13 Terrestrial Wetlands

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KEY FINDINGS

1. The assessment of terrestrial wetland carbon stocks has improved greatly since the *First State of the Carbon Cycle Report* (CCSP 2007) because of recent national inventories and the development of a U.S. soils database. Terrestrial wetlands in North America encompass an estimated 2.2 million km^2, which constitutes about 37% of the global wetland area, with a soil and vegetation carbon pool of about 161 petagrams of carbon that represents approximately 36% of global wetland carbon stock. Forested wetlands compose 55% of the total terrestrial wetland area, with the vast majority occurring in Canada. Organic soil wetlands or peatlands contain 58% of the total terrestrial wetland area and 80% of the carbon (*high confidence, likely*).

2. North American terrestrial wetlands currently are a carbon dioxide sink of about 123 teragrams of carbon (Tg C) per year, with approximately 53% occurring in forested systems. However, North American terrestrial wetlands are a natural source of methane (CH_4), with mineral soil wetlands emitting 56% of the estimated total of 45 Tg C as CH_4 (CH_4 –C) per year (*medium confidence, likely*).

3. The current rate of terrestrial wetland loss is much less than historical rates (about 0.06% of the wetland area from 2004 to 2009), with restoration and creation nearly offsetting losses of natural wetlands. Although area losses are nearly offset, there is considerable uncertainty about the functional equivalence of disturbed, created, and restored wetlands when comparing them to undisturbed natural wetlands. Correspondingly, there remains considerable uncertainty about the effects of disturbance regimes on carbon stocks and greenhouse gas (GHG) fluxes. For this reason, studies and monitoring systems are needed that compare carbon pools, rates of carbon accumulation, and GHG fluxes across disturbance gradients, including restored and created wetlands. Those studies will produce data that are needed for model applications (*high confidence, likely*).

Note: Confidence levels are provided as appropriate for quantitative, but not qualitative, Key Findings and statements.

13.1 Introduction

The objective of this chapter is to characterize the distribution of carbon stocks and fluxes in terrestrial wetlands within North America. The approach was to synthesize available literature from field measurements with analyses of resource inventory data to estimate wetland area, carbon stocks, and net ecosystem exchange (NEE) of carbon and methane (CH_4) fluxes of terrestrial wetlands (see Appendices 13A, p. 547, and 13B, p. 557, for details[1]). Then, the findings employed from large-scale simulation studies provided additional context, with consideration given to the effects of disturbance regimes, restoration and creation of terrestrial wetlands, and the application of modeling tools to assess the carbon cycle of terrestrial wetlands.

13.1.1 Terrestrial Wetland Definition

This chapter focuses on carbon cycling in nontidal freshwater wetlands (referred to hereafter as "terrestrial wetlands"). Although there are various definitions of terrestrial wetlands (Cowardin et al., 1979; IUSS Working Group WRB 2006), all recognize a high water table level as the driver of biological and chemical processes characteristic of wetlands. The United States defines wetlands as soils that are inundated or saturated by surface or groundwater at a frequency and duration sufficient to support, and that do support under normal circumstances, a prevalence of vegetation typically adapted for life in saturated conditions (U.S. EPA 2015). The distribution of U.S. wetlands is considered on the basis of vegetation and hydrogeomorphical setting

[1] The assessment described in this chapter required additional background and parallel analyses of recently published and accessible databases. These analyses pertain only to Ch. 13 and are presented in Appendices 13A and 13B, beginning on p. 547.

using remote-sensing data (Federal Geographic Data Committee 2013). Soils are also indicative of wetland conditions; two major soil types useful for assessing carbon stocks and fluxes recognized here are mineral soils and organic soils. Wetland ecosystems with organic soils, also known as peatlands, are classified as Histosols by the U.S. Department of Agriculture (USDA) Natural Resources Conservation Service (NRCS) Soil Survey (Soil Survey Staff 2010). The Histosol order represents soils with a thick (>40-cm) accumulation of organic matter on top of mineral sediments or rock. Most Histosols are formed under wet conditions (e.g., peat soils), but some of these soils form under aerated conditions. Not considered a wetland, aerated Histosols are distinctly recognized (e.g., suborder Folists) and thus are not considered here. However, all peatlands are formed under wet conditions (Joosten and Clarke 2002), and they are classified as wetlands in Canada (Zoltai and Vitt 1995) and throughout North America (Gorham et al., 2012). The amount and distribution of accumulated soil organic matter reflect the balance between inputs from vegetative production and losses from decomposition or overland transport (e.g., erosion or drainage). While the depth for defining organic soils (Histosols) or peatlands ranges from 10 to 50 cm among different countries, the USDA Soil Survey uses the top 40 cm in the upper 80 cm of soil, which is the definition used here (Soil Survey Staff 2010). Mineral soil wetlands vary widely in the composition and depth of the surface organic layer, varying from a few centimeters to nearly 40 cm in histic-mineral soil wetlands ("histic" refers to soils with a 20- to 40-cm organic horizon, differentiating them from Histosols).

13.1.2 Relationship to Other Chapters and SOCCR1

For this chapter, assessments were made of terrestrial wetlands that occur in boreal, temperate, and tropical climatic zones in Canada, the United States, Mexico, and Puerto Rico. Tidally influenced saltwater and freshwater wetlands are assessed in Ch. 15: Tidal Wetlands and Estuaries, p. 596. Terrestrial wetlands, including peatlands, occurring in the Arctic permafrost zone are assessed in Ch. 11: Arctic and Boreal Carbon, p. 428. Some types of wetlands are transition zones to inland waters (e.g., riparian wetlands). This report considers that inland waters (see Ch. 14: Inland Waters, p. 568) begin at the shoreline of lake, reservoir, and fluvial systems. Both Ch. 9: Forests, p. 365, and this chapter use the definition of forests from the USDA Forest Service's Forest Inventory and Analysis (FIA). As a result, there is overlapping data between Ch. 9 and this chapter. Also, Ch. 10: Grasslands, p. 399, describes wetlands in those domains and thus has some overlapping data with this chapter. Similarly, there are overlapping data with Ch. 12: Soils, p. 469, where organic and mineral soil wetlands are assessed. Since Ch. 5: Agriculture, p. 229, includes no jurisdictional wetlands, it does not have overlapping data.

In the *First State of the Carbon Cycle Report* (SOCCR1; CCSP 2007), the Wetlands chapter (Chapter 13; Bridgham et al., 2007) was inclusive of all terrestrial and tidal wetlands, from tropical to Arctic ecosystems. In the *Second State of the Carbon Cycle Report* (SOCCR2), wetlands are assessed in several chapters as described above.

This chapter adds new information on carbon pools and fluxes from terrestrial wetlands that occur in boreal, temperate, and tropical climate zones within North America. It breaks down carbon pools and fluxes between mineral soil wetlands and peatland ecosystems. It also differentiates carbon pools and fluxes between forested and nonforested wetlands (not done in SOCCR1) because of the influence of trees on ecosystem carbon dynamics (see Figure 13.1, p. 510). The term "flux" is used for carbon dioxide (CO_2) and CH_4 as the net balance between uptake and release of these gases relative to the atmosphere. Finally, this chapter reviews dissolved organic carbon (DOC) fluxes from terrestrial wetlands as well as restored wetlands, but it does not consider constructed wetlands or detention ponds, which typically are engineered systems.

Figure 13.1. Forested Peatland in Northern Minnesota. This bog is part of the U.S. Department of Agriculture (USDA) Forest Services's Marcell Experimental Forest. [Figure source: USDA Forest Service.]

13.2 Current and Historical Context

13.2.1 Wetland Regulations

During the settlement of North America, wetlands were viewed as unproductive areas that were impediments to transportation and development, as well as a breeding ground for disease. That sentiment lasted for over 150 years, during which draining of wetlands for agriculture, forestry, and urban development was routine to make these ecosystems productive for commercial use. Once drained, wetlands generally have very productive soils because of their high organic matter and associated nutrients. Not until the mid-1900s did the effects of wetland drainage on both inherent wetland values and larger landscape impacts begin to be identified. Wetlands are now known to provide critical habitats for many rare species, serve as filters for pollutants and sediment, store water to prevent flooding, and sequester and store carbon, but those ecosystem services were not broadly recognized until relatively recently.

Currently, vegetation removal, surface hardening (e.g., pavement and soil compaction), and drainage are identified as the most common physical stressors on U.S. wetlands (U.S. EPA 2016). To address the threats and subsequent losses of wetlands, wetland policies have been developed to avert further wetland conversion, degradation, or loss. The United States has an overarching policy of "no net loss" of wetlands adopted in 1989. This policy has dramatically slowed U.S. wetland losses and led to the development of wetland banking programs whereby losses due to development are offset by wetlands restored or created elsewhere. In Canada, the main causes for wetland losses are from land conversion to urban or agriculture, water-level control including flooding from hydroelectric development, and climate change (Federal Provincial and Territorial Governments of Canada 2010). In 1991, the Canadian government enacted the Federal Policy on Wetland Conservation (Canadian Wildlife Service 1991). Similarly, the Natural Protected Areas Commission of Mexico announced a national wetland policy in 2014 designed to protect wetlands and avert losses. Recent research in Mexico indicates that drainage for agriculture and conversion to aquaculture are two major threats to wetlands (De Gortari-Ludlow et al., 2015).

These national-level policies are not the only regulations in place designed to protect wetlands. The United States and Canada have wetland-focused state and provincial regulations, as well as other federal regulations that, while not focused on wetlands, do protect wetland habitat. Migratory bird agreements among the United States, Mexico, and Canada often have wetland protection implications. In 1986, the United States and Canada adopted the North American Waterfowl Management Plan and were later joined by Mexico in 1994 (North American Waterfowl Management Plan Committee 2012). This plan establishes strategies to protect wetland habitat for the primary purpose of sustaining migratory bird populations with the associated benefit of protecting carbon pools.

Competing land uses and economic development will continue to threaten wetlands in North America. Multiple policies have been designed to protect against, and mitigate for, wetland loss. However, while losses are greatly stemmed, the United States continues to experience net losses of wetlands in

terms of absolute acreage in spite of the no net-loss policy. Canada and Mexico currently have no nationwide wetlands inventory, limiting the ability to estimate wetland conversion or function, including carbon fluxes and pools. It is important to remember that no net-loss policies do not protect against reduced functionality in restored versus natural wetlands.

13.2.2 Change in Wetland Area

As a result of socioeconomic drivers, there have been massive disturbances and conversions of wetlands over the past 150 or more years in North America. The latest assessment of the status and trends of wetlands in the conterminous United States (CONUS) estimates that there are 445,000 km^2 of wetlands, which includes 395,197 km^2 of terrestrial wetlands (USFWS 2011). In colonial America, there were an estimated 894,000 km^2; between 1870 and 1980, the United States experienced a 53% loss of wetland area (Dahl 1990). From 2004 to 2009, increased wetland restoration on agricultural lands occurred; however, wetland losses continued to outpace gains, leading to a total wetland area decline of 0.06% (USFWS 2011). The current rate of loss is 23 times less than that of the historical trend (e.g., 1870 to 1980), an indication of changing attitudes toward wetlands and the effectiveness of policies to protect them (USFWS 2011).

Although Canada does not have a national wetlands inventory, estimated losses are approximately 14% of the country's original 1,470,000 km^2 of wetlands (Environment Canada 1991). Similarly, an estimated 62% of wetland area has been lost from Mexico's original 112,166 km^2 of wetlands (Casasola 2008; Landgrave and Moreno-Casasola 2012). Mexico's small area of peatlands covers about 20,000 km^2 generally found in high-elevation ecosystems and near-coastal freshwater marshes (Instituto Nacional de Estadística y Geografía 2010). The country has another 15,000 km^2 of mineral soil wetlands.

In CONUS, about 468,000 km^2 of wetlands have been lost, 96% of which have been mineral soil wetlands and 4% peatlands (Bridgham et al., 2007). Similarly, in Canada, of the 212,000 km^2 of wetlands lost, 94% have been mineral soil wetlands and 6% peatlands (Bridgham et al., 2007). However, Canadian peatlands are now being lost in large numbers due to urban development, hydroelectric development, and energy production (Chimner et al., 2016), including in the oil sands region where nearly 300 km^2 have been destroyed by mining (Rooney et al., 2012). In the United States, forested wetlands are undergoing the most rapid losses among terrestrial wetland types. From 2004 to 2009, 1.2% of forested wetlands were lost (2,562 km^2) per year, compared to gains of 1,084 km^2 per year for emergent wetlands and 729 km^2 per year for shrub wetlands (Dahl 2011).

The change in wetland area is quite high in the U.S. Midwest where Iowa, Missouri, Illinois, Ohio, and Indiana have experienced a greater than 85% loss of their wetlands. California has lost 96% of its original wetlands (Dahl 2011; Garone 2011). Other notable ecosystem examples include bottomland hardwood forests of the Lower Mississippi River Alluvial Plain (i.e., southern Illinois to the Gulf of Mexico); these forests, once comprising an area of approximately 85,000 km^2, were reduced to about 20,000 km^2 by 1990, primarily through agricultural conversion and alterations to the hydrological system for flood protection (Stanturf et al., 2000). Major federal flood-control projects that began following a significant flood in 1927 contributed to more than 30% of wetland losses and subsequent agricultural conversions in the Mississippi River Valley (King et al., 2006; Stavins and Jaffe 1990). Similarly, the Prairie Pothole Region (see Section 13.3.3, p. 520) of the United States and Canada included 200,000 km^2 of wetland area prior to European settlement but has since decreased to 70,000 km^2 of intact (i.e., not drained) wetland area (Dahl 2014; Euliss et al., 2006). In contrast, Alaska is reported to have had negligible wetland loss (Bridgham et al., 2007), although the state does not have a completed assessment under the U.S. Fish and Wildlife Service (USFWS) National Wetlands Inventory.

Areal extent alone does not indicate the ecosystem function and services that wetlands deliver. In 2011, the U.S. Environmental Protection Agency (EPA) released the first national assessment of the condition of U.S. wetlands. Findings indicated that 48% of wetlands were in good condition, 20% were in fair condition, and 32% were in poor condition (U.S. EPA 2016). While wetlands may remain intact, their alterations by humans are still affecting the ability of wetlands to function similarly to an unaltered state. Carbon sequestration is one of those important functions affected by wetland condition. Connecting wetland condition to carbon stocks and fluxes will be an important next step for assessing impacts on the carbon cycle.

13.2.3 Overview of Disturbance Effects on Carbon Stocks and Fluxes

Wetlands have been sequestering carbon from the atmosphere for thousands of years. Following the end of the last glacial period about 12,000 years ago, wetlands developed over much of the northern part of North America. Low areas or areas with less permeable soils tended to pond water and create the anoxic environment critical for peatland and mineral soil wetland formation. In undisturbed wetlands, carbon pools are relatively stable over short time intervals, but carbon fluxes may be quite variable due to complex interactions of climate, vegetation, soils, and hydrology. For example, annual CO_2 fluxes ranged from a sink of 2 to 112 grams of carbon (g C) per m^2 per year, and CH_4 fluxes ranged from a source of 2.8 to 4.4 g C per m^2 per year during a 6-year study in a peatland in southern Ontario (Roulet et al., 2007). Carbon dioxide fluxes generally decrease (i.e., sinks or lesser sources) and CH_4 fluxes generally increase (i.e., sources or lesser sinks) as water tables get nearer to the surface (Olson et al., 2013). During droughts or high-water events, CO_2 and CH_4 fluxes can vary greatly, even in undisturbed wetlands. Changes in carbon fluxes resulting from disturbance lead to changes in carbon pools. Drainage is the main human-caused disturbance that has led to a variety of local- to landscape-level impacts. Wetland drainage causes an abrupt change from anaerobic conditions during flooding to aerobic conditions subsequent to drainage, resulting in rapid acceleration of decomposition through microbial oxidation of organic matter (Drexler et al., 2009). As a result, wetland drainage generally leads to lower carbon stocks, lower CH_4 fluxes, and a long-term increase in CO_2 fluxes (Bridgham et al., 2006). In peatlands, drainage also can result in significant land-surface subsidence (Drexler et al., 2009). Other human-caused disturbances include filling of wetlands for development, construction of dams that permanently flood wetlands, stream channelization and road construction that can disconnect wetlands from their water source, removal of vegetation (including forest harvesting), and agricultural conversion of surrounding uplands.

13.3 Current Understanding of Wetland Stocks and Fluxes

The occurrence of the water table within the upper soil layers during the growing season differentiates wetlands from upland ecosystems, influencing the biological communities that must adapt to withstand prolonged periods of soil saturation and biogeochemical processes that are a function of the anoxic soil conditions. While net primary production (NPP) of wetlands is comparable to upland ecosystems (Ahl et al., 2004), the rate of organic matter decomposition is generally less due to the anaerobic soil conditions. As a result, wetland soils typically contain considerably more carbon per unit volume than do upland soils. In areas with prolonged periods of soil saturation and high rates of organic matter production, organic matter may accumulate on top of the mineral substrate, forming organic soils or peatlands with thicknesses ranging from 40 cm to many meters.

The anaerobic conditions of wetland soils also influence greenhouse gas (GHG) fluxes. Unlike upland soils that generally are a sink for atmospheric CH_4, wetland soils typically are a net source of CH_4 to the atmosphere. Methane flux from wetlands is regulated largely by oxygen availability and associated water table position, soil temperature, and vegetation type (Bansal et al., 2016; Green and Baird 2012; Hanson et al., 2016). Hence, fluxes can

be highly variable, even within a wetland, as subtle differences in surface topography, temperature gradients, and vegetation affect fluxes (Bridgham et al., 2006). Accordingly, carbon fluxes and storage in wetlands are likely to change dramatically as a result of climate and land-use changes, which alter water-table dynamics, temperatures, and vegetation communities, ultimately affecting the ecosystem carbon balance. Drainage is the common modification to wetlands for agriculture and silviculture and causes most of the wetland loss noted above. The organic matter decomposition rates of those drained wetlands can be very high, and, for peatlands, the effect may persist for many decades. The soil carbon content of converted wetlands may be greater than the surrounding upland, while the fluxes of GHGs, especially CO_2, are likely larger.

This chapter assessed the state of the wetland carbon cycle, considering organic and mineral soils separately because the soil carbon density, or the amount of carbon per unit volume, varies between the two soil types, and they generally reflect different hydrological settings and vegetation communities. Correspondingly, differentiating between forested and nonforested organic and mineral soil wetlands provides a basis to consider the influence of vegetation on the carbon cycle. The approach for quantifying the wetland carbon pools was based primarily on analyses of recently developed geospatial data, providing a more robust basis for the assessment, as contrasted with summarization based on studies reported in the literature. The general framework, using CONUS as an example, consisted of identifying the distribution of forested and nonforested terrestrial wetlands using the USFWS National Wetlands Inventory. The soil carbon stocks were then determined by summarizing USDA's NRCS Soil Survey databases. Forest vegetation carbon stocks were estimated based on the U.S. Forest Service FIA database (U.S. Forest Service 2003), and nonforest vegetation carbon content was estimated using a mean carbon density based on reported values in the literature. Variations to that framework were necessitated by available databases. For example, in Alaska, where the National Wetlands Inventory has not been completed, a remote sensing–based approach to wetland identification was used (Clewley et al., 2015). Similarly, because Canada does not have a comprehensive national soil inventory, independent assessments of Canadian peatlands and soil landscapes were used. Details about the databases used to calculate the wetland area and associated carbon stocks are provided in Appendix 13A, p. 547.

There are approximately 2.2 million km^2 of terrestrial wetlands in North America (see Table 13.1, p. 514); the majority of those wetlands (81%) occurs in Canada and Alaska. This estimate is approximately 176,000 km^2 less than the one used in SOCCR1 (CCSP 2007). The difference in nonpermafrost peatlands and freshwater mineral soil wetlands among the two reports is due primarily to a smaller and more accurate and current assessment of wetland area in Alaska (Clewley et al., 2015), which reduced the total wetlands in the state by approximately 360,000 km^2; Canadian wetlands increased by approximately 198,000 km^2 due primarily to a larger estimate of mineral soil wetlands. The uncertainty in wetland area is greatest at the higher latitudes, hence the reliance on remote-sensing methods for spatial extent estimates, which are expected to improve further as data and processing tools advance. The report on Alaskan wetlands by Clewley et al. (2015) is an example of achieving an accuracy of approximately 94% in discriminating wetlands from uplands. There remains uncertainty in the reported area of Canadian peatlands, which ranges from the 755,000 km^2 reported by Kroetsch et al. (2011) to the 1.1 million km^2 reported in SOCCR1 (Bridgham et al., 2007). In contrast to reported inventories and assessments used in SOCCR1, Zhang et al. (2017a) used six models to estimate wetland area for North America (including coastal wetlands), with the modeled estimates ranging from about 1.1 to 3.3 million km^2, effectively placing the estimated total in Table 13.1 in the middle of that range. Correspondingly, there are large ranges in estimated global wetland area. Based on modeled and observational estimates (Bridgham et al., 2006; Melton et al., 2013; Zhang et al., 2017a), North

Table 13.1. Area, Carbon Pool, Net Ecosystem Exchange of Carbon, and Methane Emissions from Wetlands in North America[a–c]

Wetland Type	Area[d] (km^2)	Carbon Pool[e] (Pg C)	NEE[f]	CH_4 Emissions	
			Net Balance (Tg C per Year)[g]	CH_4-C (Tg C per Year)[g]	CH_4 (Tg per Year)
Canada					
Peatland					
Nonforested	415,450	37.8	–6.9 ± 3.5	9.4 ± 2.4	12.6
Forested	703,785	76.7	–33.6 ± 5.9	6.3 ± 7.4	8.4
Mineral					
Nonforested	103,932	9.5	–10.6 ± 7.2	2.7 ± 0.7	3.6
Forested	268,337	5.1	–12.9 ± 6.8	7.2 ± 4.3	9.6
Total	1,491,504	129.0	–64.0 ± 12.0	25.6 ± 8.9	34.2
Conterminous United States					
Peatland					
Nonforested	42,903	3.9	–5.8 ± 3.6	1.0 ± 0.3	1.3
Forested	40,823	4.4	–4.9 ± 3.8	0.4 ± 0.4	0.5
Mineral Soil					
Nonforested	138,381	1.9	–14.1 ± 9.5	3.6 ± 1.0	4.8
Forested	173,091	3.3	–11.6 ± 8.2	4.7 ± 2.8	6.2
Total	395,197	13.5	–36.5 ± 13.6	9.6 ± 3.0	12.8
Alaska					
Peatland					
Nonforested	73,836	5.5	–4.2 ± 4.7	1.7 ± 0.4	2.2
Forested	5,747	0.4	–0.3 ± 0.4	0.1 ± 0.1	0.2
Mineral Soil					
Nonforested	192,013	9.3	–10.9 ± 12.3	5.0 ± 1.4	6.7
Forested	40,162	2.0	–2.3 ± 2.6	1.1 ± 0.6	1.4
Total	311,758	17.3	–17.6 ± 13.5	7.9 ± 1.6	10.5
Puerto Rico					
Peatland					
Nonforested	8	0.001	–0.003 ± 0.003	3.38E-04[h] ± 2.88E-04	0.0
Forested	1	0.000	0.000 ± 0.000	2.68E-05 ± 2.28E-05	0.0
Mineral Soil					
Nonforested	252	0.006	–0.030 ± 0.110	1.36E-02 ± 0.488E-02	0.0
Forested	50	0.001	–0.006 ± 0.022	2.70E-03 ± 0.966E-03	0.0
Total	311	0.008	–0.039 ± 0.110	1.67E-02 ± 0.500E-02	2.22E-02
Mexico					
Peatland					
Nonforested	17,191	0.43	–5.33 ± 5.25	0.69 ± 0.59	0.9
Forested	3,394	0.24	–1.05 ± 1.04	0.14 ± 0.12	0.2

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Table 13.1. Area, Carbon Pool, Net Ecosystem Exchange of Carbon, and Methane Emissions from Wetlands in North America[a–c]

Wetland Type	Area[d] (km^2)	Carbon Pool[e] (Pg C)	NEE[f]	CH_4 Emissions	
			Net Balance (Tg C per Year)[g]	CH_4-C (Tg C per Year)[g]	CH_4 (Tg per Year)
Mexico (continued)					
Mineral Soil					
Nonforested	10,320	0.35	−1.25 ± 4.51	0.56 ± 0.20	0.7
Forested	5,288	0.16	−0.64 ± 2.31	0.29 ± 0.10	0.4
Total	36,193	1.17	−8.27 ± 7.37	1.67 ± 0.640	2.22
North America					
Peatland					
Nonforested	549,388	47.7	−22.2 ± 17.1	12.8 ± 3.7	17.0
Forested	753,749	81.8	−39.9 ± 11.0	6.9 ± 8.0	9.2
Mineral Soil					
Nonforested	444,898	21.1	−36.9 ± 33.6	11.9 ± 3.3	15.9
Forested	486,928	10.4	−27.4 ± 19.9	13.3 ± 7.8	17.7
Total	2,234,963	161.0	−126.4 ± 23.8	44.8 ± 9.5	59.8

Notes

a) Positive emissions indicate net gains to the atmosphere, and negative emissions indicate net gains or sequestration into the ecosystem.

b) Citations and assumptions in calculations are in the text of this chapter and in Appendices 13A, p. 547, and 13B, p. 557.

c) Key: C, carbon; NEE, net ecosystem exchange; CH_4, methane; Pg C, petagrams of carbon; Tg C, teragrams of carbon.

d) Includes freshwater and nontidal terrestrial wetlands. Accuracy of wetland area estimates: Canada: >66% (Tarnocai 2009), conterminous United States: >90% (Nichols 1994), Alaska: 95% (Clewley et al., 2015), Puerto Rico: >90% (Nichols 1994), Mexico: <75% (this report); see Appendix 13A, p. 547, for more information.

e) Includes soil and plant carbon; soil carbon accounts for approximately 93% of the total pool.

f) Includes net exchange of CO_2 from the wetland; it does not include lateral fluxes or CH_4 fluxes.

g) The values here are mean values plus or minus 2 times the standard errors to approximate the minimum and maximum values of a 95% confidence interval.

h) E = 10x.

America contains 20% to 47% of the global wetland area, depending on the basis.

The dominant carbon flux from terrestrial wetlands is characterized as NEE of CO_2, which is a measure of the difference in CO_2 uptake and CO_2 release; NEE is positive when the net flux is from the wetland to the atmosphere. In addition to NEE of CO_2, this chapter also reports CH_4 fluxes from the wetlands. Estimates of these fluxes are based on studies reported in SOCCR1 (CCSP 2007) and subsequent literature that used field-based measurements to estimate NEE and CH_4 fluxes (either chamber based or eddy covariance). This chapter categorizes the studies by soil, vegetation type, and region and utilizes a mean flux as the basis for the flux density (flux per unit area) used in the reported regions (see Appendix 13B, p. 557, for flux density factors used in the analyses). Though NEE and CH_4 fluxes are the primary fluxes considered, the wetland net ecosystem carbon balance (Chapin et al., 2006), which is the overall net change in wetland carbon

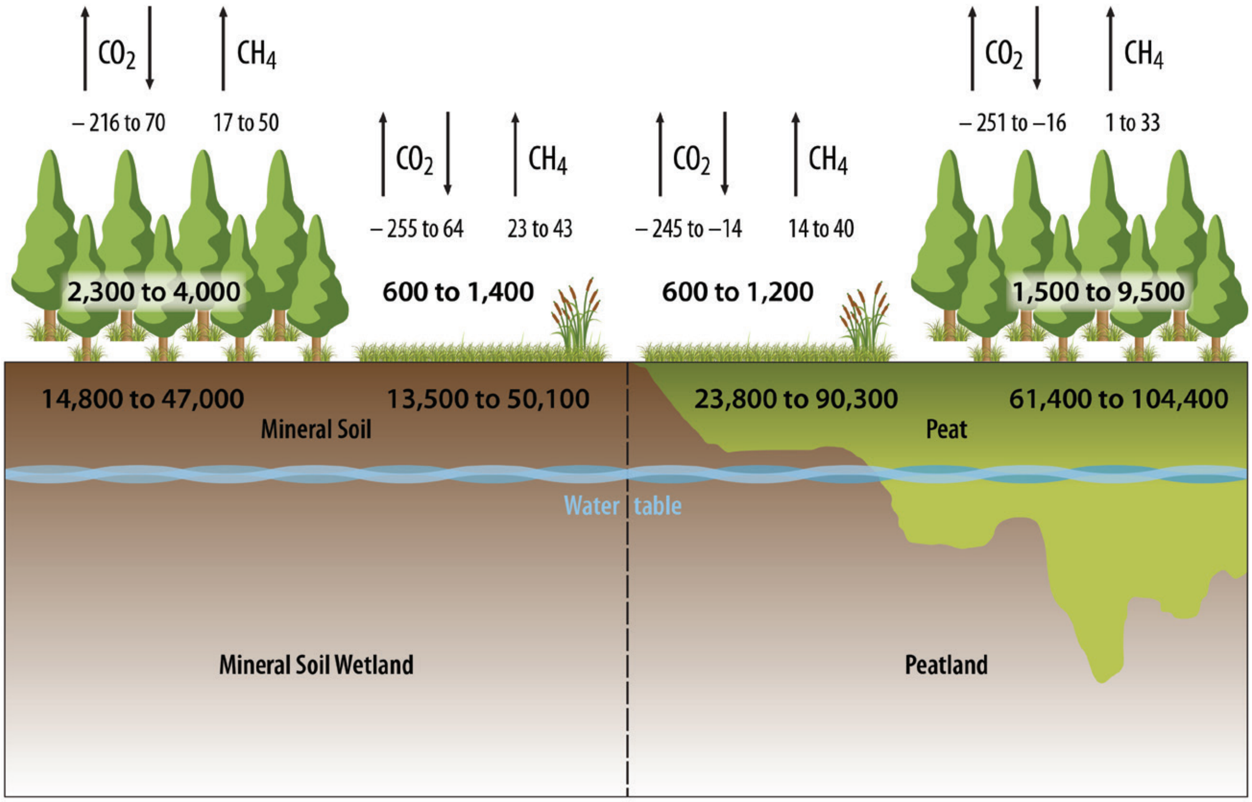

Figure 13.2. Carbon Pools and Fluxes in Forested and Nonforested Mineral Soil Wetlands and Peatlands in North America. The soil and vegetation carbon pools are represented by the range of carbon densities (minimum to maximum) among Canada, Mexico, and the United States. Annual carbon dioxide (CO_2) and methane (CH_4) fluxes (arrows) are represented by a 95% confidence interval; a negative flux indicates a transfer of carbon from the atmosphere to the ecosystem. Stocks and fluxes are in grams of carbon (g C) per m^2. [Data sources: Table 13.1, p. 514, and Appendices 13A and 13B, p. 547 and p. 557, respectively.]

over a specified time, is also influenced by other fluxes. These additional fluxes include carbon monoxide and volatile organic carbon to the atmosphere (e.g., from fires), lateral fluxes of DOC (see Section 13.3.3, p. 520), dissolved inorganic carbon (DIC), and particulate carbon (Chapin et al., 2006).

Peatlands tend to store more soil carbon than mineral soil wetlands, and forested wetlands store more carbon in the vegetation than nonforested wetlands (see Figure 13.2, this page). Across all studies used in this chapter's analysis, fluxes of CO_2 are overlapping across all wetland types but both forested and nonforested mineral soil wetlands tend to be larger sources (or lesser sinks) of CO_2 (see Figure 13.2). Similarly, CH_4 fluxes overlap across all wetland types, yet all wetland types tend to be sources of CH_4 (see Figure 13.2, this page).

13.3.1 Peatlands—Carbon Stocks and Fluxes

Peatlands include those ecosystems with organic soils generally classified as either fens or bogs, both of which are defined by water source and pH. Fens tend to be fed by groundwater and precipitation and have circumneutral pH values with vegetation generally dominated by sedges (*Carex* spp.) and brown mosses. In contrast, bogs are predominantly precipitation fed and have much lower pH and *Sphagnum* mosses. Other types of peatlands include riparian systems such as bottomland hardwood ecosystems

in the Mississippi River Valley, pocosins, Atlantic white cedar swamps, Carolina bays in the southeastern United States, and high-elevation peatlands in the Rocky Mountains from Canada to Mexico and throughout the Sierra Nevada of California. The total area of peatland in North America is about 1.3 million km^2 (see Table 13.1, p. 514).

Peatlands contain about 80% of the wetland carbon stock in North America and account for 48% of the net annual carbon uptake and 44% of the annual CH_4 flux. Approximately 58% of peatlands in North America are forested. The peatland carbon pool in Canada is currently estimated at 114 petagrams of carbon (Pg C), about 67% of which occurs in forests. This pool represents 88% of the total peatland carbon stock for North America (see Table 13.1, p. 514). Canadian peatlands have an estimated annual uptake of 41 teragrams of carbon (Tg C) and an estimated release of 16 Tg CH_4-C per year, 61% from nonforested peatlands. Alaska contains 42% of the U.S. peatland carbon stock and accounts for approximately 39% of the carbon uptake. Forests compose 49% of the peatland carbon stock in CONUS and 7% in Alaska. Methane from U.S. peatlands is 7% of the North American annual peatland flux; CONUS contributes 43% of the U.S. CH_4 flux. This difference in stocks and fluxes between the two countries having the majority of North American peatlands is attributable to the much larger peatland area in Canada. Mexico contains the largest area of tropical peatlands (~20,600 km^2), which constitutes approximately 57% of the total wetland area of the country (see Table 13.1, p. 514). Those peatlands contribute 2% of the North American peatland CH_4 flux as a result of the high flux rates in the tropics. Additionally, small areas of tropical peatlands occur in Puerto Rico (9 km^2). The estimated CH_4 emission is quite variable for each country or state, with the 95% confidence interval varying from 26% to 118% and 85% to 269% of the mean for temperate and tropical wetlands (see Table 13.1, p. 514), which is a reflection of the high degree of variability in the reported measurement data. The CH_4 fluxes applied for forested and nonforested peatlands (8.9 and 22.7 g C per m^2 per year, respectively) are less than the 26 g C per m^2 per year average for bogs and fens reported by Turetsky et al. (2014).

There is wide variation in intrinsic peat properties that influences the carbon stored in peat and how fast it accumulates after disturbances or with succession. Peat properties related to carbon storage are directly linked to the source material that changes with peatland type (Kracht and Gleixner 2000; Schellekens et al., 2012). For example, "peat moss," or *Sphagnum*-derived peat, is different in soil carbon density than peat derived from woody plants ("silvic peat"). Also, peat decomposition rates tend to increase with decreases in water tables (Ise et al., 2008). As such, care is needed in making broad assessments of peat accumulation in forested versus open peatlands, especially since dominant cover types can change (e.g., from silvic peat to *Sphagnum* peat) over time, and water tables can be influenced by short- and long-term precipitation patterns (e.g., droughts) and anthropogenic disturbances (e.g., draining). These factors all contribute to the large amount of variation in peatland carbon cycling and rates of peat accumulation. Peat carbon accumulation rates since the last glaciation range from 7 to 300 g C per m^2 per year (Kolka et al., 2011) in North America, with an average of 23 g C per m^2 per year during the Holocene (Loisel et al., 2014), but values commonly range from 20 to 30 g C per m^2 per year (Manies et al., 2016). In terms of peat accumulation, long-term rates range from 0.2 to 10 mm per year but typically range from 0.4 to 2.0 mm per year across all North American peatland types (Kolka et al., 2011). Peatland carbon pools are dependent on the depth of peat, ranging from 20,000 g C per m^2 in shallow peatlands to more than 300,000 g C per m^2 in peatlands >5 m deep (Kolka et al., 2011).

Generally, any factor that lowers the water table relative to the peat surface will result in increased CO_2 production, increased decomposition, and decreased CH_4 production (Waddington et al., 2015). There are also generalizations that can be made across peatland types, although variation in CO_2 and CH_4 production is high (e.g., McLaughlin and Webster

2014). Fen ecosystems are generally characterized by having relatively low CH_4:CO_2 fluxes compared with systems having very little water movement such as bogs, though fluxes vary greatly, both seasonally and latitudinally. In northern peatlands, CH_4 fluxes are generally highest when water tables are near the peat surface and seasonal temperatures are high (Turetsky et al., 2014). Pocosin ecosystem soils are in contact with groundwater except during seasonal droughts, thus their gaseous fluxes can be variable but generally produce less CH_4 than northern peatlands (Bridgham and Richardson 1992). The reduced gaseous fluxes of pocosins may be related to the high polyphenol content of their peats that resists decomposition even during moderate drought (Wang et al., 2015). The composition of the organic matter in peatlands also affects fluxes of CH_4 and CO_2, with low-quality peat maintaining low rates of decomposition, even when aerated (see Figure 13.3, this page). Those effects are evident both within and between climatic zones.

Gaps in research and monitoring activities to better understand how peatland carbon storage may change in an altered future climate are related mainly to disturbance events that dramatically alter the mechanisms of peat carbon accumulation and stability. Disturbance events of concern are those that alter wetland hydrology, which has a direct feedback to primary production and decomposition. While there is well-developed literature demonstrating that lower water tables coincident with changing precipitation patterns or altered drainage often result in a decline in the carbon sink strength of northern peatlands (Waddington et al., 2015), altered hydrology also has been shown to increase the vulnerability of northern latitude peatlands to wildfire (Benscoter et al., 2011; Turetsky et al., 2011a; Waddington et al., 2012), hence further increasing the vulnerability of peatland carbon pools to decomposition. Research has demonstrated that the extent of fires in boreal North America has steadily increased over the past five decades (Kasischke and Turetsky 2006), often with substantial peat combustion (Turetsky et al., 2011b). For example, a single fire event in northern peatlands can consume 3.3 to

Figure 13.3. Organic Soil Peat Core. Composed primarily from partially decomposed organic matter, this peat sample is from Drosera Fen in Yosemite National Park. [Figure source: Judith Drexler, U.S. Geological Survey.]

3.6 kg C per m^2 (Reddy et al., 2015; Turetsky et al., 2011b), recovery from which would require about 140 years. Disturbance-mediated changes in vegetation community composition also have implications for gas production because different plant species functionally alter rates of CO_2 and CH_4 fluxes from peat, or they affect the ability of peat to resist decomposition (Armstrong et al., 2015; Turetsky et al., 2014). Taken together, the effects of altered hydrology (whether induced by management or as a climatic response) on fire regime and productivity and changes in plant species composition represent key uncertainties in the current understanding of peatland carbon storage in an altered future climate.

13.3.2 Mineral Soil Wetlands—Carbon Stock and Fluxes

The total area of mineral soil wetlands in North America is about 0.9 million km^2 (see Table 13.1, p. 514). The United States contains 52% of the mineral soil wetland carbon stock in North America. Mineral soil wetlands in CONUS have an estimated carbon stock of 5.2 Pg C, with a net annual sequestration of 25.7 Tg C as CO_2 (Tg CO_2-C) and an estimated emission of 8.3 Tg CH_4-C per year (see Table 13.1). Alaska has a larger stock (11.3 Pg C), annual sequestration as CO_2 (13.2 Tg C), and CH_4

release (6.1 Tg CH_4-C). Canadian mineral soil wetlands have a carbon stock of 14.6 Pg C, with an annual CO_2 uptake of 23.5 Tg C and an estimated release of 9.9 Tg CH_4-C per year (see Table 13.1). Mexico has much smaller mineral soil wetland stock (0.5 Pg C), CO_2 sequestration, and CH_4 emissions. The estimates of the exchange of CO_2-C and CH_4-C are quite variable, with the 95% confidence interval ranging from 18% to 360% of the reported mean. Mineral soil wetland carbon stocks in North America are nearly equally divided between nonforested and forested wetlands, 48% and 52%, respectively. Methane releases from the wetlands are greatest for mineral soil wetlands in Canada, followed by CONUS and Alaska (see Table 13.1, p. 514); these estimates also are variable, having a 95% confidence interval ranging from 28% to 61% of the reported mean.

Different national agencies classify mineral soil wetlands differently, using various terms such as marshes, swamps, riverine wetlands, palustrine wetlands, prairie potholes, playas, and Carolina bays, as well as many other local and regional terms. Geography and geomorphology are distinguishing factors in some classifications and influence carbon dynamics. Although there is value in broad classifications, such as forested versus nonforested as in Table 13.1, it is important to recognize that boreal, temperate, and tropical regions in North America span from just over 14°N latitude along the Mexican border with Guatemala to boreal regions of Alaska and Canada positioned to 60° to 70°N latitude. Variation in the carbon pool within these mineral soil wetland types and regions correlates strongly with latitude. Modeled NPP of wetlands across all types, including organic soil wetlands, ranged from 461 to 618 g C per m^2 per year for tropical and lower-latitude temperate regions to as little as 172 to 183 g C per m^2 per year in boreal regions (Cao et al., 1996). Summarizing carbon dynamics in tropical wetlands, Sjogersten et al. (2014) reported an average NPP of 880 g C per m^2 per year for tropical mineral soil wetlands. The proportion of carbon being returned to the atmosphere as CH_4 also decreased with increasing latitude, with CH_4 fluxes varying slightly with respect to whether wetlands were forested or nonforested along this latitudinal gradient (see Table 13.1, p. 514). The data reported by Cao et al. (1996) do not differentiate organic soil wetlands from mineral soil wetlands, but reductions in NPP and CH_4 fluxes for mineral soil wetlands are included and would track with these overall patterns.

Mineral soil wetland carbon pools include those with soil organic layers that are less than 40 cm thick. The Intergovernmental Panel on Climate Change (IPCC) considers a soil depth down to 30 cm as the lower limit for reporting of mineral soil wetland carbon pools (IPCC 2013). To a depth of 30 cm, carbon pools range from 2,200 g C per m^2 in dry tropical mineral soil wetlands to greater than 10,000 g C per m^2 in boreal and moist temperate wetlands (Batjes 2011; Wickland et al., 2014). U.S. soil surveys consider soil properties in the upper 200 cm, but values in the top 150 cm are reported in this chapter to provide a uniform basis of comparison that includes both the surface soil layers and the subsoil.

Seasonal and diurnal fluxes of GHGs from boreal and temperate mineral soil wetlands have a wide range. For example, from temperate forested wetlands, CO_2 fluxes ranged from –0.444 to 3.303 g C per m^2 per day and CH_4 fluxes ranged from –0.014 to 0.0199 g C per m^2 per day (Alford et al., 1997; Harriss and Sebacher 1981; Harriss et al., 1982, 1988; Kelley et al., 1995; Krauss and Whitbeck 2012; Miller and Ghiors 1999; Mulholland 1981; Pulliam 1993; Wilson et al., 1989; Yu et al., 2008). The fluxes depend on the wetland type, soil temperature, and soil water regime. These factors are affected not only by latitude, but also by land-use change, leading to much assessment difficulty and uncertainty. North American wetlands release approximately 44 Tg CH_4-C per year, but the uncertainty surrounding this value is considerable (see Table 13.1, p. 514). For nonforested mineral soil wetlands of North America, NEE of carbon as CO_2, ranged from an average of –264 to 527 g C per m^2 per year. Methane was emitted from these same wetlands at rates of 0.8 to 127 g C per m^2 per year. Such broad ranges of CO_2 and CH_4 fluxes reflect

sensitivity to biotic and abiotic factors, which drive high uncertainty in estimating the net carbon balance and changes in carbon sinks at large scales and time periods.

Understanding the carbon balance across gradients of hydrology and vegetation within a mineral soil wetland is crucial to determining landscape-scale fluxes, especially for systems associated with fluvial networks. For instance, in a short-hydroperiod floodplain wetland in Virginia, GHG fluxes varied dramatically depending on the floodplain geomorphic unit (i.e., levee, backswamp, and toe slope) and in relation to longitudinal position (i.e., upstream versus downstream; Batson et al., 2015). The focus is often on the *in situ* capacity of forested mineral soil wetlands in controlling the carbon balance. However, many forested mineral soil wetlands are positioned for *allochthonous* inputs, (i.e., organic and inorganic carbon [including dissolved CO_2] that moves across terrestrial landscapes to aquatic environments). Such inputs, along with erosion, may influence the carbon balance significantly through external drivers (Ensign et al., 2013; Noe et al., 2016). Data on these inputs are few, as research has focused intently over the past several decades on carbon balance from organic soil wetlands (e.g., fens, bogs, and coastal marshes).

Prairie "potholes" represent one type of mineral soil wetland that has been studied intensively. The Prairie Pothole Region (PPR) is home to the largest inland mineral soil wetland ecosystem in North America. Covering about 777,000 km^2 of north-central United States and south-central Canada, the PPR is characterized by millions of closed depressional, mineral soil wetlands or potholes encompassing approximately 70,000 km^2 of undrained wetlands (Dahl 2014; Euliss et al., 2006). The distinguishing feature of prairie potholes is their lack of a discernable surface drainage network. These wetlands have the potential to represent a considerable contribution to the North American GHG balance, both as carbon storage and sequestration sites and as sources of GHGs (Badiou et al., 2011; Bansal et al., 2016; Tangen et al., 2015). PPR wetlands, also characterized by periods of inundation ranging from ephemeral to permanent, exist along a water-salinity gradient from fresh to hypersaline and occur primarily within a matrix of croplands and grasslands (Euliss et al., 2004; Goldhaber et al., 2014; Niemuth et al., 2010; Winter and Rosenberry 1998). Many PPR wetlands contain sulfate concentrations comparable to coastal systems, resulting in inhibition of CH_4 production (Goldhaber et al., 2014). Consequently, the biotic and abiotic factors that regulate the carbon dynamics and GHG balance of these systems are highly variable, both temporally and spatially.

Previous work recognizing PPR wetlands as significant carbon storage sites (Euliss et al., 2006) and identifying mineral soil wetlands as a major data gap (Bridgham et al., 2006, 2007) spurred considerable research in recent years pertaining to the overall GHG balance of these wetlands. Soil carbon stores are reduced by 12% to 26% when wetlands are converted from native grasslands to agricultural uses, presumably due to wetland drainage and soil disturbance (Gleason et al., 2008, 2009; Tangen et al., 2015). Peak CH_4 fluxes can exceed 0.75 g C per m^2 per day, and maximum cumulative seasonal CH_4 fluxes have been shown to be among the greatest reported for North American wetlands (Bansal et al., 2016; Bridgham et al., 2006; Tangen et al., 2015). In terms of the overall radiative balance of PPR mineral soil wetlands, CO_2 contributes the most (about 90%) to net GHG flux, followed by CH_4 (about 9%) and N_2O (about 1%; Gleason et al., 2009).

13.3.3 Lateral Carbon Fluxes from Terrestrial Wetlands

The lateral flux of carbon may occur in the form of DIC, DOC, dissolved CH_4, and particulates. The DOC flux is generally the largest of these fluxes from wetlands and is particularly important because it can be a source of carbon to both surface and groundwater. The rates of DOC production and loss are variable across time, space, and wetland types and appear to be climate dependent (Drösler et al., 2014). The transport of DOC to surface waters is fairly well studied for peatlands (Hope et al., 1994).

The IPCC Wetlands Supplement (2013) chapter on drained inland organic soils reviewed the literature and estimated DOC flux from natural systems across biomes. As part of that supplement, Drösler et al. (2014) found 1) boreal peatland flux to surface waters to be 8.4 g C per m^2 per year (95% confidence interval ranging from 6.0 to 11.1 g C per m^2 per year), 2) temperate peatland flux to surface waters to be 21.2 g C per m^2 per year (17.3 to 26.2 g C per m^2 per year), and 3) tropical DOC fluxes to surface waters to be 56.9 g C per m^2 per year (49.2 to 63.8 g C per m^2 per year). Higher temperatures lead both to more production and decomposition and to higher DOC fluxes.

However, mineral soil wetlands are not well studied, possibly because many mineral soil wetlands have no surface stream drainage outlet. Studies conducted in the temperate northeastern United States summarized data for 30 forested watersheds with no wetlands present and found DOC fluxes to range from 0.5 to 4.9 g C per m^2 per year (mean = 2.4 g C per m^2 per year; Raymond and Saiers 2010), considerably lower than the aforementioned mean of 21.2 g C per m^2 per year found for peatlands. At least for the temperate zone, these fluxes can be considered as the lower bound of mineral soil wetland fluxes. Aitkenhead and McDowell (2000) reviewed the literature and compared riverine DOC fluxes across a wide range of climate and vegetation biomes but did not differentiate DOC contributions between peatland and mineral soil wetlands. Here, the studies in known mountainous and peatland watersheds were removed, with the caveat that they are stream and river fluxes, not wetland fluxes. This chapter estimated the mean DOC flux for streams and rivers that have considerable mineral soil wetlands in their watersheds. The mean DOC flux for mineral soil wetlands in 1) tropical systems is estimated as 9.9 g C per m^2 per year (n = 2; Day et al., 1977; Malcolm and Durum 1976); 2) in temperate systems, as 5.4 g C per m^2 per year (n = 6; Clair et al., 1994); and 3) in boreal systems, as 2.1 g C per m^2 per year (n = 16; Clair and Ehrman 1996; Mulholland and Watts 1982).

Interestingly, this chapter's estimates of mineral soil wetland DOC fluxes as a percentage of organic soil DOC fluxes are relatively consistent across the three biomes (25%, 25%, and 17%, respectively, for boreal, temperate, and tropical ecosystems). DOC fluxes from North American terrestrial wetlands can be estimated using the wetland areas in Table 13.1, p. 514, and characterizing Alaska and Canada as boreal, CONUS as temperate, and Puerto Rico and Mexico as tropical. Boreal DOC fluxes are 11.4 Tg (10.1 Tg from organic wetland soils and 1.3 Tg from mineral wetland soils). Temperate DOC fluxes are 3.5 Tg (1.8 Tg from organic wetland soils and 1.7 Tg from mineral wetland soils). Tropical DOC fluxes are 1.4 Tg (1.2 Tg from organic wetland soils and 0.2 Tg from mineral wetland soils). Together, these fluxes total 16.3 Tg DOC for North America. Although there is low confidence in the amount of lateral DOC fluxes, especially those related to mineral soil wetlands, these fluxes are lower but of similar magnitude as the NEE and about 37% of the CH_4 fluxes from terrestrial wetlands (see Table 13.1).

13.3.4 Carbon Stock and Balance

The estimated North American terrestrial wetland carbon pool of 161 Pg C is less than the 214 Pg C reported in SOCCR1 for permafrost peatlands, nonpermafrost peatlands, and freshwater mineral soil wetlands (CCSP 2007). This difference is attributable to the inclusion of permafrost wetlands in the SOCCR1 report (CCSP 2007) and differences in nonpermafrost wetland area. The estimate here (129 Pg) for the amount of carbon stored in North American peatlands is less than that (163 Pg) reported by Gorham et al. (2012), again, likely a result of the Arctic permafrost area.

The development of a carbon balance sheet for the terrestrial wetlands of North America provides a useful perspective for considering the relative contributions of the various pathways, the relative differences in fluxes, and uncertainties. The wetland carbon balance sheet can be simplified by considering NEE as the net change in the CO_2-carbon exchange between the wetland and the atmosphere (negative values indicate net transfer to

the ecosystem). Net gains to the wetland, assuming a negative NEE, are effectively allocated among vegetation and soils. The principal losses of carbon from the wetlands that are not included in NEE are CH_4 fluxes (see Sections 13.3.1, p. 516, and 13.3.2, p. 518), DOC (see Section 13.3.3, p. 520), hydrological fluxes of DIC and suspended particulates, and losses due to episodic disturbance regimes (e.g., fire). Unfortunately, there is very little information about the loss of carbon as DIC or particulates for terrestrial wetlands. Thus, for current purposes, they are not considered further. Accordingly, the net ecosystem carbon balance for terrestrial wetlands in North America is –65.3 Tg C (–126.4 Tg C input, see Table 13.1, + 44.8 Tg CH_4-C flux, see Table 13.1, + 16.3 Tg DOC loss, see Section 13.3.3), indicating that the wetlands are a net carbon sink. However, the estimated annual accumulation in carbon among the soil and vegetation pools, 47.9 and 43.6 Tg C per year, respectively, yields an imbalance of +30 Tg C, indicating that the estimated NEE is too low or that one or more of the components are overestimated.

There is considerable variability in estimates of wetland carbon fluxes, whether it is from field measurements or large-scale simulations. Accordingly, comparison among reports provides useful perspectives. The North American terrestrial wetland CH_4 flux, based on measurements and extrapolated to the wetland area, is estimated at 45 Tg C per year, which is considerably higher than the estimated amount in SOCCR1 (6.1 Tg C per year). SOCCR1 also used measurements as the basis (CCSP 2007); however, the SOCCR2 estimate is nearer the range of several recent modeling studies. Using an ensemble of models to simulate CH_4 emissions in North America, Poulter et al. (2017) reported annual emissions of 31.8 to 33.5 Tg C for 2007 to 2012. Similarly, using six different datasets, Zhang et al. (2017a) reported an average CH_4 emission rate of 22.6 Tg C per year for the region from 2000 to 2006. This amount is similar to the average annual emission estimated for 1979 to 2008 of 17.8 Tg C per year by Tian et al. (2010). The annual global CH_4 flux from wetlands is estimated between 124 and 139 Tg C per year (Saunois et al., 2016; Bloom et al., 2017; Poulter et al., 2017; Zhang et al., 2017a, b); accordingly, the contribution of North America to the global CH_4 budget is likely within the range of 20% to 30%. While there are not any large-scale NEE assessments, synthesizing measurement data for terrestrial wetlands, Lu et al. (2017) report an average annual accumulation rate of 93 g C per m^2, which is considerably higher than the average rate of 53 g C per m^2 reported here.

Assessing the pools associated with the carbon balance sheet provides additional perspective. Both organic and mineral soils accumulate carbon. Estimates here of carbon accumulation in the soil are 25 and 17 g C per m^2 per year for peat and mineral soils, respectively; those aggregated rates are based on the mean accumulation rates, reported by Bridgham et al. (2006), weighed by the wetland area. Accordingly, peat and mineral soils gain approximately 32.2 and 15.9 Tg C per year, respectively. Although there is a wide range in vegetation productivity, an estimated 43.6 Tg C is sequestered in biomass annually. The estimate assumes that accumulation in plant biomass is balanced with decomposition in nonforested wetlands and that forested wetlands have a net accumulation of 30 to 50 g C per m^2 per year (Bridgham et al., 2006; Stinson et al., 2011). The resulting summation of carbon sequestration by the soil and vegetation components (92 Tg C) is greater than the allocation to CH_4 fluxes or DOC.

13.4 Wetland Management, Restoration, and Creation

Generally, terrestrial wetlands are managed for one or more of the ecosystem services they provide. In many cases, wetlands are managed as set-aside areas used as natural filters for water quality, areas for rare species, and land for hunting and trapping due to their faunal diversity. For example, several international conservation organizations consider the PPR of the midwestern United States and Canada as the most important waterfowl habitat in North America. Management decisions and development that change the hydrology, soils, or vegetation will

affect carbon dynamics, often leading to enhanced decomposition, decreased CH_4 flux, and reduced carbon sequestration, particularly when wetlands are drained. In contrast, restoration of drained wetlands (or avoided loss of wetlands through easements) increases carbon sequestration and CH_4 production. Policies using wetlands as carbon banks and using the carbon gained through wetland restoration to trade in carbon markets are becoming increasingly common globally.

13.4.1 Effects of Wetland Management, Restoration, and Creation on Carbon

This section considers wetland management that does not convert wetlands to another land use. Wetland management occurs on a gradient from very intensive management to preservation. As they have been for thousands of years, wetlands managed for preservation or their intrinsic ecosystem services generally are carbon sinks, although there are some indications that rising temperatures from climate change may be changing wetlands from sinks to sources. For example, an undisturbed bog in Canada was a carbon source for 3 years of a 6-year study (Roulet et al., 2007). Even if wetland sinks are smaller than they once were, management or restoration practices could have dramatic feedbacks to atmospheric concentrations of CO_2 and CH_4. In a management example, there are approximately 658 km^2 of terrestrial wetlands under "moist-soil" management in the U.S. National Wildlife Refuge System, where lands are flooded for wintering and migrating waterfowl. Research has demonstrated that seasonal drainage in moist soil regimes leads to major losses of soil carbon (Drexler et al., 2013). The practice of deeply flooding marshes is not as common in the national wildlife refuges as seasonal drainage, but deep flooding may be an option for increasing carbon sequestration rates (Bryant and Chabreck 1998).

The effect of altered hydrology does not necessarily cause a loss of ecosystem carbon from managed wetlands. Studies of carbon pool response to managed peatlands in Finland have shown that increased forest productivity may offset losses due to water management resulting in a net increase of carbon, but this response is site dependent (Minkkinen et al., 2008). Similarly, forest harvesting only had a transient effect on the soil carbon pool of a mineral soil wetland (Trettin et al., 2011). In contrast, peat utilization, as in peat mining for fuel or horticultural purposes, is the extreme where the peat itself is removed from the wetland. Although peat mining is not common in North America, Canada is the third largest producer of horticultural peat in the world, with much of the peat originating from the peatlands in the St. Lawrence Lowlands on the Canadian side of the Great Lakes (Van Seters and Price 2001). For production agriculture where wetlands remain wetlands, water levels are typically controlled to maximize production, usually at the expense of carbon pools. Prairie potholes and other hydrologically isolated wetlands are often nested within agricultural lands but remain undrained. These cropped, undrained wetlands can be major sources of GHGs due to increased nutrient loading and associated nitrous oxide (N_2O) fluxes. In addition, temporarily ponded wetlands that dry down during the growing season can be tilled and farmed, increasing decomposition rates. Approximately 6,500 km^2 of U.S. peatlands are being used for crop production (ICF International 2013). The converted peatlands are usually highly productive for agriculture, but they also have high potential as GHG mitigation sites if the land is restored to vegetated wetlands (Richardson et al., 2014; Wang et al., 2015). Specific GHG mitigation benefits accrue from 1) decreases in CO_2 fluxes related to the oxidation of soil carbon while in crop production, 2) decreases in the use of nitrogen fertilizers, 3) decreases in lime application amendments, and 4) increases in carbon sequestered in soils and perennial vegetation (ICF International 2013). Crops such as sugarcane lead to large losses of carbon through enhanced decomposition (Baker et al., 2007). Paddy rice production systems are well-known sources of CH_4 (Lindau et al., 1993) and N_2O. Other crops such as sugar beet, radish, cranberry, blueberry, lettuce, celery, carrot, potato, onion, and mint are grown in wetlands, but little data exist on their influence on ecosystem carbon balance.

Similarly, aquaculture has altered wetlands in North America, but, again, little data exist on the impact on carbon storage or fluxes. Although forest harvesting causes short-term changes in carbon sequestration during the period of stand regeneration, it generally has little impact on long-term wetland soil carbon balance (Roulet 2000; Trettin et al., 2011).

Wetland restoration usually includes the re-establishment of hydrological regimes to support hydrophytic vegetation. Wetland restoration and creation of new wetlands (where none existed previously) and small ponds have counteracted much of the wetland losses in CONUS (Dahl 2011). For instance, from 1998 to 2004 and 2004 to 2009, areas reclassified as wetlands in the United States increased by 17%, meaning that 802 km^2 of new wetlands were created, but this figure does not indicate how many additional square kilometers of the restored wetlands were still classified as wetlands. In addition, creation of small ponds has increased over the last few decades, with 838 km^2 per year created from 2004 to 2009 (Dahl 2011).

Wetland restoration can lead to the opposite effects of drainage, with increases in carbon pools and in CH_4 fluxes and lower CO_2 fluxes (Wickland et al., 2014). Research has found that restoring wetlands by rewetting them increases soil carbon storage (Lucchese et al., 2010). IPCC guidelines for mineral soil wetlands state that cultivation leads to losses of up to 71% of the soil organic carbon in the top 30 cm of soil over 20 years and that restoration increases depleted soil carbon pools by 80% over 20 years, and by 100% after 40 years (Wickland et al., 2014). Rewetting also may increase CH_4 fluxes, not only above the previously drained levels, but also above reference levels temporally (Badiou et al., 2011). However, some studies have found that restoration did not increase CH_4 fluxes (Richards and Craft 2015). In the long term, restoring degraded wetlands appears to be a positive for GHG mitigation.

Creating new wetlands and small ponds also can affect both long-term soil carbon storage and gaseous fluxes. Created wetlands tend to have carbon accumulation rates higher than those of natural wetlands (Bridgham et al., 2006). In addition, created wetlands often have similar or lower CH_4 fluxes (Mitsch and Hernandez 2013; Winton and Richardson 2015). However, assessments have found that small ponds are large sources of CH_4 (Holgerson and Raymond 2016). Similar to created wetlands and some riparian zones, small ponds may sequester carbon at high rates due to high sediment deposition rates from the surrounding land.

Many restored wetlands do not provide the level of ecosystem services they did before their degradation, usually a result of inadequate hydrology restoration. One survey found that only 21% of wetland restoration sites have ecologically equivalent natural functions (Turner et al., 2001). Post-restoration monitoring is critical to determining restoration success and providing opportunities to modify restoration techniques if necessary. Assessment of success usually occurs over relatively short periods (1 to 3 years) and with relatively simple protocols because of time, resource, and technical constraints. Determining success over the short term is difficult because wetland processes, such as soil formation or forest recovery, occur over decades. Also, most current assessment techniques are fairly simple and may not adequately characterize the condition of a wetland, especially if critical functions such as hydrology or processes such as carbon and nutrient cycling are not fully understood. Moreover, inadequate study of many wetland types challenges efforts to understand both the processes that lead to carbon accumulation and fluxes and the impact of wetland restoration on carbon. Furthermore, due to the developmental trajectory of restored wetlands, their capacity to store carbon may change through time, with considerable storage initially and then much less storage thereafter once vegetation has fully colonized and root systems have developed (Anderson et al., 2016).

13.4.2 Processes and Policies that Affect Wetland Management, Restoration, and Creation

Recognition of the values that wetlands provide has led to changes in federal policies aimed at protecting, restoring, and creating wetlands over the past

four decades. Four significant policies are 1) Section 404 of the Clean Water Act (1972); 2) the Highly Erodible Land Conservation and Wetland Conservation Compliance provisions of the 1985 Food Security Act and subsequent amendments, commonly known as the "Swampbuster program"; 3) President George H. W. Bush's "no net-loss" policy (1989); and 4) the U.S. Army Corps of Engineers and EPA compensatory mitigation rule (USACE 2008). Initially passed as part of the Federal Water Pollution Control Act of 1972, the Clean Water Act focused on nonagricultural wetland conversions (U.S. EPA 2015). In its initial form, the Swampbuster program discouraged farmers from converting wetlands by withholding federal farm program benefits if conversion occurred on nonexempt wetlands. Farm Bill 1990 amendments created the Wetland Reserve Program, which was later consolidated with other easement programs into the Agricultural Conservation Easement Program (ACEP). Rather than withholding incentives, the USDA NRCS incentivizes farmers to restore, protect, and enhance wetlands by purchasing wetland reserve easements via ACEP (USDA 2014). The Agricultural Act of 2014 (i.e., Public Law 113-79, commonly referred to as the 2014 Farm Bill) provided NRCS with the authority to enroll wetlands in 1) permanent easements, with 100% of the easement value and 75% to 100% of restoration costs covered, 2) 30-year easements funded at 50% to 75% of the easement value with 50% to 75% of the restoration costs covered, and 3) term easements with stipulations dependent on state laws.

The no net-loss policy, which sought to replace lost wetland habitat with new habitat by restoring and creating wetlands, is now the cornerstone of U.S. wetland conservation (Mitsch and Gosselink 2015). As a result, numerous federal and state agencies, non-governmental organizations, and private landowners are engaged in wetland restoration and creation across the United States with a keen focus on establishing the proper hydrological conditions needed to support flora and fauna specific to a certain wetland type. Such activities often result in preserving or expanding the carbon pool of wetlands, but little attention has been given to ensuring the long-term sustainability of such newly formed carbon sinks. Wetland restoration is still a relatively new field, and management approaches for maintaining the sustainability of carbon sinks are still being developed, tested, and refined.

The Federal Policy on Wetland Conservation in Canada (Canadian Wildlife Service 1991) also encourages no net-loss of wetlands. The regulation is focused largely on activities undertaken by the Canadian government on its federal land. Although the policy discourages wetland destruction or degradation, the Canadian government does not require compensatory mitigation. Though currently limited, the Natural Protected Areas Commission of Mexico has a national wetland policy to protect wetlands and avert losses.

13.5 Terrestrial Wetland Trends and Feedbacks

An important concern globally is how wetlands will respond to a changing climate. Climate change has the potential to affect carbon cycling of natural, degraded, created, and restored wetlands. However, there is considerable uncertainty regarding the likely responses, including how warming and variations in precipitation regimes will influence the balance between plant productivity and organic matter decomposition. An example pattern might be warming followed by drier conditions leading to wetland carbon losses, as has occurred in simulated peatland droughts (Fenner and Freeman 2011). Altered precipitation regimes also may shift the hydrological balance in the absence of warming. Even on an annual timescale, individual wetlands can alternate between a carbon sink in wet years to a carbon source in dry years, illustrating the sensitivity of wetlands to biotic and abiotic conditions. However, the direct correspondence of increased peat oxidation with a lowered water table is not universal. Instead, Makiranta et al. (2008) showed soil temperature controlled more of the variability in peatland soil respiration than did the water-table position. Similarly, CH_4 fluxes in high-latitude wetland ecosystems with high

water tables were more sensitive to soil temperature than were those ecosystems with lower water tables, which were more sensitive to water-table position (Olefeldt et al., 2013). Accordingly, changes in carbon pools and fluxes in response to changes in temperature and precipitation regimes will vary greatly based on wetland type and interactions with hydrology because carbon cycling may be different under warmer and wetter conditions than under warmer and drier conditions. For example, CH_4 fluxes from PPR wetlands were four times higher under warmer and wetter conditions than the fluxes were under warmer and drier conditions (Bansal et al., 2016). Northern seasonally frozen peatlands already are undergoing rapid changes, and increased carbon fluxes are likely to continue over the coming decades to centuries as conditions continue to warm (Schuur et al., 2015). Another general pattern is that drier conditions will facilitate and exacerbate fires, especially in peatlands, resulting in large fluxes from the oxidized peat (Turetsky et al., 2011b; see also Ch. 11: Arctic and Boreal Carbon, p. 428).

The response of mineral soil wetlands to changes in temperature and precipitation regimes is uncertain, largely because of the wide range in properties and geomorphic setting. Histic-mineral soil wetlands ("histic" refers to soils with a 20- to 40-cm organic horizon) may be expected to respond similarly to peatlands. For other types, such as mineral soil wetlands in floodplains where the surface organic layer is thin due to high turnover rate, the changes in that layer associated with climate change are likely small. Changes in the hydrological regime also are expected to alter the carbon balance. Increased periods of a high water table or flooding may be expected to reduce productivity (Trettin et al., 2006) and increase CH_4 fluxes (Sharitz and Pennings 2006). The effect of climate change on organic matter decomposition and carbon export from the wetland is an important uncertainty and feedback to adjoining aquatic ecosystems. The uncertainty in mineral soil wetland response is high, largely because there are far fewer studies on mineral soil wetlands than on peatlands.

Rising atmospheric CO_2 is considered likely to increase GHG fluxes from wetlands due to increased CH_4 fluxes offsetting gains from increased plant carbon sequestration (Bridgham et al., 2007; Hyvonen et al., 2007). Hyvonen et al. (2007) suggest that soil carbon in the temperate and boreal zones will increase because of increased litter input, but the magnitude of the response will depend on available nitrogen and land management. Little is known about interactions between changes in water regime and plant productivity. In upper Michigan, lowered water tables led to increased productivity in vascular plants (e.g., shrubs and sedges) and *Polytrichum*; higher water tables led to higher *Sphagnum* production (Potvin et al., 2015). Demonstrating the importance of field experimentation, Dijkstra et al. (2012) measured increases in CH_4 in both mineral soil wetlands and peatlands following manipulation of the water regime. Understanding these interactions with CH_4 fluxes is fundamental to considering the feedback associated with rising atmospheric CO_2 (Petrescu et al., 2015; Zhang et al., 2017b).

13.6 Global, North American, and Regional Context

13.6.1 Global and Continental Perspectives

Observational studies suggest that wetlands cover an estimated 8.2 million km² globally (Lehner and Döll 2004). However, based on recent studies that use both observations and models, the mean global area may be 12.3 million km² (Melton et al., 2013). The largest concentrations of wetlands generally are found between 50° and 70°N latitude, with substantial concentrations also found between 0° to 10°S latitude (Lehner and Döll 2004). North of 70°N latitude, continuous permafrost ecosystems also contain considerable soil carbon (see Ch. 11: Arctic and Boreal Carbon, p. 428). Wetlands are estimated to cover approximately 2.2 million km² in North America (see Table 13.1, p. 514), or about 9% of the continental land area. Although approximate global and regional extents of wetlands are generally known, there are significant challenges that hinder estimating wetland coverage with a high degree of confidence. These challenges include, but are not limited to, lack of detailed inventories, nonuniform definitions of wetlands, limitations of remotely sensed data and models, and continuing drainage and conversion of wetlands worldwide.

Positioning the North American wetland carbon stock in a global context is difficult due to the broad range (300 to 530 Pg C) reported (Mitra et al., 2005). Accordingly, the North American wetlands (161 Pg C) compose a significant but uncertain proportion (30% to 54%) of the global wetland carbon stock.

Natural wetlands are the largest natural source of CH_4 fluxes to the atmosphere (Kirschke et al., 2013) and thus are an important consideration of large-scale modeling assessments. Saunois et al. (2016) conducted a comprehensive assessment of the global atmospheric CH_4 budget using "top-down" and "bottom-up" approaches, which respectively are based on inversions of atmospheric CH_4 data and process-based wetland biogeochemical models. Twenty top-down and 11 bottom-up estimates were provided for North American wetland fluxes averaged from 2003 to 2012. The multimodel mean (±1 standard deviation) was 16 ± 4 Tg CH_4-C emitted per year for the top-down estimates, and 35 ± 11 Tg CH_4-C per year for the bottom-up estimates. Boreal North America (i.e., Alaska and Canada) account for most of the difference between these two estimates, with the bottom-up approaches exceeding the top-down approaches by 19 Tg CH_4-C per year. Estimating the CH_4 flux from North American wetlands between 1979 and 2008, Tian et al. (2010) estimated an average of 17.8 Tg CH_4-C per year. Those simulation approaches are less than the estimate of North American wetland fluxes reported in this chapter, 44.8 Tg CH_4-C per year (see Table 13.1, p. 514). Both approaches have relatively large uncertainty levels associated with the CH_4 flux. Extrapolation of measurement data across the wetland area presumes a uniform response that belies the considerable differences among wetlands across the landscape. The large-scale model assessments suffer from the same issue of not having the capacity to consider variation among wetlands, but they have the ability to accommodate some aspects of spatial variability. The relative correspondence of the wetland CH_4 flux attests to the merits of both the large-scale process-based models and the need for additional empirical studies, particularly on mineral soil wetlands, to provide a broad base for model validation.

13.6.2 Regional Perspectives—United States, Canada, and Mexico

Within North America, Canada has the greatest wetland coverage, with estimates ranging from 1.27 to 1.60 million km^2, followed by Alaska with an estimated 0.18 to 0.71 million km^2 of wetlands (Lehner and Döll 2004; Zhu and McGuire 2016). Estimates of terrestrial wetlands for CONUS from the USFWS National Wetlands Inventory (0.39 million km^2) and Mexico (~0.05 million km^2) are smaller than the total wetland area suggested by Lehner and Döll (2004), 0.45 and 0.16 million km^2, respectively. The reported soil carbon stock for CONUS terrestrial wetlands (12.6 Pg C) approximates the estimate (10.6 Pg C) provided through the U.S. EPA's National Wetland Condition Assessment (NWCA; Nahlik and Fennessy 2016). The relatively small difference in soil carbon stock is attributable to less wetland area as reported in the NWCA (a difference of about 11,000 km^2) and a shallower reporting depth (120 cm). Wetlands in Canada are dominated by peatlands, which harbor large carbon stocks estimated at 115 Pg C for this assessment (see Table 13.1, p. 514) and 150 Pg C by Tarnocai et al. (2005). The greatest concentration of wetlands is in the provinces of Manitoba and Ontario, which contain about 41% of Canada's wetlands (Mitsch and Hernandez 2013).

The recent cartographic assessment of Mexico's wetlands provides important new information about the distribution of wetlands and context for assessing their loss (Landgrave and Moreno-Casasola 2012). Inland marshes are found in deltaic regions of the southeastern states of Veracruz, Tabasco, and Campeche, where the floodplains have deep organic soils (Smardon 2006). Marshes also are found in mountain ranges of central Mexico and in localized

Table 13.2. Estimates of Wetland Area, Total Carbon Storage, Carbon Dioxide and Methane Fluxes, and Net Carbon Flux by Major U.S. Region[a–b]

Region	Wetland Area (km^2)	Total Carbon Storage[c] (Pg C)	CO_2 Exchange[d] (Pg CO_2 per Year)	CH_4 Exchange[e] (Pg CO_2e per Year)	Net Carbon Flux[f] (Pg C per Year)
Eastern United States[g]	271,482	3.8, 4.2	–0.18, –0.048	0.186, 0.187	–0.049, –0.013
Great Plains[h]	30,380	0.22	NR[i]	0.082	–0.02
Western United States[j]	10,114	0.06, 0.07	–0.005, 0.0002	0.002	–0.0015, 0
Boreal Alaska – North[k]	112,007	2.4	NR	0.020	–0.002
Boreal Alaska – South[k]	18,627	0.9	NR	0.006	0.001

Notes

a) From U.S. Geological Survey's LandCarbon Program. Cells with two numbers represent the reported minimum and maximum. Carbon amounts are in petagrams (Pg).

b) See references for uncertainty analyses for the respective regions.

c) Total carbon storage for the eastern United States, Great Plains, and western United States is for 2005 and is the sum of biomass (live and dead) and the upper 20 cm of soil; for Alaska, total carbon storage is the average stock from 2000 to 2009 and is the sum of biomass (live above ground, live below ground, and dead), moss, litter, surface organic soil layers, and the upper 1 m of mineral soil.

d) Carbon dioxide (CO_2) flux for the eastern United States, Great Plains, and western United States is for 2001 to 2005; for Alaska, it is for 2000 to 2009.

e) Methane (CH_4) flux for the eastern United States, Great Plains, and western United States is for 2001 to 2005 and is presented in CO_2 equivalent (CO_2e) using a global warming potential (GWP) of 21; for Alaska, the flux is for 2000 to 2009 and is presented in CO_2e using a GWP of 25. Note that CO_2e is the amount of CO_2 that would produce the same effect on the radiative balance of Earth's climate system as another greenhouse gas, such as CH_4 or nitrous oxide, on a 100-year timescale. For comparison to units of carbon, each kg CO_2e is equivalent to 0.273 kg C (0.273 = 1/3.67). See Box P.2, Global Carbon Cycle, Global Warming Potential, and Carbon Dioxide Equivalent, p. 12, in the Preface for more details.

f) Net carbon fluxes for the eastern United States, Great Plains, and western United States are for 2001to 2005; for Alaska, they are for 2000 to 2009.

g) Zhu and Reed (2014).

h) Zhu and McGuire (2011).

i) Not reported.

j) Zhu and Reed (2012).

k) Zhu and McGuire (2016).

areas in the Sonoran and Chihuhuan deserts where springs feed shallow swamps (Mitsch and Hernandez 2013). However, little is known about their carbon stock or CO_2 and CH_4 fluxes.

The U.S. Geological Survey's LandCarbon Program developed ecoregion estimates of current and future projections of carbon storage, net CO_2 exchange and CH_4 fluxes, and net carbon balance of U.S. wetlands (Zhu and McGuire 2010), providing context for the current assessment. Wetland area, carbon stocks, and fluxes were estimated using process-based models and land-use and land-cover maps. These estimates, originally reported by level II ecoregion in a series of reports, are summarized by region in Table 13.2, this page. The LandCarbon assessment provides a basis for regional comparisons using a common methodology. However, the reported pools and fluxes are substantially different than those included in Table 13.1, p. 514, which uses the National Wetlands Inventory as the basis for wetland area, summarizes geospatial databases for the pools, and synthesizes observational studies as the basis for the pools and fluxes.

13.7 Synthesis, Knowledge Gaps, and Outlook

13.7.1 Summary of Terrestrial Wetlands Carbon Cycling

North American wetlands constitute a significant proportion (37%) of the global wetland area. The uncertainty in wetland area for North America is relatively low because wetlands in CONUS and Alaska, Mexico, and Canada have relatively recent inventories and assessments. However, more information about soil carbon and vegetation biomass within the wetlands is needed to assess carbon pools and fluxes and reduce uncertainties in the estimates. Wetland soil type varies significantly with latitude, with Alaska and Canada having the majority of the peatland area. Mineral soil wetlands are predominant (79%) in CONUS and contain 38% of its wetland carbon stock. An important consideration regarding the estimate of carbon pools in peatlands, which consist of 58% of the North American wetland area, is that total depth of peat is seldomly reported, while the average depth commonly exceeds the typical assessment depths of 1 to 2 m. Peatlands contain approximately 80% of the North American carbon, a proportion that is likely to increase substantially if the entire peat depth were considered. Nonforested vegetation communities compose 44% of the wetland area in North America, contain approximately 43% of the carbon pool, and accumulate 47% of the net carbon gain.

Historically, the wetland loss in North America has been significant, particularly in CONUS. However, to assess contemporary losses, periodic inventories at the national scale are needed. Currently, only the United States has regular updates to its wetlands inventory. Restoration and creation of new wetlands are major offsets to loss of natural U.S. wetlands. Whether these new wetlands have the same carbon dynamics as natural wetlands is a major uncertainty that will become more important as restored wetlands become a larger proportion of the total wetland area. A global meta-analysis comparing 621 restored and created wetlands to 556 reference wetlands indicated that functions related to biogeochemical cycling (mainly to carbon storage) were 23% lower in the restored and created wetlands (Moreno-Mateos et al., 2012). Successful functioning of those wetlands will be critical to mitigate the long-term losses of carbon from degraded wetlands.

13.7.2 Knowledge Gaps and Associated Uncertainties in the Wetland Carbon Cycle

The following are some major gaps in current knowledge about the North American wetland carbon cycle.

1. Future wetland response to climate change is uncertain. Because temperatures are predicted to increase at greater rates at higher latitudes, northern temperate wetlands, especially peatlands, are expected to be the most affected. More uncertainty exists in the predictions of precipitation, changes in which could either mitigate or exacerbate carbon sequestration rates in terrestrial wetlands. Although contemporary measurements and modeling offer perspective, additional manipulative experiments—such as the U.S. Department of Energy's Spruce and Peatland Responses Under Changing Environments (SPRUCE) experiment in northern Minnesota (Hanson et al., 2017) and USDA's former PEATcosm experiment in the Upper Peninsula of Michigan (Potvin et al., 2015)—are critical to test how wetlands will respond to changes in temperature and hydrological regime in the field. Work in mineral soil wetlands is particularly needed because of the paucity of studies and the functional linkages with aquatic systems.
2. Greater understanding is needed of the factors controlling carbon cycling in wetlands. Additional measurements of GHG fluxes and processes regulating the fluxes and carbon storage using improved inventories and methods at multiple spatial scales are required to 1) understand the interactions of soil, vegetation, and climatic factors; 2) provide a basis for quantifying fluxes to reduce significant uncertainties; and 3) evaluate biogeochemical and inverse-atmospheric models.

Particularly needed are studies that assess convergence across diverse spatial and temporal scales or lead to a process-based understanding of why convergence does not occur.

3. Dissolved carbon export, including both DIC and DOC, is a major uncertainty in the wetland carbon cycle. Dissolved carbon affects water quality and is an important food source for aquatic systems and estuaries, and dissolved gases may contribute to atmospheric loading. Understanding the mechanisms controlling dissolved carbon production and transformation is a major gap requiring field and watershed-scale assessments.
4. A better understanding is needed of the relationship between the sustainability of stored carbon and the particular chemistry of the carbon compounds that make up the carbon sink. Preliminary research shows that polyphenol content may serve to preserve peats under moderate drought conditions (Wang et al., 2015), but little is known about either the exact types of polyphenols or the plant communities that have the highest sustainability under projected climate and environmental conditions.
5. Data on restored and managed wetlands are sparse and insufficient to support assessment and modeling needs. Measurements to document the carbon balance in these wetlands are needed. Also necessary are standardized measurements and methods for collecting basic data in the field at the same depth and for analyzing parameters such as bulk density and percent of organic carbon. Monitoring of wetland restoration needs to extend through the entire trajectory of the project to gain a functional understanding of the differences in gaseous fluxes and carbon accumulation between natural and restored wetlands.

13.7.3 Tools for Assessing the Wetland Carbon Cycle

Due to the extremely wide variation in wetlands across North America, as well as the certainty that there will never be enough measurements to adequately quantify the wetland carbon stocks and fluxes, models present the means to represent the biophysical processes inherent to wetlands at variable spatial scales. Those tools provide needed capabilities to inform conservation, management, and mitigation strategies to sustain ecosystem services inherently linked to the wetland and global carbon cycle. Models also are useful for addressing the uncertainties within the carbon cycle and, in turn, for focusing field monitoring and experiments to fill critical information gaps. Mechanistic models provide the capabilities for simulating the processes that regulate carbon dynamics in wetlands reflecting the myriad soil, vegetation, and climatic conditions and management influences. Because of the water table's regulatory function in the wetland carbon cycle, an accurate representation of wetland hydrology is critical to model performance. There are fewer models for wetlands compared to those for uplands. Among biogeochemical models that are widely applicable to terrestrial wetlands and have the broadest capabilities with respect to soil and vegetation types are the Forest DNDC (or DeNitrification DeComposition) model, which was identified by USDA in the development of its carbon accounting framework (Ogle et al., 2014), and the DayCent model (Parton et al., 1998), which is widely used in grassland and agroecosystem simulations. Scaling wetland hydrology within a biogeochemical model is difficult; hence, coupling a biogeochemical model with a hydrological model can provide an effective basis for considering the inherent spatial variability among uplands and wetlands (Dai et al., 2012a). Simulating CH_4 fluxes is particularly difficult because of various interactions among controls of CH_4 production and transport from wetlands, including ebullition, that vary over very short distances such as 10 m or less (Bridgham et al., 2013). Correspondingly, uncertainties associated with plant carbon allocation and organic matter quality and decomposition impair the ability of field-scale biogeochemical models to predict CH_4 flux from the soil surface. These considerations are particularly important for small-scale models that are evaluated with field data.

Another major challenge to modeling carbon dynamics in wetlands is the inherent heterogeneity

of conditions within a wetland and the spatial heterogeneity of wetlands across the landscape. Accordingly, new approaches for accommodating high-resolution geospatial data with robust biogeochemical models are needed to provide capabilities to simulate wetland carbon dynamics at large scales. Such capabilities, in turn, would provide a basis for linking wetland biogeochemical models with atmospheric models (Gockede et al., 2010), thereby improving the basis for simulating the effects of climate change on wetland carbon. Large-scale bottom-up and top-down models are providing those capabilities to address CH_4 fluxes at the regional and global scales (Melton et al., 2013; Saunois et al., 2016; Bloom et al. 2017; Zhang et al., 2017a). However, estimates among the CH_4 models can vary considerably (Miller et al., 2016). Correspondingly, there is a real need for tools to assess wetland NEE; unfortunately, the large-scale models for assessing wetland NEE are not available or widely reported. Accordingly, ecosystem models must be upscaled to develop the components to simulate wetland NEE.

SUPPORTING EVIDENCE

KEY FINDING 1

The assessment of terrestrial wetland carbon stocks has improved greatly since the *First State of the Carbon Cycle Report* (CCSP 2007) because of recent national inventories and the development of a U.S. soils database. Terrestrial wetlands in North America encompass an estimated 2.2 million km^2, which constitutes about 37% of the global wetland area, with a soil and vegetation carbon pool of about 161 petagrams of carbon that represents approximately 36% of global wetland carbon stock. Forested wetlands compose 55% of the total terrestrial wetland area, with the vast majority occurring in Canada. Organic soil wetlands or peatlands contain 58% of the total terrestrial wetland area and 80% of the carbon (*high confidence, likely*).

Description of evidence base

Key Finding 1 is supported by an extensive analysis of the most current wetland soil and vegetation information available across the conterminous United States (CONUS), Alaska, Hawai'i, Puerto Rico, Canada, and Mexico, updating previous estimates made in SOCCR1 (see SOCCR2 Appendices 13A, p. 547 and 13B, p. 557).

Major uncertainties

Uncertainties are high where wetlands are present but not extensively mapped, such as in Alaska.

Assessment of confidence based on evidence and agreement, including short description of nature of evidence and level of agreement

Over much of the area under consideration, confidence is high that this assessment has accurately mapped carbon pools in mineral soil wetlands and peatlands.

Estimated likelihood of impact or consequence, including short description of basis of estimate

Understanding current carbon pools is critical in predicting how changes in, for example, climate, land use, and restoration will affect the carbon stored in terrestrial wetlands.

Summary sentence or paragraph that integrates the above information

Terrestrial wetlands are the largest reservoir of carbon in North America. Understanding the processes that lead to carbon storage and fluxes is important to predict how future changes will influence this large carbon pool and subsequent feedbacks to the atmosphere.

KEY FINDING 2

North American terrestrial wetlands currently are a carbon dioxide sink of about 123 teragrams of carbon (Tg C) per year, with approximately 53% occurring in forested systems. However, North American terrestrial wetlands are a natural source of methane (CH_4), with mineral soil wetlands emitting 56% of the estimated total of 45 Tg as CH_4 (CH_4-C) per year (*medium confidence, likely*).

Description of evidence base

Key Finding 2 and this chapter's narrative are based on the most recently reported wetland inventories integrated with reported values of soil carbon density (mass per unit area) and gaseous

fluxes of carbon dioxide (CO_2) and CH_4. Accordingly, the projections are dependent on estimates of wetland area and the pool and flux values assigned to the wetland types (see Appendices 13A, p. 547, and 13B, p. 557).

Major uncertainties

Similar to Key Finding 1, one major uncertainty is the mapped area, especially in areas with considerable wetlands that have not been adequately mapped. A second important uncertainty are the flux rates, which are applied globally to wetland types but are highly variable in time and space. Moreover, in many cases, few data exist.

Assessment of confidence based on evidence and agreement, including short description of nature of evidence and level of agreement

Confidence is medium, given both the incompleteness in mapping and variability in flux rates.

Estimated likelihood of impact or consequence, including short description of basis of estimate

Greenhouse gas fluxes from terrestrial wetlands in North America contribute to the global CO_2 and CH_4 cycles and associated climate forcing.

Summary sentence or paragraph that integrates the above information

Understanding both terrestrial wetland carbon pools (Key Finding 1) and net fluxes to the atmosphere (Key Finding 2) is critical because these wetlands are stable long-term carbon sinks and also an important source of CH_4.

KEY FINDING 3

The current rate of terrestrial wetland loss is much less than historical rates (about 0.06% of the wetland area from 2004 to 2009) with restoration and creation nearly offsetting losses of natural wetlands. Although area losses are nearly offset, there is considerable uncertainty about the functional equivalence of disturbed, created, and restored wetlands when comparing them to undisturbed natural wetlands. Correspondingly, there remains considerable uncertainty about the effects of disturbance regimes on carbon stocks and greenhouse gas (GHG) fluxes. For this reason, studies and monitoring systems are needed that compare carbon pools, rates of carbon accumulation, and GHG fluxes across disturbance gradients, including restored and created wetlands. Those studies will produce data that are needed for model applications (*high confidence, likely*).

Description of evidence base

The evidence for Key Finding 3 is from updated published literature for the United States and Mexico (Casasola 2008; Landgrave and Moreno-Casasola 2012; USFWS 2011) and the same data reported in SOCCR1 (CCSP 2007) for Canada. The amount of wetlands being restored is also a function of recent literature estimates (e.g., Dahl 2011). Disturbance also needs to be considered in the context of changes to carbon cycling processes.

Major uncertainties

Where wetlands are mapped well, the area of wetland loss is very certain. Some areas not mapped well, such as remote locations in Alaska, generally are not under threat from development, but changes in climatic conditions threatened the boreal region more than temperate and tropical

regions. However, the opposite is true for areas under development in Mexico. The amount of area being restored is also not tracked very well, especially when restoration fails. Crossing the gradient from disturbed to restored and/or created wetlands, there exists considerable uncertainty about the level of functions that those wetlands provide.

Assessment of confidence based on evidence and agreement, including short description of nature of evidence and level of agreement

There is high confidence that systems for reporting wetland losses and gains are accurate in the United States, but periodic inventories in other countries are lacking. Also, tracking the amount of wetlands that have been disturbed in some way is very difficult.

Estimated likelihood of impact or consequence, including short description of basis of estimate

Although the area of restored or created wetlands is small relative to the total wetland area of North America, the impact is likely important because understanding even small changes in wetland area is critical to scaling up carbon pools and fluxes.

Summary sentence or paragraph that integrates the above information

Although there are very reliable data that track wetland change across CONUS, no such data are available for Canada because regular wetland assessments for that country are lacking. In addition, field-based wetland mapping is generally poor in Alaska and Mexico, and restored and disturbed wetland areas also are difficult to track.

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Appendix 13A
Terrestrial Wetland Area and Carbon Pools

Prepared by Carl Trettin,[1] Wenwu Tang,[2] and Steven Campbell[3]
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13A.1 Introduction

This appendix provides the methodologies and data used to estimate the area and carbon pools of terrestrial wetlands in North America. Since the *First State of the Carbon Cycle Report* (SOCCR1; CCSP 2007), several developed geospatial databases have provided the opportunity to improve the estimation of carbon pools beyond what is feasible using area density factors. The development of the Gridded Soil Survey Geographic (gSSURGO) database by the U.S. Department of Agriculture's (USDA) Natural Resources Conservation Service (NRCS) was a particularly important advancement, availing gridded soil survey information for the United States and Puerto Rico. Similarly, the USDA Forest Service's Forest Inventory and Analysis (FIA) database uses forest biomass data for the United States, thereby facilitating its incorporation into carbon pool assessments. Sections 13A.2–13A.6 detail the data and methods used to obtain the reported wetland area and carbon pools.

13A.2 Conterminous United States

13A.2.1 Approach

The U.S. Fish and Wildlife Service's (USFWS) National Wetlands Inventory (NWI) was used as the basis for identifying terrestrial (i.e., nontidal) freshwater wetlands within the conterminous United States (CONUS) and for distinguishing between forested and nonforested wetlands. Subsequently, geospatial databases were used to calculate the carbon pools in soils and forests. Specifically, the gSSURGO database was used to calculate soil carbon, and the FIA database was used to calculate forest carbon based on the reported biomass. A carbon pool density factor was used for the nonforest vegetation biomass because an appropriate geospatial database was not available.

13A.2.2 Data

The datasets used for analyses of the wetland area and carbon pool computations are summarized in Table 13A.1, this page.

Table 13A.1. Source Datasets

Dataset	Year	Publisher	Download Link
Gridded Soil Survey Geographic (gSSURGO)	2016	U.S. Department of Agriculture (USDA) Natural Resources Conservation Service (NRCS)	gdg.sc.egov.usda.gov
National Wetlands Inventory (NWI)	2015	U.S. Fish and Wildlife Service	www.fws.gov/wetlands/Data/State-Downloads.html
Forest Inventory Analysis (FIA) Forest Biomass	2003	USDA Forest Service FIA	data.fs.usda.gov/geodata/rastergateway/biomass/index.php
Value-Added Look Up Table Database	2016	USDA NRCS	gdg.sc.egov.usda.gov
Cartographic Boundary	2015	U.S. Census Bureau	www.census.gov/geo/maps-data/data/cbf/cbf_state.html

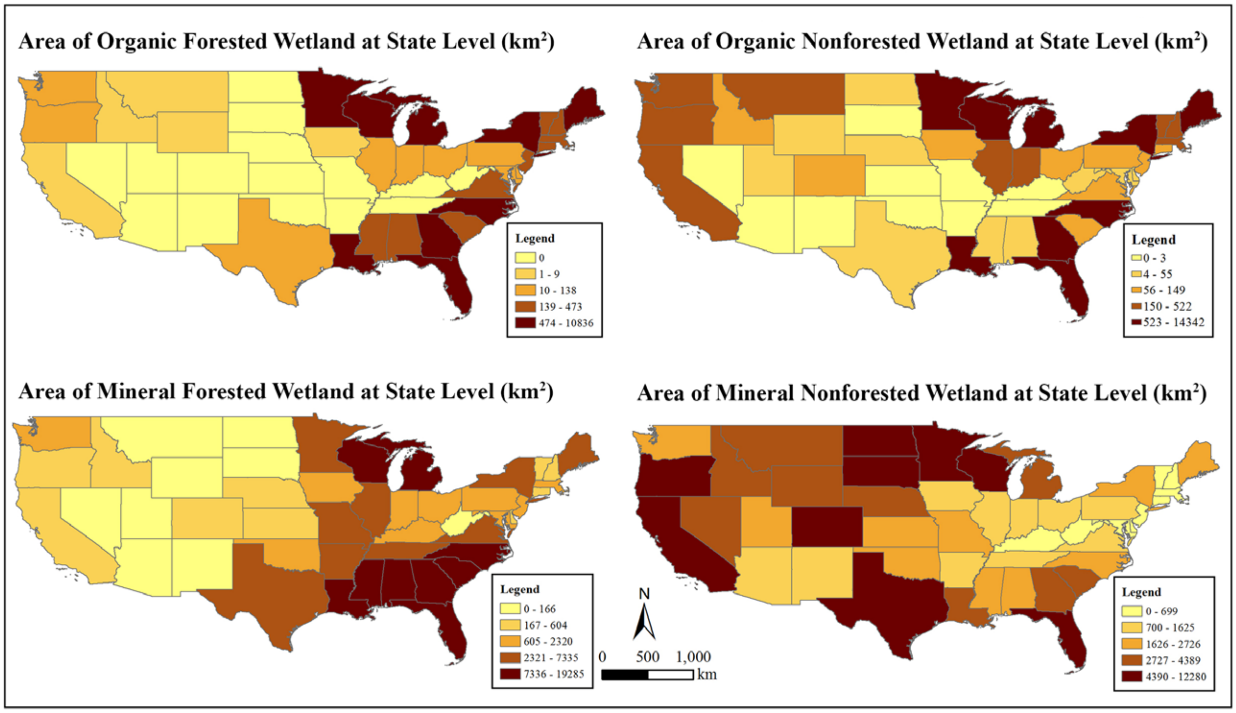

Figure 13A.1. Areal Distribution Among U.S. States of the Four Categories of Freshwater Terrestrial Wetlands. These wetland types are organic forested, organic nonforested, mineral forested, and mineral nonforested.

13A.2.3 Results

Wetland Area

According to NWI data, there are 395,197 km² of terrestrial freshwater wetlands in CONUS, 54% of which are forested and 46% nonforested (see Table 13A.2, this page). The estimate of forested freshwater wetlands is within 2% of the most recent NWI report; the total area of freshwater forested wetlands is calculated as 213,914 km², compared with 208,912 km² for 2009 from Dahl (2011). This area is smaller than the wetland area used in SOCCR1 (405,670 km²; CCSP 2007) because that report also included tidal wetlands. Mineral soils compose 79% of the terrestrial wetlands, with 21% being organic or peat soils (see Table 13A.2, this page). The distribution of wetlands among soil (organic and mineral) and vegetation (forest and nonforest) categories among states is presented in Figure 13A.1, this page.

Table 13A.2. Area of Forested and Nonforested Terrestrial Wetland and Related Soil Types in the United States

Soil Type	Forested Wetlands (km²)	Nonforested Wetlands (km²)	Total (km²)
Organic Soil	40,823	42,903	83,726
Mineral Soil	173,091	138,381	311,472
Total	213,914	181,283	395,197

The accuracy of the NWI data is considered to be over 90% for large wetlands (i.e., those > 1 hectare); uncertainties increase with smaller wetlands (Nichols 1994). Independent field-based studies also have been conducted to evaluate the accuracy of the NWI data for wetland mapping. The reported accuracies ranged from over 90% of overall accuracy in Michigan, Maine, and Massachusetts (see Kudray and Gale 2000; Nichols 1994; Swartwout et al., 1981) to underestimation of wetland area by 39% in Vermont

(see Morrissey and Sweeney 2006). With these issues considered, the NWI data are recognized as a reasonable source for estimating wetland area, particularly at large spatial extents, and thus are the source for national-level reporting.

Wetland Carbon Stock Estimation

Carbon stocks were calculated based on soil carbon content calculated from gSSURGO, forest biomass extracted from the FIA database, and a biomass density factor for nonforest vegetation. Forest vegetation consists of a carbon stock of about 0.878 petagrams of carbon (Pg C), with 79% occurring on mineral soils; nonforest vegetation contributed approximately 0.093 Pg C (see Table 13A.3, this page). Integrating forest biomass and soil carbon pools yields approximately 13.5 Pg C in terrestrial wetlands (see Table 13A.4, this page). The breakdown of carbon within forested and nonforested wetlands and of mineral and organic soils by state is summarized in Table 13A.4.

Table 13A.3. Carbon Stock in Forest and Nonforest Biomass Within Organic and Mineral Soil Terrestrial Wetlands[a]

Soil Type	Forest Carbon Pool (Pg C)	Nonforest Carbon Pool (Pg C)
Organic Soil	0.185	0.022
Mineral Soil	0.693	0.071
Total	0.878	0.093

Notes

a) Carbon stocks are measured in petagrams of carbon (Pg C) within the conterminous United States.

Table 13A.4. Carbon Stocks Within Organic and Mineral Soil, and Forested and Nonforested Freshwater Wetlands[a]

Soil Type	Forested Wetlands (Pg C)	Nonforested Wetlands (Pg C)	Total (Pg C)
Organic Soil	4.45	3.88	8.34
Mineral Soil	3.26	1.94	5.21
Total	7.71	5.82	13.55

Notes

a) Carbon stocks are measured in petagrams of carbon (Pg C) within the conterminous United States.

13A.3 Alaska

13A.3.1 Approach

The NWI and traditional soil surveys of Alaska are not available for the entire state. Fortunately, Clewley et al. (2015) recently published an inventory of wetlands based on remote-sensing data that used the Cowardin Classification system for representing the distribution of wetland types. Similarly, NRCS has produced a gSSURGO dataset for Alaska. Accordingly, those datasets were used as the basis for estimating the terrestrial wetland categories and carbon stocks following the same general approach used for CONUS. The combination of the wetland and carbon stock assessment with the distribution of frozen wetlands is considered to provide a comprehensive assessment of wetlands for the state.

13A.3.2 Data

Table 13A.5, p. 550, presents the principal datasets used in this study that include information on soil, wetlands, soil organic carbon, permafrost, and elevation.

13A.3.3 Results

Wetland Area

The total area of freshwater wetlands in Alaska, based on the Clewley et al. (2015) database, is 579,645 km^2 (see Table 13A.6, p. 550). The wetland data were classified from ALOS PALSAR[2] remote-sensing data using a random forest-based classifier. The data were processed using the adjustment factor employed by Clewley et al. (2015) to calculate the total area of freshwater wetlands, and data that overlapped into Canada were excluded. The overall accuracy of the classification is 84.5% for distinguishing specific wetland types and 94.7% for distinguishing wetlands with uplands (Clewley

[2] Advanced Land Observing Satellite-1 (ALOS) Phased Array type L-band Synthetic Aperture Radar (PALSAR)

Table 13A.5. Datasets Used to Estimate the Distribution and Carbon Stocks of Alaskan Terrestrial Wetlands[a–b]

Dataset	Year	Publisher	Download Link
Alaska Wetlands (Clewley et al., 2015)	2007	Alaska Satellite Facility	www.asf.alaska.edu/sar-data/palsar
STATSGO2	2014	U.S. Department of Agriculture (USDA) Natural Resources Conservation Service	www.nrcs.usda.gov/wps/portal/nrcs/detail/soils/survey/geo/?cid=nrcs142p2_053629
Organic Soil Probability	2016	U.S. Geological Survey (USGS) LandCarbon	pubs.er.usgs.gov/publication/pp1826
Forest Biomass	2002	USDA Forest Service Forest Inventory and Analysis	data.fs.usda.gov/geodata/rastergateway/biomass
Probability of Near-Surface 1-m Permafrost	2015	USGS[a]	sciencebase.gov/catalog/item/5602ab5ae4b03bc34f5448b4
STATSGO Depth of Permafrost	2012	USGS[a]	ckan.snap.uaf.edu/dataset/depth-to-permafrost-alaska-landcarbon-project
STATSGO Permafrost Soil	2014	USDA Natural Resources Conservation Service[b]	www.nrcs.usda.gov/wps/portal/nrcs/detail/soils/survey/geo/?cid=nrcs142p2_053629
Alaska State Boundary	2016	U.S. Census Bureau	www.census.gov/geo/maps-data/data/cbf/cbf_state.html
Elevation	1996	USGS	agdc.usgs.gov/data/usgs/erosafo/dem/dem.html

Notes

a) Provided by Neal Pastick, USGS.

b) Provided by Steve Campbell, USDA Natural Resources Conservation Service.

et al., 2015). The NWI class was used to aggregate the areas into forested and nonforested types.

Also calculated was the total area of wetlands in Alaska from STATSGO2 data using the percent in hydric soil attribute ("hydric_pct"; i.e., the percent in hydric soil). The total area is 587,143.9 km^2 based on the STATSGO2 percentage of hydric soils, which is very close to that provided by the Clewley et al. (2015) dataset.

Soil organic carbon data from STATSGO2 were employed to estimate the area of organic soils in Alaska, using the variable named "hydric_org_pct" (i.e., the percent in hydric organic soil) as the basis. This variable was multiplied by the area of map units (polygons) in the STATSGO2 dataset to obtain the area of peatland within each map unit. The total area of peatlands estimated from STATSGO2 using the hydric organic soil attribute is 107,057 km^2.

Table 13A.6. Area of Four Terrestrial Wetland Types in Alaska

Soil Type	Forested (km^2)	Nonforested (km^2)	Total (km^2)
Organic	9,947	97,111	107,057
Mineral	54,858	417,729	472,587
Total	64,805	514,840	579,645

Incorporating the distribution of organic soils into the overlay analyses yielded the distribution and area of the four wetland categories (see Figure 13A.2, p. 551). The total area of the four wetland

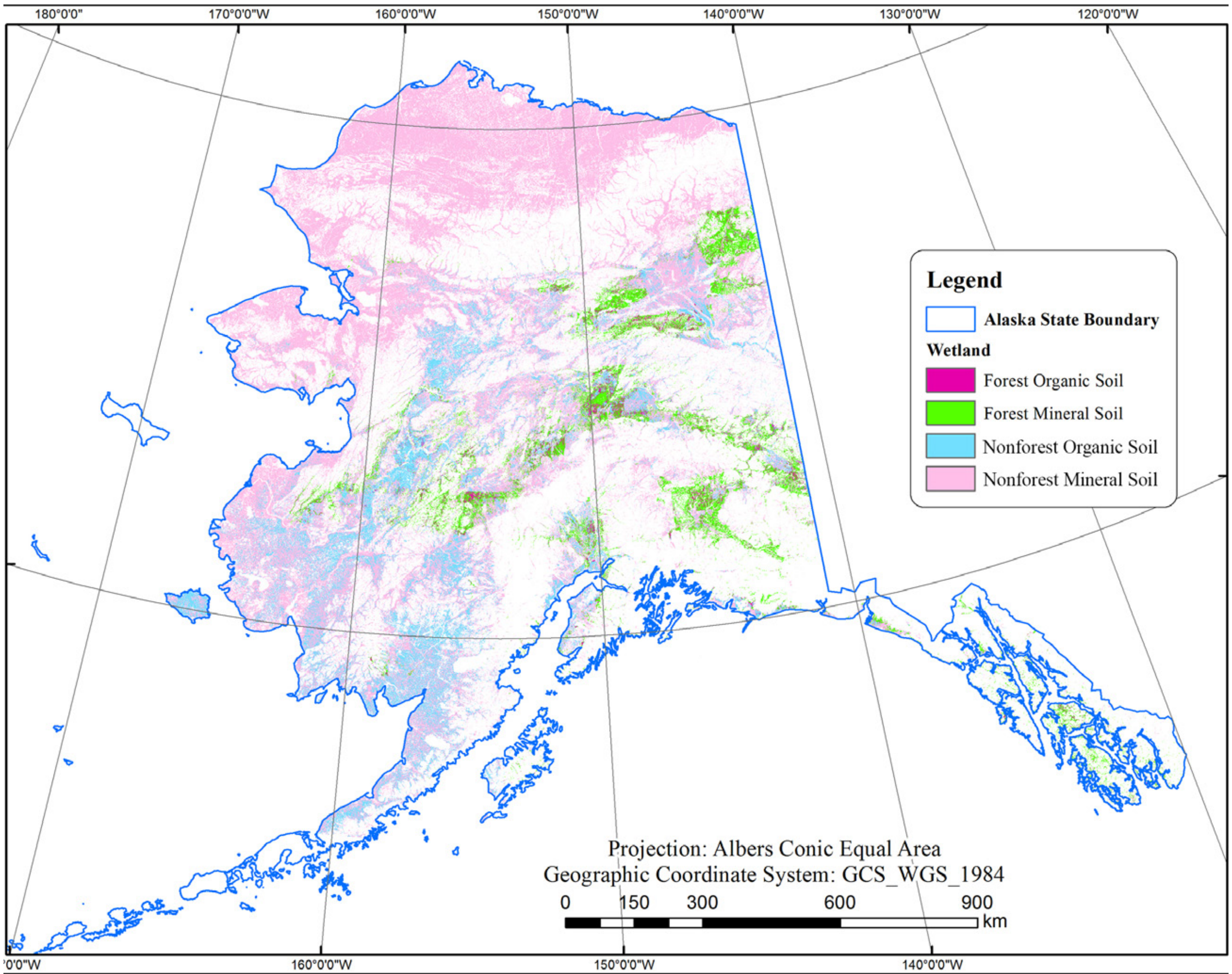

Figure 13A.2. Areal Distribution in Alaska of the Four Categories of Terrestrial Wetlands. These wetland types are forest organic soil, forest mineral soil, nonforest organic soil, and nonforest mineral soil.

categories of freshwater wetlands in Alaska are summarized in Table 13A.6, p. 550.

Assessing the overlap of wetlands and permafrost areas provided a basis for distinguishing carbon stocks. The use of the USGS probability map of permafrost provided a cut-off threshold of 60% to permafrost occurring within 1 m of the surface (with a 30-m spatial resolution). The resultant area of permafrost is 405,891 km^2, compared with 548,503 km^2 based on permafrost 2 m in depth from STATSGO2 data. Overlaying the USGS permafrost area with the wetlands shows that the total area of wetlands within the permafrost region is 267,887 km^2, which is approximately 46% of the total wetland area. The areas of the four types of freshwater wetlands in Alaska within permafrost or nonpermafrost regions are presented in Table 13A.7, p. 552.

Wetland Carbon Stocks

Ecosystem carbon stocks for the four wetland categories were derived from soil carbon stocks from USDA STATSGO data, biomass carbon data from FIA for forests, and a density factor for nonforested wetlands (see Table 13A.8, p. 552).

Table 13A.7. Distribution of Wetland Types Among Areas With and Without Permafrost in Alaska

Soil Type		Forested (km^2)	Nonforested (km^2)	Total (km^2)
Permafrost	Organic	4,199	23,274	27,474
	Mineral	14,696	225,716	240,413
	Total	18,895	248,991	267,887
Nonpermafrost	Organic	5,747	73,836	79,584
	Mineral	40,162	192,013	232,175
	Total	45,910	265,849	311,759

Table 13A.8. Total Carbon Pool of the Four Wetland Categories in Alaska[a]

Soil Type	Forested (Pg C)	Nonforested (Pg C)	Total Carbon (Pg C)
Organic	0.70	7.09	7.79
Mineral	2.80	21.21	24.01
Total	3.50	28.31	31.80

Notes

a) Carbon stocks are measured in petagrams of carbon (Pg C).

Partitioning the ecosystem carbon pools among wetlands in permafrost and nonpermafrost zones is provided in Table 13A.9, p. 553. Approximately 46% of the wetland carbon pool occurs within the permafrost areas.

13A.4 Puerto Rico

13A.4.1 Approach

The approaches to quantifying the distribution of terrestrial wetlands and the associated carbon pools for Puerto Rico follow those of CONUS, where a suite of datasets was used, including gSSURGO, NWI, Value-Added Look Up Table Dataset, Cartographic Boundary Shapefile, and FIA Forest Biomass Dataset. An overlay analysis was conducted between NWI and gSSURGO to identify vegetation and soil types for wetlands. Cartographic Boundary identified the boundary of Puerto Rico. The FIA Forest Biomass dataset provided the forest biomass information. Soil Data Development Tools for ArcGIS were used to extract the soil class of freshwater wetlands.

13A.4.2 Data

Datasets used in this study are summarized in Table 13A.10, p. 553.

13A.4.3 Results

Wetland Area

The total area of terrestrial wetlands derived from NWI data is 311.4 km^2. However, gSSURGO data coverage was missing for approximately 9.8% of the terrestrial wetland area. Distributing the area of missing soil data among the forested and non-forested categories yields the final area of the four wetland categories (see Table 13A.11, p. 553).

Ecosystem Carbon Pool

Ecosystem carbon pools, including soil and biomass, for freshwater wetlands in Puerto Rico are summarized in Table 13A.12, p. 553.

13A.5 Canada

13A.5.1 Approach

Canadian terrestrial freshwater wetlands were estimated based on a combination of spatial data because there was not a single wetland database that could produce estimates of organic and mineral soil wetlands and of forest and nonforest vegetation.

13A.5.2 Data

Datasets in this study are summarized in Table 13A.13, p. 554.

Table 13A.9. Ecosystem Carbon Pools in Freshwater Wetlands Occurring in Permafrost and Nonpermafrost Areas in Alaska[a]

Soil Type		Forested (Pg C)	Nonforested (Pg C)	Total Carbon (Pg C)
Permafrost	Organic	0.27	1.56	1.83
	Mineral	0.83	11.87	12.70
	Total	1.11	13.43	14.53
Nonpermafrost	Organic	0.42	5.54	5.96
	Mineral	1.97	9.34	11.30
	Total	2.39	14.88	17.26

Notes

a) Carbon stocks are measured in petagrams of carbon (Pg C).

Table 13A.10. Datasets Used to Estimate Terrestrial Wetland Area and Carbon Pools in Puerto Rico

Dataset	Year	Provider	Download Link
Gridded Soil Survey Geographic (gSSURGO)	2016	U.S. Department of Agriculture (USDA) Natural Resources Conservation Service	gdg.sc.egov.usda.gov
National Wetlands Inventory	2010	U.S. Fish and Wildlife Service	www.fws.gov/wetlands/Data/State-Downloads.html
Forest Biomass	2008	USDA Forest Service's Forest Inventory and Analysis	data.fs.usda.gov/geodata/rastergateway/biomass
Puerto Rico Boundary	2016	U.S. Census Bureau	www.census.gov/geo/maps-data/data/cbf/cbf_state.html

Table 13A.11. Area of Terrestrial Wetland Categories in Puerto Rico

Soil Type	Forested (km^2)	Nonforested (km^2)	Total (km^2)
Organic Soil	0.67	8.4	9.1
Mineral Soil	49.9	252.3	302.3
Total	50.6	260.7	311.4

Table 13A.12. Ecosystem Carbon Pools Among the Four Terrestrial Wetland Categories in Puerto Rico[a]

Soil Type	Forested (Pg C)	Nonforested (Pg C)	Total (Pg C)
Organic Soil	0.000	0.001	0.001
Mineral Soil	0.001	0.006	0.007
Total	0.001	0.007	0.008

Notes

a) Carbon pools are measured in petagrams of carbon (Pg C).

13A.5.3 Results

Organic and Mineral Soil in Forested and Nonforested Terrestrial Wetlands in Canada

Organic and mineral soils for forested and nonforested wetlands were estimated by overlaying land-cover datasets (GLWD and North America land-cover data) with soil datasets (FAO soil data, Peatland Database of Canada, and Soil Landscape of Canada). Those analyses routinely underestimated wetland area compared with estimates in published reports, especially for organic soils (Tarnocai 2006; Warner 2005; see Table 13A.14, p. 554, for examples of the differences in wetland area based on data sources).

Table 13A.13. Datasets Used in Canadian Terrestrial Wetland Assessment				
Code[a]	**Dataset**	**Year**	**Publisher**	**Download Link**
W_1	North America Land Cover	2010	U.S. Geological Survey	landcover.usgs.gov/nalcms.php
W_2	Global Lakes and Wetlands Database Level 3 (GLWD-3)	2004	World Wild Life Organization; The Center for Environmental Systems Research, University of Kassel, Germany	worldwildlife.org/pages/global-lakes-and-wetlands-database
S_1	FAO/UNESCO[b] Digital Soil Map of the World 3.6	2007	Food and Agriculture Organization of the United Nations	fao.org/geonetwork/srv/en/metadata.show?id=14116
S_2	Soil Landscapes of Canada 3.2	2010	Agriculture and Agri-Food Canada	sis.agr.gc.ca/cansis/nsdb/slc/v3.2/index.html
S_3	Peatlands of Canada	2005	Natural Resources Canada	geogratis.gc.ca/api/en/nrcan-rncan/ess-sst/4e9e791c-ebad-594a-a3ba-14b8b974f239.html

Notes

a) The W_1 and W_2 and S_1, S_2, and S_3 abbreviations are used in this and subsequent tables to indicate, respectively, the wetlands and soils datasets outlined here.

b) Key: FAO, U.N. Food and Agriculture Organization; UNESCO, United Nations Educational, Scientific and Cultural Organization.

Table 13A.14. Areas of Forested Wetland and Nonforested Terrestrial Wetland and Related Soils in Canada[a–b]						
Soil Type	***W_1 * S_1* (km²)**			***W_1 * S_2* (km²)**		
	Forested	**Nonforested**	**Total**	**Forested**	**Nonforested**	**Total**
Organic Soil	582,078	194,895	776,973	499,271	35,692	534,963
Mineral Soil	215,794	40,933	256,727	360,249	21,345	381,594
Total	797,872	235,828	1,033,700	859,520	57,037	916,557
Soil Type	***W_2 * S_1* (km²)**			***W_2 * S_2* (km²)**		
	Forested	**Nonforested**	**Total**	**Forested**	**Nonforested**	**Total**
Organic Soil	503,810	187,765	691,575	351,529	32,084	383,613
Mineral Soil	161,886	38,960	200,846	193,374	17,685	211,059
Total	665,696	226,725	892,421	544,903	49,769	594,672

Notes

a) Areas estimated using different data sources.

b) W_1: 2010 North America Land Cover dataset (wetland class available); W_2: Global Lakes and Wetlands Database; S_1: FAO/UNESCO Digital Soil Map of the World; S_2: Soil Landscapes of Canada; S_3: Peatlands of Canada dataset.

Asterisk (*) denotes the use of multiple datasets (GIS-based overlay analysis applied).

Table 13A.15. Areas of Forested and Nonforested Wetland and Related Soil in Canada from Peatland Dataset (S_3)[a]

Soil Type	Forested (km^2)	Nonforested (km^2)	Total (km^2)
Organic Soil	703,785	415,450	1,119,235
Mineral Soil	268,337	103,932	372,270
Total	972,122	519,382	1,491,505

Notes
a) S_3, Peatlands of Canada dataset.

Table 13A.16. Carbon Pools of Forested and Nonforested Wetland and Peat and Mineral Soils in Canada[a]

Soil Type	Forested (Pg)	Nonforested (Pg)	Total[a]
Organic Soil	76.7	37.8	114.5
Mineral Soil	5.1	9.5	14.6
Total	81.8	47.3	129.0

Notes
a) Carbon pools are calculated in petagrams (Pg).

Table 13A.17. List of Datasets Used to Assess the Area of Terrestrial Wetlands in Mexico

Dataset	Year	Publisher	Download Link
North America Land Cover	2010	U.S. Geological Survey, Natural Resources Canada, Insituto Nacional de Estadística y Geografía (INEGI), Comisión Nacional para el Conocimiento y Uso de la Biodiversidad (CONABIO), and Comisión Nacional Forestal (CONAFOR)	landcover.usgs.gov/nalcms.php
Mapa Potencial de Humedales	2012	INEGI	www.inegi.org.mx/geo/contenidos/recnat/humedales/datosvec.aspx

Because the accepted area of peatlands is 1,135,610 km^2 as reported by Tarnocai (2006), it was used as the basis for the total peatland area; the 16,375 km^2 of permafrost peatlands (Tarnocai et al., 2011) were excluded from the final area table (see Table 13A.15, this page). Wetland-specific soil types from the Peatlands of Canada and the Soil Landscapes of Canada datasets were used to identify mineral and organic soil wetlands. The analysis of wetland area in Canada is based on the Peatlands of Canada database, which was updated from its previous version. The accuracy of the wetland area estimated using this database is equal to or greater than 66%, as suggested by Tarnocai (2009). The distribution of terrestrial freshwater wetlands in Canada is presented in Table 13A.15. For comparison, Warner (2005) reported 1.056 million km^2 of peatland area (organic soil wetland) for Canada, a difference of 7%.

Carbon Pools

Carbon pools of the Canadian wetlands were calculated using the area carbon density factors for the four wetland categories, derived from CONUS (see Table 13A.16, this page).

13A.6 Mexico

13A.6.1 Approach

An assessment of terrestrial wetlands in Mexico was used as the basis for identifying wetland areas and soil types. The North American Land Cover dataset (see Table 13A.17, this page) and a recent dataset from Mexico were used to segregate the wetlands into vegetation categories. Area carbon density factors were used to develop the estimates of wetland carbon pools.

13A.6.2 Data

The datasets used to estimate the area of terrestrial wetlands in Mexico are presented in Table 13A.17.

Table 13A.18. Area of Freshwater Wetlands in Mexico Categorized by Soils and Vegetation

Soil Type	Forested (km²)	Nonforested (km²)	Total (km²)
Organic Soil	3,394	17,191	20,585
Mineral Soil	5,288	10,320	15,608
Total	8,682	27,511	36,193

13A.6.3 Results

Organic and Mineral Soil in Forested and Nonforested Wetlands in Mexico

This estimate of freshwater wetlands is greater than other reported values (e.g., 31,000 km^2; Bridgham et al., 2006). A review of the map units from the Mapa Potencial de Humedales could not ensure that selected wetlands were adequately constrained to freshwater systems (due to problems with data code translations). Accordingly, the calculated wetland area was reduced by 25% to provide a conservative estimate (see Table 13A.18, this page), thereby reducing the accuracy to at least 75%. The metadata for the database did not provide an estimate of the mapping error.

Acknowledgments

Spatial analyses were conducted at the Center for Applied GIScience, University of North Carolina, Charlotte, by Yu Lan, Jiyang Shi, Wenpeng Feng, Yu Lan, Douglas Shoemaker, Minrui Zheng, and Xiang Zhao. Julie Arnold, USDA Forest Service Southern Research Station, assisted with the literature and preparation of data tables and spreadsheets. Also used were computational facilities at the Portland office of the USDA Natural Resources Conservation Service.

Appendix 13B
Terrestrial Wetland–Atmosphere Exchange of Carbon Dioxide and Methane

Prepared by Carl Trettin,[1] Judy Drexler,[2] Randall Kolka,[1] Scott Bridgham,[3] Sheel Bansal,[2] Brian Tangen,[2] Brian Bescoter,[4] Wenwu Tang,[5] and Steven Campbell[6]
[1]USDA Forest Service; [2]U.S. Geological Survey; [3]University of Oregon; [4]Florida Atlantic University; [5]University of North Carolina, Charlotte; [6]USDA Natural Resources Conservation Service

13B.1 Introduction

This chapter used published observational studies and recent syntheses to develop the basis for estimating both the net uptake of atmospheric carbon dioxide (CO_2) by terrestrial wetlands, which is equal to negative net ecosystem exchange (NEE), and the net fluxes of methane (CH_4) from terrestrial wetlands to the atmosphere. The primary source documents were the *First State of the Carbon Cycle Report* (SOCCR1; CCSP 2007) and the recent Intergovernmental Panel on Climate Change (IPCC) Wetlands Supplement (IPCC 2013). That information was augmented where possible with additional references. There were very few recent reports of measured NEE in comparison to reports on CH_4 flux. Accordingly, there was reliance on the previously published synthesis, with considerable uncertainty remaining in the NEE estimates. Tropical wetland fluxes were derived from the recent synthesis by Sjögersten et al. (2014).

Section 13B.2, this page, summarizes the observational data used as the basis for the area density flux factors. The flux estimates were based on those data and specific references, depending on the assessment area. Section 13B.3, p. 558, presents the area density flux factors used for each country and region.

Table 13B.1 Average Methane and Net Ecosystem Exchange for Nonforested and Forested Wetlands on Peat Soils[a–c]

Wetland Area	Average	Standard Error	n
CH_4 (g CH_4-C per m^2 per Year)			
Nonforested	23.6	3.1	73
Forested	8.9	5.2	14
NEE (g C per m^2 per Year)			
Nonforested	–135.0	42.5	14
Forested	–124.7	43.1	5

Notes
a) Negative net ecosystem exchange (NEE) indicates net transfer to the ecosystem.
b) See Tables 13B.8 and 13B.9 in Supplement, p. 561, for values and references.
c) Key: CH_4, methane; C, carbon; g, gram; n, number of studies.

13B.2 Literature Review
13B.2.1 Peat Soils

The mean CH_4 and NEE are presented in Table 13B.1, this page. The mean CH_4 flux rate for nonforested and forested wetlands are 23.6 and 8.9 grams (g) of CH_4-C per m^2 per year, respectively. In comparison, the mean CH_4 flux rate used for peatlands in SOCCR1 was 1.9 g CH_4-C per m^2 per year. The difference in CH_4 flux rates is attributable to the additional references and the wide range in conditions from the reported studies. The mean NEE for the nonforested and forested wetlands are –135.0 and –124.7 g C per m^2 per year, respectively. However, there are relatively few reports of measured NEE from peatlands; hence, the basis provided by the published studies is relatively weak. For SOCCR1, NEE was estimated on the basis of net changes in soil

and plant carbon, yielding an NEE of –19.0 to –121.0 g C per m^2 per year for northern and temperate peatlands (CCSP 2007). Plant carbon accumulation was considered negligible for the northern biomes, due the paucity of data. Accordingly, soil carbon accumulation accounted for 100% of the gain in the northern peatlands and 58% in the temperate peatlands.

13B.2.2 Mineral Soils

The mean CH_4 and NEE fluxes for mineral soil wetlands are presented in Table 13B.2, this page. The mean CH_4 flux rate for nonforested and forested wetlands are 26.1 and 26.9 g CH_4-C per m^2 per year, respectively. In comparison, the mean CH_4 flux rate used for mineral wetlands in SOCCR1 (CCSP 2007) was 6 g CH_4-C per m^2 per year. As was the case with the peatlands, the variation in CH_4 flux rates is due to the wide range in conditions from the reported studies. The mean NEE for the nonforested areas is –102.1 g C per m^2 per year. There were too few reports of measured NEE for mineral soil forests; hence, another metric was used. In SOCCR1, NEE was estimated on the basis of net changes in soil and plant carbon, yielding an NEE of –17 to –67 g C per m^2 per year, for northern and temperate mineral soil wetlands, respectively (CCSP 2007). For that analysis, plant carbon accumulation was considered negligible for the northern biomes, due in large part to the paucity of data. Accordingly, soil carbon accumulation accounted for 100% of the gain in the northern mineral soil wetlands and 25% in the temperate mineral soil wetlands.

Table 13B.2. Methane and Net Ecosystem Exchange Means and the Associated Standard Errors for Nonforested and Forested Wetlands on Mineral Soils[a–c]

Wetland Area	Mean	Standard Error	n
CH_4 (g CH_4-C per m^2 per Year)			
Nonforested	26.1	3.6	46
Forested	26.9	7.9	16
NEE (g C per m^2 per Year)			
Nonforested	–102.1	34.4	13
Forested	NA[d]	NA	

Notes

a) Negative net ecosystem exchange (NEE) indicates net transfer to the ecosystem.

b) See Tables 13B.10 and 13B.11 in Supplement, p. 561, for values and references.

c) Key: CH_4, methane; C, carbon; g, gram; n, number of studies.

d) Not applicable.

13B.3 Country and Regional Density Factors

13B.3.1 Conterminous United States

Carbon flux within the conterminous United States (CONUS) was estimated using area carbon flux density factors (see Table 13B.3, this page). The NEE flux density factors are based on the mean for the peat soil nonforested wetland and mineral

Table 13B.3. Flux Density Factors Used to Estimate Net Ecosystem Exchange and Methane Fluxes from Freshwater Wetlands in the Conterminous United States[a–d]

Flux	Organic Soil		Mineral Soil	
	Forested	Nonforested	Forested	Nonforested
NEE (g CO_2-C per m^2 per Year)	–120.97 (45.60)	–134.97 (42.53)	–66.99 (23.55)	–102.15 (34.43)
CH_4 (g CH_4-C per m^2 per Year)	8.90 (5.24)	23.58 (3.13)	26.93 (7.95)	26.09 (3.60)

Notes

a) Negative net ecosystem exchange (NEE) indicates net transfer to the ecosystem.

b) Standard error in parentheses.

c) Source: Appendix 13B Supplement: Carbon Pools and Fluxes, p. 561.

d) Key: CO_2, carbon dioxide; CH_4, methane; g, gram; C, carbon.

Table 13B.4. Area Density Factors Used to Estimate Net Ecosystem Exchange and Methane Flux from Freshwater Wetlands in Alaska[a–d]

Flux	Organic		Mineral	
	Forested	Nonforested	Forested	Nonforested
NEE ($g\ CO_2$-C per m^2 per Year)	–56.53 (32.14)	–56.53 (32.14)	–56.53 (32.14)	–56.53 (32.14)
CH_4 ($g\ CH_4$-C per m^2 per Year)	8.90 (5.24)	23.58 (3.13)	26.93 (7.95)	26.08 (3.60)

Notes

a) Negative net ecosystem exchange (NEE) indicates net transfer to the ecosystem.

b) Standard error in parentheses.

c) Source: Appendix 13B Supplement: Carbon Pools and Fluxes, p. 561.

d) Key: CO_2, carbon dioxide; CH_4, methane; g, gram; C, carbon.

soil nonforested wetlands (see Tables 13B.1 and 13B.2, p. 557 and p. 558, respectively). To estimate NEE for the forested wetlands, the SOCCR1 values (Bridgham et al., 2007) were used due to the small number of field-based reports. The estimate in SOCCR1 was based on the annual change in soil and plant carbon; the conservative estimate of 50 g C per m^2 per year sequestered in forests was used for both peat and mineral soil wetlands (Bridgham et al., 2007). The small number of studies that directly measure NEE in wetlands remains a constraint; hence, the segmented approach used by Bridgham et al. (2007) provides a functional basis.

The CH_4 flux density factors are based on the mean of data reported for the four wetland categories (see Section 13B.2, p. 557). These mean flux factors are similar to those used in SOCCR1 (CCSP 2007), where the mean for freshwater wetlands was 5.3 g CH_4-C per m^2 per year.

13B.3.2 Alaska

The available data for establishing the carbon flux for Alaska is very limited. The area density factor for NEE employs the values reported by He et al. (2016), which are based on simulation results (see Table 13B.4, this page). For the CH_4 flux, the mean values used were derived from the literature compilation (see Section 13B.2, p. 557). In comparison, He et al. (2016) estimated the CH_4 flux at 47.5 g C per m^2 per year, an amount which is almost twice the value used here; the paucity of data determined use of the more conservative CH_4 flux estimate based on field measurement data.

Table 13B.5. Area Density Factors Used to Estimate Net Ecosystem Exchange and Methane Flux for Tropical Terrestrial Wetlands[a–d]

Wetland Type	NEE	CH_4 Flux
	g C per m^2 per Year	
Organic Soil Wetland	–310.3 (152.8)	40.1 (17.1)
Mineral Soil Wetland	–120.8 (218.2)	54.0 (9.7)

Notes

a) Negative net ecosystem exchange (NEE) indicates net transfer to the ecosystem.

b) Standard error in parentheses.

c) Source: Sjögersten et al. (2014).

d) Key: C, carbon; g, gram; CH_4, methane.

13B.3.3 Puerto Rico

Estimates of NEE and CH_4 fluxes (see Table 13B.5, this page) were obtained using area density factors for mineral and organic soils derived from the synthesis of tropical wetlands provided by Sjögersten et al. (2014). The same area density factors were used for forested and nonforested wetlands.

Table 13B.6. Area Density Factors Used to Estimate Net Ecosystem Exchange and Methane Flux from Freshwater Wetlands in Canada[a–c]

Flux	Organic		Mineral	
	Forested	Nonforested	Forested	Nonforested
NEE ($g\ CO_2$-C per m^2 per Year)	–47.71 (4.18)	–16.71 (4.18)	–47.98 (12.74)	–102.15 (34.44)
CH_4 ($g\ CH_4$-C per m^2 per Year)	8.90 (5.24)	23.58 (3.13)	26.93 (7.95)	26.09 (3.60)

Notes

a) Negative net ecosystem exchange (NEE) indicates net transfer to the ecosystem.

b) Standard error in parentheses.

c) Key: CH_4, methane; CO_2, carbon dioxide; g, gram; C, carbon.

13B.3.4 Canada

Carbon flux for Canada was estimated using area carbon flux density factors (see Table 13B.6, this page) calculated on the basis of reported values. The area density factor for NEE in nonforested peatlands and mineral soil wetlands uses the mean reported from measurement studies (see Section 13B.2, p. 557). For forested wetlands, the value reported in SOCCR1 was used, reflecting the soil carbon accretion, to which was added 31 g C per m^2 per year sequestered in vegetation, an amount which is based on an 18-year assessment of Canadian forests (Stinson et al., 2011). The analyses of Stinson et al. (2011) did not include changes in soils as a result of bryophytes or sedimentation; hence, adding the soil component seemed appropriate because it was the only component used in SOCCR1 (CCSP 2007).

The CH_4 flux density factors are based on the data average reported for the four categories (see Section 13B.2, p. 557). These mean flux factors for peatlands are higher than the factor used in SOCCR1 (2.8 g C per m^2 per year). For freshwater wetlands, the SOCCR1 CH_4 flux was 5.3 g CH_4-C per m^2 per year, which is considerably lower than the forested and nonforested values (CCSP 2007).

Table 13B.7. Area Density Factors Used to Estimate Net Ecosystem Exchange and Methane Flux for Mexico[a–d]

Wetland Type	NEE	CH_4 Flux
	g C per m^2 per Year	
Organic Soil Wetland	–310.3 (152.8)	40.1 (17.1)
Mineral Soil Wetland	–120.8 (218.2)	54.0 (9.7)

Notes

a) Negative net ecosystem exchange (NEE) indicates net transfer to the ecosystem.

b) Standard error in parentheses.

c) Source: Sjögersten et al. (2014).

d) Key: CH_4, methane; g, gram; C, carbon.

13B.3.5 Mexico

Estimates of NEE and CH_4 fluxes (see Table 13B.7, this page) were obtained using area density factors for mineral and organic soils derived from the synthesis of tropical wetlands developed by Sjögersten et al. (2014). The negative number for NEE indicates net uptake by the ecosystem. The same area density factors were used for forested and nonforested wetlands.

Appendix 13B Supplement: Carbon Pools and Fluxes

Tables 13B.8–13B.11

Table 13B.8. Forested Peatland Area Density Flux Factors[a–b]

Location	Vegetation Type	NEE Emission (g CO_2-C per m^2 per Year)	CH4 Emission (g CH_4-C per m^2 per Year	Reference
New York	Forested peatland		0.150	Coles and Yavitt (2004)
Minnesota	Forest bog hummock		2.625	Dise (1993)
Minnesota	Forest bog hollow		10.350	Dise (1993)
Minnesota	Forest bog hollow		3.513	Dise (1992)
Minnesota	Hummock		1.317	Dise (1992)
Wisconsin	Forest bog	–80.0	0.800	Desai et al. (2015)
West Siberia	Pine peatland		0.132	Golovatskaya and Dyukarev (2008)
West Siberia	Stunted pine peatland		0.198	Golovatskaya and Dyukarev (2008)
Southern Germany	Bog	–62.0	5.300	Hommeltenber et al. (2014)
Boreal	Swamp	–256.0		Lu et al. (2017); Lund et al. (2010)
Boreal	Swamp	–195.5		Lu et al. (2017); Sulman et al. (2012); Syed et al. (2006)
Temperate	Bog	–30.0		Lu et al. (2017); Sulman et al. (2012); Syed et al. (2006)
West Virginia	Appalachian bog		74.646	Wieder et al. (1990)
Florida	Swamp		2.026	Villa and Mitsch (2014)
Florida	Swamp		1.661	Villa and Mitsch (2014)
Maryland	Appalachian bog		19.320	Wieder et al. (1990)
West Virginia	*Sphagnum*/Forest		2.625	Yavitt et al. (1990)

Notes

a) Negative net ecosystem exchange (NEE) indicates net transfer to the ecosystem.

b) Key: CO_2, carbon dioxide; CH_4, methane; g, gram; C, carbon.

Table 13B.9. Nonforested Peatland Area Density Flux Factors[a]

Location	Vegetation Type	Annual Flux (CO_2 g C per m^2 per Year)	Annual Flux (CH_4 g C per m^2 per Year)	Reference
Minnesota	Open bog		61.473	After Crill et al. (1988); after Mitsch and Wu (1995)
Minnesota	Natural fen		65.864	After Crill et al. (1988); after Mitsch and Wu (1995)
Minnesota	Acid fen		21.077	After Crill et al. (1988); after Mitsch and Wu (1995)
West Virginia	Mountain bog		51.374	After Gorham (1991); after Crill et al. (1988)
Minnesota	Bog		36.006	After Harriss et al. (1985)
Minnesota	Fen		1.098	After Harriss et al. (1985)
California	Marsh	–412.5	56.300	Anderson et al. (2016)
Minnesota	Open bog		0	Bridgham et al. (1995)
New Hampshire	Poor fen		82.950	Carroll and Crill (1997)
Boreal Canada	Swamp		0.922	Derived from Moore and Roulet (1995)
Boreal Canada	Fen		2.503	Derived from Moore and Roulet (1995)
Boreal Canada	Bog		1.713	Derived from Moore and Roulet (1995)
Minnesota	Fen Lagg		9.450	Dise (1993)
Minnesota	Bog (open bog)		32.325	Dise (1993)
Minnesota	Fen (open poor fen)		49.275	Dise (1993)
Minnesota	Open poor fen		13.173	Dise (1992)
Minnesota	Open bog		3.074	Dise (1992)
Minnesota	Poor fen, control		66.075	Dise and Verry (2001)
Minnesota	Poor fen, ammonium nitrate added		70.255	Dise and Verry (2001)
Minnesota	Poor fen, ammonium sulfate added		44.788	Dise and Verry (2001)
Minnesota	Nonforested		17.250	Dise and Verry (2001)
Wales	Peat monoliths		63.230	Freeman et al. (1993)
New Hampshire	Poor fen		51.975	Frolking and Crill (1994)
West Siberia	Sedge fen		14.490	Golovatskaya and Dyukarev (2008)
Florida	Wet prairie (marl)		5.625	Happell et al. (1994)
Florida	Marsh (marl)		6.131	Happell et al. (1994)
Florida	Marsh (marl)		10.125	Happell et al. (1994)
Florida	Marsh (peat)		9.281	Happell et al. (1994)
Florida	Marsh (peat)		2.644	Happell et al. (1994)
Florida	Marsh (peat)		33.525	Happell et al. (1994)

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Table 13B.9. Nonforested Peatland Area Density Flux Factors[a]

Location	Vegetation Type	Annual Flux (CO_2 g C per m^2 per Year)	Annual Flux (CH_4 g C per m^2 per Year)	Reference
Florida	Marsh (peat)		4.163	Happell et al. (1994)
Quebec, Canada	Fen		6.225	Helbig et al. (2017)
Florida	Marsh	–44.9		Jimenez et al. (2012)
California	Young restored wetland	–368.0	53.000	Knox et al. (2015)
California	Old restored wetland	–397.0	38.700	Knox et al. (2015)
Washington	Bog		19.950	Lansdown et al. (1992)
Ontario, Canada	Fen		18.825	Lai et al. (2014)
Ontario, Canada	Fen		3.960	Lai et al. (2014)
Ontario, Canada	Fen		10.478	Lai et al. (2014)
Quebec, Canada	Bog	–60.78		Lu et al. (2017); Sulman et al. (2012); Lund et al. (2010)
Ireland	Bog	–47.78		Lu et al. (2017); Koehler et al. (2011)
Sweden	Fen	–58.0		Lu et al. (2017); Pleichel et al. (2014)
Finland	Natural fen		15.324	Nykänen et al. (1995)
Finland	Drained fen		0.132	Nykänen et al. (1995)
Minnesota	Fen	–35.3	16.300	Olsen et al. (2013)
Michigan	Bog		52.650	Shannon and White (1994)
Michigan	Bog		7.650	Shannon and White (1994)
Ontario, Canada	Marsh	–224.0	127.000	Strachan et al. (2015)
Quebec, Canada	Poor fen, control		0.032	Strack and Waddington (2007)
Quebec, Canada	Poor fen, control		39.080	Strack et al. (2004)
Quebec, Canada	Poor fen, with water table drawdown		17.564	Strack et al. (2004)
Northern Sweden	Ombrotrophic bog, hummocks		0.220	Svensson and Rosswall (1984)
Northern Sweden	Ombrotrophic bog, between hummocks		0.615	Svensson and Rosswall (1984)
Northern Sweden	Ombrotrophic bog, shallow depressions		3.381	Svensson and Rosswall (1984)
Northern Sweden	Ombrotrophic bog, deeper depressions		5.313	Svensson and Rosswall (1984)
Northern Sweden	Ombrominerotrophic		11.987	Svensson and Rosswall (1984)
Northern Sweden	Minerotrophic fen		74.163	Svensson and Rosswall (1984)
Western Canada	Bog		1.756	Turetsky et al. (2007)
North America and Europe	Bogs and fens		26.000	Turetsky et al. (2014)

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Table 13B.9. Nonforested Peatland Area Density Flux Factors[a]

Location	Vegetation Type	Annual Flux (CO_2 g C per m^2 per Year)	Annual Flux (CH_4 g C per m^2 per Year)	Reference
Minnesota	Bog		0.036	Updegraff et al. (2001)
Florida	Swamp		19.455	Villa and Mitsch (2014)
Northern England	Acidic blanket peat		0.025	Ward et al. (2007)
Maryland	*Sphagnum* bog		–0.300	Yavitt et al. (1990)
West Virginia	*Sphagnum*/ *Eriophorum* (poor fen)		1.800	Yavitt et al. (1990)
West Virginia	*Sphagnum*/Shrub (fen)		0	Yavitt et al. (1993)
West Virginia	*Polytrichum*/Shrub (fen)		0	Yavitt et al. (1993)
New York	*Typha* marsh		17.775	Yavitt (1997)
West Virginia	*Eriophorum*		14.250	Yavitt et al. (1993)
West Virginia	*Polytrichum*		11.250	Yavitt et al. (1993)
West Virginia	Shrub		1.200	Yavitt et al. (1993)
Alaska	Fen		53.66	Gorham (1991); after Crill et al. (1988)
Ontario, Canada	Mesocosms		0.510	Blodau and Moore (2003)
Quebec, Canada	Gatineau Park		0.020	Buttler et al. (1994)
Alaska	Waterlogged tundra		32.493	Derived from Sebacher et al. (1986)
Alaska	Wet meadows		10.977	Derived from Sebacher et al. (1986)
Alaska	Alpine fen		79.037	Derived from Sebacher et al. (1986)
Florida	Freshwater marsh	106.0		Malone et al. (2014)
Canada	Hummock	–39.814		Waddington et al. (1998)
Canada	Moss sedge	–148.308		Waddington et al. (1998)
Canada	Hollow	–153.285		Waddington et al. (1998)
Canada	Deep hollow	–5.972		Waddington et al. (1998)
Colorado	Fen		40.700	Chimner and Cooper (2003)

Notes

a) Key: CO_2, carbon dioxide; g, gram; C, carbon; CH_4, methane.

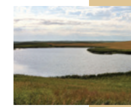

Table 13B.10. Mineral Soil Forest Area Density Flux Factors for Methane[a]

Vegetation (Species/Community)	Climate Zone	Location	Annual Flux CH_4 (g C per m^2 per Year)	Reference
Temperate	Temperate	Georgia	17.25	Pulliam (1993)
Dwarf cypress	Subtropical	Florida	2.025	Bartlett et al. (1989)
Swamp forest	Subtropical	Florida	18.825	Bartlett et al. (1989)
Hardwood hammock	Subtropical	Florida	0.000	Bartlett et al. (1989)
Cypress swamp, flowing water	Subtropical	Florida	18.300	Harriss and Sebacher (1981)
Cypress swamp, deep water	Subtropical	Georgia	25.200	Harriss and Sebacher (1981)
Cypress swamp, floodplain	Subtropical	South Carolina	2.700	Harriss and Sebacher (1981)
Maple/Gum forested swamp	Temperate	Virginia	0.375	Harriss et al. (1982)
Wetland forest	Temperate	Florida	16.125	Harriss et al. (1988)
Swamp forests	Temperate	Louisiana	39.825	Alford et al. (1997)
Pools forested swamp	Temperate	New York	51.750	Miller and Ghiors (1999)
Open water swamp	Subtropical	Florida	131.025	Schipper and Reddy (1994)
Waterlily slough	Subtropical	Florida	24.825	Schipper and Reddy (1994)
Lowland shrub and forested wetland	Temperate	Wisconsin	9.300	Werner et al. (2003)
Oak swamp (bank site)	Temperate	Virginia	31.950	Wilson et al. (1989)
Ash tree swamp	Temperate	Virginia	41.475	Wilson et al. (1989)

Notes

a) Key: CH_4, methane; g, gram; C, carbon.

Table 13B.11. Mineral Soil Nonforested Area Density Flux Factors[a]

Climate Zone	Location	NEE Emission (g CO_2-C per m^2 per Year)	CH_4 Emission (g CH_4-C per m^2 per Year)	Reference
Temperate	Prairie Pothole Region, Canada		4.900	Badiou et al. (2011)
Tropical	Global		41.900	Bartlett and Harriss (1993)
Temperate	Global		32.800	Bartlett and Harriss (1993)
Temperate	Ottawa, Ontario, Canada	–264.0		Bonneville et al. (2008)
Temperate	Ohio	65.4	37.650	Chu et al. (2015)
Temperate	Sanjiang Plain, China		35.100	Ding and Cai (2007)
Temperate	North Dakota		10.650	Gleason et al. (2009)
Temperate	North Florida		23.700	Happell et al. (1994)
Temperate	North Florida		7.500	Happell et al. (1994)
Tropical	South Florida		16.875	Harriss et al. (1988)
Temperate	Denmark		8.250	Herbst et al. (2011)
Tropical	Louisiana		35.100	Holm et al. (2016)
Temperate	Sanjiang Plain, China		22.500	Huang et al. (2010)
Temperate	Sanjiang Plain, China		16.875	Huang et al. (2010)
Tropical	Everglades, Florida	–44.9		Jimenez et al. (2012)
Temperate	Nebraska		60.000	Kim et al. (1999)
Temperate	Nebraska		48.000	Kim et al. (1999)
Temperate	Louisiana	–289.9	35.325	Krauss et al. (2016)
Tropical	Southwest Florida		0.600	Li and Mitsch (2016)
Tropical	Southwest Florida		92.925	Li and Mitsch (2016)
Tropical	Everglades, Florida	–40.24		Malone et al. (2014)
Temperate	North Carolina		0.525	Morse et al. (2012)
Temperate	Ohio		56.850	Nahlik and Mitsch (2010)
Temperate	Minnesota		8.775	Naiman et al. (1991)
Temperate	Minnesota		10.800	Naiman et al. (1991)
Temperate	Colorado		30.525	Neff et al. (1994)
Temperate	Virginia		54.113	Neubauer et al. (2000)
Temperate	Saskatchewan, Canada		24.100	Pennock et al. (2010)
Temperate	Saskatchewan, Canada		26.175	Pennock et al. (2010)
Temperate	Saskatchewan, Canada		18.075	Pennock et al. (2010)
Boreal	Saskatchewan, Canada		10.875	Rask et al. (2002)
Tropical	Everglades, Florida	–49.9		Schedlbauer et al. (2010)
Temperate	Georgia	92.4		Segarra et al. (2013)
Temperate	Minnesota		14.600	Shurpali and Verma (1998)
Temperate	Colorado		7.725	Smith and Lewis (1992)
Temperate	Sanjiang Plain, China		21.675	Song et al. (2003)
Temperate	Sanjiang Plain, China		32.550	Song et al. (2003)

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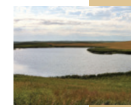

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Table 13B.11. Mineral Soil Nonforested Area Density Flux Factors[a]

Climate Zone	Location	NEE Emission (g CO_2-C per m^2 per Year)	CH_4 Emission (g CH_4-C per m^2 per Year)	Reference
Temperate	Sanjiang Plain, China		4.350	Song et al. (2009)
Temperate	Sanjiang Plain, China		0.225	Song et al. (2009)
Temperate	Ottawa, Ontario, Canada	–223.8	127.000	Strachan et al. (2015)
Tropical	Everglades, Florida		39.975	Villa et al. (2014)
Temperate	Colorado		31.275	Wickland et al. (1999)
Temperate	Colorado		23.456	Wickland et al. (1999)
Temperate	Virginia		31.725	Wilson et al. (1989)
Temperate	Virginia		16.988	Wilson et al. (1989)
Temperate	Three Gorges Reservoir, China		0.975	Yang et al. (2012)
Temperate	New York		93.975	Yavitt et al. (1997)
Temperate	New York		13.331	Yavitt et al. (1997)
Temperate	New York		41.906	Yavitt et al. (1997)
Temperate	Maryland and West Virginia		0.281	Yavitt et al. (1990)
Temperate	New York		10.688	Yavitt et al. (1993)
Temperate	New York		8.438	Yavitt et al. (1993)
Temperate	New York		0.900	Yavitt et al. (1993)
Temperate	Czech Republic	–126.3		Lu et al. (2017); Marek et al. (2011)
Boreal	Quebec, Canada	–264.0		Lu et al. (2017); Bonneville et al. (2008)
Boreal	Finland	–37.0		Lu et al. (2017); Lund et al. (2010)
Temperate	China	–61.67		Lu et al. (2017); Yu et al. (2013)
Temperate	Wisconsin	–83.99		Lu et al. (2017); Sulman et al. (2009)

Notes

a) Key: NEE, net ecosystem exchange; CO_2, carbon dioxide; CH_4, methane; g, gram; C, carbon.