current climate conditions they have modest CH_4 emissions (Moore and Roulet, 1995; Roulet, 2000; Joosten and Clarke, 2002, and 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 F.2 in Appendix F) 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 per year 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. We did not add this flux to our estimate of the net carbon balance of North American wetlands because historical oxidation of peat by fire should be integrated in the peat sequestration estimates and recent changes due to anthropogenic effects are highly uncertain.

Our compilation shows that attention needs to be directed toward understanding climate change impacts to FWMS wetlands, which collectively emit similar amounts of CH_4 and potentially sequester an equivalent amount of carbon than peatlands. 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 CH_4 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 CH_4 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. Changes in tidal wetland area with sea-level rise will depend on rates of inland migration, erosion at the wetland-estuary boundary, and wetland elevation change. 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 CH_4 emissions from wetlands because of their large GWP. Wetlands emit about 105 Mt CH_4 per year (Table 13.1), or 20% of the global total. Increases in atmospheric CH_4 concentrations over the past century have had the second largest radiative forcing (after CO_2) in human-induced climate change (Ehhalt *et al.*, 2001). Moreover, CH_4 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 CH_4 emissions from wetlands (Fletcher *et al.*, 2004a; Wang *et al.*, 2004; Zhuang *et al.*, 2004).

It is likely that increased rates of sea level rise will cause an overall decline in estuarine marsh area and soil carbon sequestration.

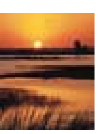


Data (Bartlett and Harriss, 1993; Moore *et al.*, 1998; Updegraff *et al.*, 2001) and modeling (Gedney *et al.*, 2004; Zhuang *et al.*, 2004) strongly support the contention that water table position and temperature are the primary environmental controls over CH_4 emissions. How this generalization plays 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 CH_4-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°C increase in air temperature (Oechel *et al.*, 1994). It was hypothesized that soil warming relieved a severe nutrient limitation on photosynthesis by increasing nitrogen mineralization.

A consistent response to elevated CO_2-enhanced photosynthesis in wetlands is an increase in CH_4 emissions ranging from 50 to 350% (Megonigal and Schlesinger, 1997; Vann and Megonigal, 2003). It is generally assumed that the increased supply of plant photosynthate stimulates anaerobic microbial carbon metabolism, of which CH_4 is a primary end product. An increase in 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 GHG (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 (*i.e.*, increased labile carbon inputs from elevated CO_2 enhance decomposition of the overall soil carbon pool) (Hoosbeek *et al.*, 2004; Lichter *et al.*, 2005). Elevated CO_2 has the potential to influence the carbon budgets of adjacent aquatic ecosystems by increasing export of dissolved organic carbon (Freeman *et al.*, 2004) and dissolved inorganic carbon (Marsh *et al.*, 2005).



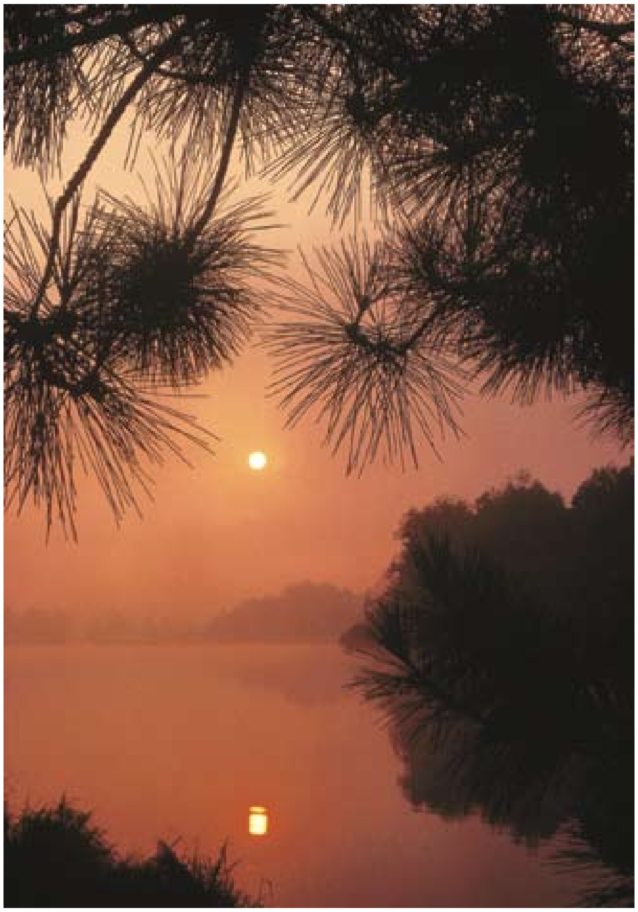
Other important anthropogenic forcing factors that will affect future CH_4 emissions include atmospheric sulfate deposition (Vile *et al.*, 2003; Gauci *et al.*, 2004) and nutrient additions (Keller *et al.*, 2005). These external forcing factors in turn will interact with internal ecosystem constraints 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.

13.4 OPTIONS FOR MANAGEMENT

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 CH_4 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. Decreases in the rates of loss of all wetlands have helped to maintain their soil carbon sequestration potential.

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 CH_4 emissions from conterminous United States' 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 CH_4 compared to the carbon they sequester. Restoration of drained peatlands may stop the rapid loss of their soil carbon, which may compensate for increased CH_4 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,



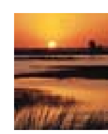
only 4157 km^2 of uplands were converted into wetlands in the conterminous United States (Dahl, 2000). Using the soil carbon sequestration rate of 3.05 Mg C per hectare per year found by Euliss *et al.* (2006) for restored prairie pothole wetlands[3], we estimate that wetland restoration in the United States would have sequestered 1.3 Mt 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 per year through 2045 (Barker *et al*., 1996). Euliss *et al.* (2006) estimated that if all cropland on former prairie pothole wetlands in the United States and Canada (162,244 km^2) were restored that 378 Mt C would be sequestered over 10 years in soils and plants. However, neither study accounted for the GWP of increased CH_4 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, primarily because of storage as peat, and are a small-to-moderate carbon sink (excluding CH_4 effects). The largest unknown in the wetland carbon budget is the amount and significance of sedimentation in FWMS and estuarine wetlands, and CH_4 emissions in freshwater wetlands. With the exception of estuarine wetlands, CH_4 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 CH_4 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 additional measurements of annual CH_4 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.

Larger areas of wetland restoration may have a significant impact on carbon sequestration, but may also increase methane emissions offsetting any positive greenhouse gas effects.



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

[3] 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 ($r^2 = 0.04$, $P = 0.241$).

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 CO_2 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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