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The Importance of Wetland Carbon Dynamics to Society: Insight from the Second State of the Carbon Cycle Science Report

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ABSTRACT

The Second State of the Carbon Cycle Report (SOCCR2) culminated in 19 chapters that spanned all North American sectors – from Energy Systems to Agriculture and Land Use – known to be important for understanding carbon (C) cycling and accounting. Wetlands, both inland and coastal, were found to be significant components of C fluxes along the terrestrial to aquatic hydrologic continuum. In this chapter, we synthesize the role of wetlands in the overall C footprint of North America (from Canada to Mexico) as one metric of the societal values placed on these terrestrial-aquatic interfaces. We also summarize the effects of management activities and climate change on the wetland C cycle and give some perspectives on the current and future importance of wetlands to society.

24.1. INTRODUCTION

24.1.1. Why Wetlands and Their Carbon Balance are Important to Society: We Have Come a Long Way

While North America was being settled in the 1800s and early 1900s, society viewed wetlands as unproductive areas that were breeding grounds for diseases and impediments to transportation and development. As a result, for about the first 150 years after settlement, wetlands were drained for agriculture and urban development to make these areas more productive for society. Drained wetlands are generally very productive because of the high concentration of organic matter and nutrients in the soil. By the mid-1900s, we were beginning to see the effects of wetland drainage on both inherent wetland functions and on the larger landscape. We know now that wetlands provide many ecosystem services for society, including critical habitat for many rare species, water storage for flood prevention, filtration of nutrients, pollutants, and sediment, and C sequestration and storage (Box 24.1). Presently, we have approximately 47% of our historical wetland acreage in the conterminous US, with much of that lost in the Midwest (~85%) due to wetland drainage and conversion to agriculture (Dahl & Stedman, 2013). Current relatively small losses in the US are a result of vegetation clearing, drainage, and compaction from roads and parking lots (USEPA, 2016), while losses in Canada are mainly from land conversion to agriculture or urban environments or flooding due to hydroelectric power (Federal Provincial and Territorial Governments of Canada, 2010). In Mexico, current losses are also a result of agriculture either by draining for crop production or flooding for aquaculture (De Gortari-Ludlow et al., 2015). Our more recent understanding of the societal values of the ecosystem services that wetlands provide has led to a number of important polices to conserve and restore wetlands. For example, the US instituted a policy of “no net loss” of wetlands in 1989. This policy

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Box 24.1 Relevance of the wetland carbon cycle to the provision of ecosystem services.

Carbon Cycle Process / Soil Condition	Ecosystem Services	Explanation
Carbon sequestration	Forest and agricultural products Wildlife habitat Shoreline stabilization	Growth and development of wetland vegetation communities.
Anaerobic soils	Water quality GHG emissions	Microbial mediated processes provide for removal of pollutants, and they also mediate the greenhouse gas emissions from the wetland.
Dissolved organic carbon export	Fisheries Waterfowl	Organic matter developed in the wetland is integral to the downstream food chain supporting waterfowl and fisheries.
Carbon storage	Reduction in atmospheric carbon dioxide	A higher proportion of the carbon accumulated in wetlands is sequestered compared to upland ecosystems
Soil Accretion	Storm surge attenuation Sea-level rise mitigation	Soil elevation gain due to organic matter accumulation increases shoreline resiliency.

has led to wetland banking programs by which if wetland area is lost, a comparable or greater area of functional wetland needs to be restored. Similarly, in Canada the Federal Policy on Wetland Conservation (Canadian Wildlife Service, 1991) encourages no net loss of wetlands. Also, in 2014, the Natural Protected Areas Commission of Mexico declared a national policy designed to protect wetlands and lessen losses. Moreover, international polices such as migratory bird agreements, in particular for waterfowl, between the US, Mexico, and Canada include wetland protection policy (North American Waterfowl Management Plan, 2012).

Carbon is the currency of many ecosystem services and the wetland carbon cycle is inextricably linked to the global carbon cycle and global change. Wetlands, especially terrestrial wetlands, are disproportionately important as C sinks in North America, and the world. For example, globally peatlands (organic soil wetlands) only occupy about 3% of the terrestrial area of the planet but contain 30% of soil C pools (Gorham, 1991). When comparing rates of C sequestration per unit area in North America, terrestrial wetlands are 11 times more efficient than grasslands and approximately 125 times more efficient than forests. Tidal wetlands are also efficient sinks for C with high rates of C sequestration. For wetlands to persist as C sinks mitigating C sources from the burning of fossil fuels, we need to continue to preserve and restore wetland functions and better understand how the wetland C sink will be influenced by climate change.

The Second State of the Carbon Cycle Report (SOCCR2) assessed the C footprint of North America, specifically the United States, Canada, and Mexico (USGCRP, 2018). The North American C budget is the net balance between the release (i.e., source) or storage (i.e., sink) of C by all North American sectors. Although there are a number of greenhouse gases (GHGs), carbon dioxide (CO_2) and methane (CH_4) are the important GHGs related to the C balance, with CH_4 having approximately 32 times the global warming potential of CO_2 over 100 years (Neubauer & Megonigal, 2016). Here we summarize the tidal and inland wetland findings from SOCCR2, discuss the effect of management and restoration on wetland C cycles, and provide some example studies assessing the effect of climate change on wetlands. We end by providing some perspective on the hurdles we still face in understanding wetland C cycles and the knowledge gaps that still exist that would help us better understand the role of wetlands in future global C cycles and feedbacks on climate change.

24.2. SUMMARY OF FINDINGS FROM SOCCR2

Wetlands in North America cover a wide range of locations with varied hydrology, climate, soils, and vegetation (Fig. 24.1). While current wetland distribution is small in area (< 2% of land cover), they are spatially extensive, ranging from inland (or terrestrial) wetlands (e.g., bogs, fens, swamps, pocosins, Carolina bays, playas, riparian wetlands, etc.) to tidal (or coastal) wetlands (e.g., salt marshes, tidal marshes, freshwater tidal marsh, freshwater tidal forest, mangroves, etc.). Terrestrial wetlands in North America represent 37% of global wetland area, with an estimated 2.2 million km^2 (Lehner & Döll, 2004). The soil plus vegetation C pool of North American ter-

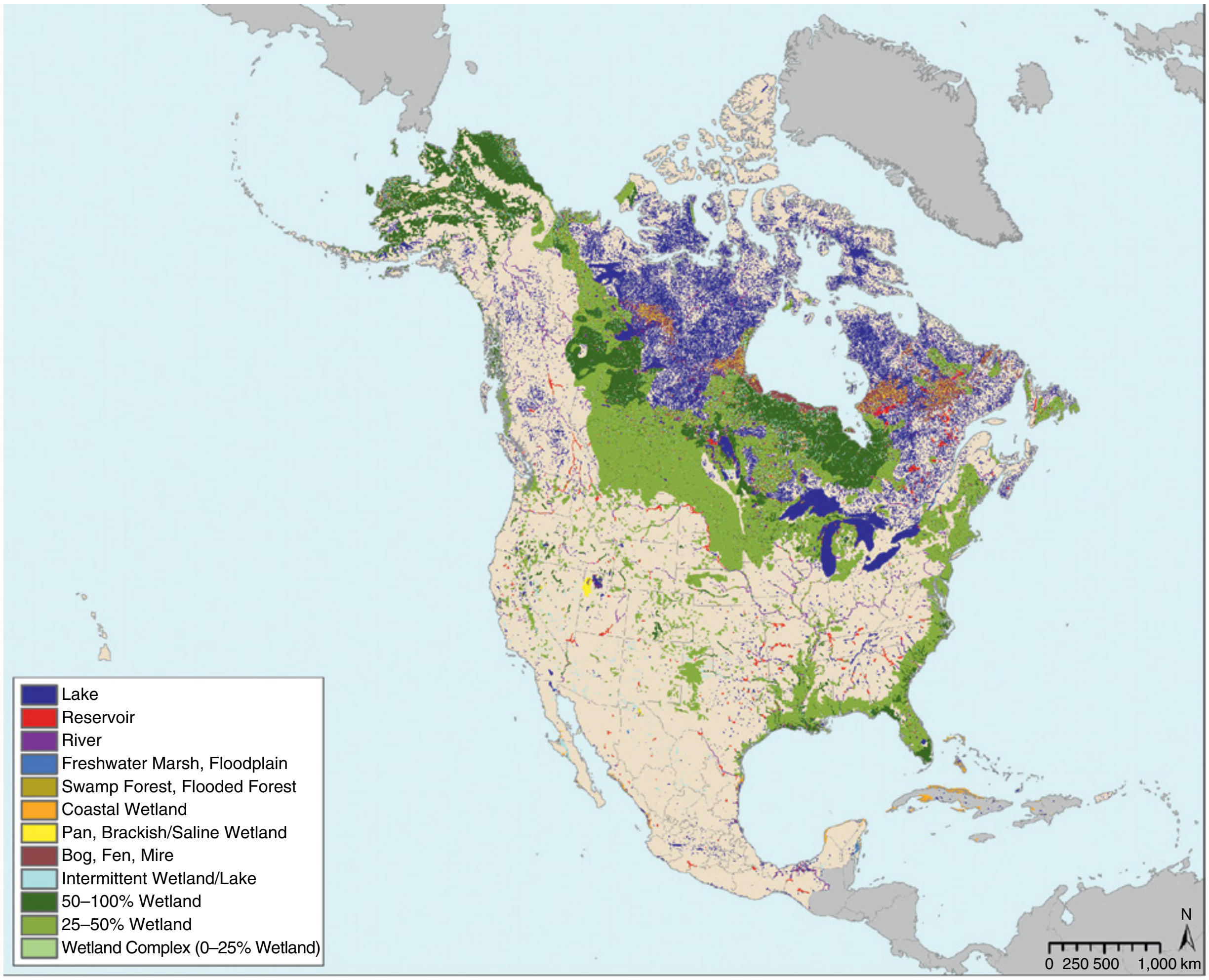

Figure 24.1 Map of North American wetlands (*Source*: Lehner & Döll, 2004). The 0–25% wetland category represents a mix of different wetland types where the authors could not identify a clear spatial coverage.

restrial wetlands is about 161 Pg of C, similarly representing roughly 36% of the global wetland C pool (Kolka et al., 2018). At about 55% of the total terrestrial wetland area, forested wetlands are more abundant in North America than non-forested wetlands, with most forested wetlands occurring in Canada. Also of importance, North American peatlands (organic soil wetlands), regardless of vegetation type, occupy approximately 58% of the total terrestrial wetland area but contain 80% of the overall C pool. From a GHG flux perspective, North American terrestrial wetlands are also disproportionately important as an annual sink of CO_2 of about 123 Tg C as CO_2/yr, with much of that occurring in forested wetlands (~53%) (Fig. 24.2). Although a large sink for CO_2, terrestrial wetlands are also the largest natural source of CH_4 in North America, with mineral soil wetlands emitting 56% of the total of 45 Tg C as CH_4/yr (Kolka et al., 2018). Within the conterminous US mineral soil wetlands constitute 79% of the terrestrial wetland area, but account for only 39% of the ecosystem C pool (Table 24.1); this is because peatlands have a much greater carbon density than mineral soil terrestrial wetlands.

Tidal wetlands are broadly distributed throughout the North American coastline, but these wetlands differ with respect to their ontology, hydrology, and biogeochemistry compared to terrestrial wetlands. These wetlands include both freshwater and saltwater systems, and are collectively recognized as blue carbon, reflecting C storage and dynamics in tidally mediated wetlands. In the conterminous US, they represent a far smaller area (38,609 km^2) and C pool (1,100 Tg to 1 m depth) (Table 24.1) but sustain some of the highest rates of annual atmospheric CO_2 uptake (27 ± 13 Tg/yr), while emitting very little CH_4 (~0.7 Tg/yr) (Fig. 24.3). Mean tidal wetland soil C pools (27.0 ± 13.1 kg C to 1 m depth) are ~100-fold greater than their mean vegetation pools (0.21 ± 0.2 kg C/m^2), and are

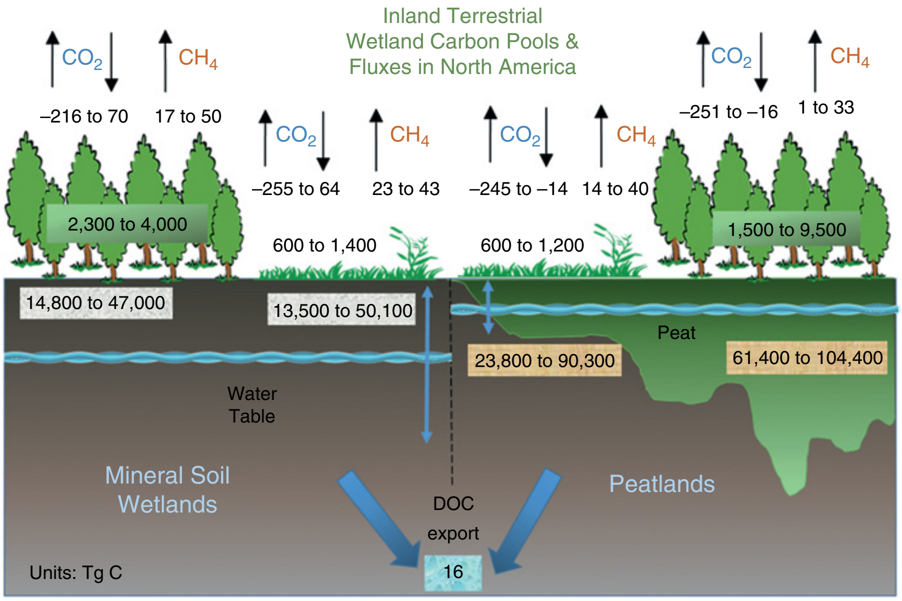

Figure 24.2 Terrestrial wetland carbon pool and CO_2 and CH_4 fluxes in North America (Source: Kolka et al., 2018).

Table 24.1 Ecosystem carbon pool (to a depth of 1 m) and net CO_2 and CH_4 flux in terrestrial and coastal wetlands in the conterminous US

	Area	Ecosystem Carbon Pool	Carbon Flux	
Wetland Type	(km^2)	(Pg C)	CO_2 (Gg C/yr)	CH_4-C (Gg C/yr)
Terrestrial Wetlands – Mineral Soils				
Forested	173,091	3.3	–11.6 ± 8.2	4.7 ± 2.8
Non-forested	138,381	1.9	–14.1 ± 9.5	3.6 ± 1.0
Terrestrial Wetlands – Organic Soils				
Forested	40,823	4.4	–4.9 ± 3.8	0.4 ± 0.4
Non-forested	42,903	3.9	–5.8 ± 3.6	1.0 ± 0.3
Tidal Wetlands				
Freshwater Marsh	2,234	0.06	na	na
Freshwater Forested	3,257	0.09	na	na
Brackish – Saline Marsh	19,809	0.53	na	na
Mangrove	13,309	0.42	na	na

(*Sources*: Kolka et al., 2018; Windham-Myers et al., 2018).

not well predicted by primary productivity, vegetation type, salinity class (fresh or saline waters) or climate zones, either by latitude or region (Byrd et al., 2018; Holmquist et al., 2018; Windham-Myers et al., 2018). Approximately 51% of the coastal wetlands are brackish saline salt marshes, contributing 48% of the ecosystem C pool (Table 24.1). The C density is highest in mangrove ecosystems, with little difference between the salt marshes and tidal freshwater wetlands. Because more than 40% of human populations are located within 100 km of the coast (Kummu et al., 2016), these tidally maintained soil C pools are particularly vulnerable to human influences, in addition to ocean-driven erosional forces.

Historical wetland loss across inland and coastal wetlands was unprecedented following European settlement, but current wetland losses are much lower with creation and restoration of wetlands nearly offsetting those wetlands lost as a result of development and agriculture (Hanson, 2006). Even though the overall wetland area of North America is not changing considerably, extant wetlands appear to be losing important functions, especially those restored or otherwise disturbed but still meeting the criteria of a wetland (e.g., C storage), (Nahlik & Fennessy, 2016). Also, there is little information on how restoration, creation, or disturbance affect the long-term storage of soil C and the associated balance between emissions of CO_2 and CH_4. Understanding those relationships would be extremely helpful for modeling terrestrial GHG emissions and C storage across anthropogenic disturbance gradients.

24.2.1. Wetland Carbon Cycling at a Landscape Scale

Terrestrial wetlands play a critical role in the overall land sector C balance, accounting for a net sink of atmospheric C of approximately 82 Tg C/yr (~ 126 Tg C sink of CO_2, 44 Tg C source of CH_4). At a continental scale, that compares to other ecosystem sectors that are sinks of C including Forests (217 Tg C/yr of CO_2), Agriculture (15 Tg C/yr of CO_2), and Grasslands (25 Tg C/yr of CO_2). In the case of the Forest sector, 124 Tg C/yr of CO_2 of the 217 Tg C/yr of CO_2 is offset annually from emissions attributed to wood products (difference is 93 Tg C/yr of CO_2 for the Forest sink) which overall is comparable to the terrestrial Wetland sector sink even though wetlands only occupy approximately 0.3% of the area of forests. Tidal Wetlands and Estuaries are also C sinks of ~ 17 Tg C/yr of CO_2, and Permafrost and Arctic Areas of ~ 14 Tg C/yr of CO_2. All the ecosystem sinks, including near shore oceans (160 Tg C/yr of CO_2) account for 766 Tg C/yr of CO_2 which is more than offset by the burning of fossil fuels annually (1,774 Tg C/yr of CO_2), indicating that North America is still a large source of C to the global budget (USGCRP, 2018).

Within the aquatic continuum (from terrestrial wetlands to the coastal ocean), wetlands serve as significant sources of C transported as lateral fluxes between landscape components, especially in dissolved forms, such as DOC (dissolved organic carbon) and DIC (dissolved inorganic carbon) (Fig. 24.4). From terrestrial wetlands, the DOC flux is usually the largest of the two fluxes and is a source of C, and associated nutrients and pollutants, to both surface waters and groundwater. Incorporating C

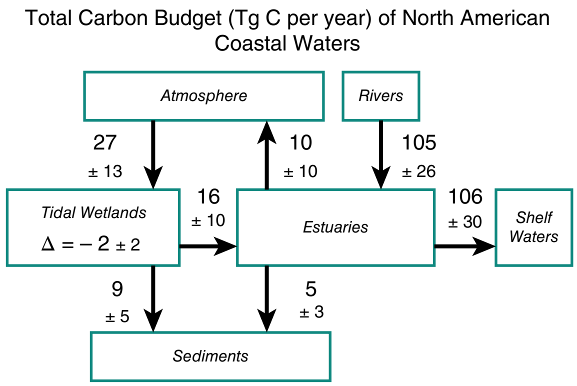

Figure 24.3 Total carbon budget of North American coastal waters including ± 2 standard errors (Source: Windham-Myers et al., 2018).

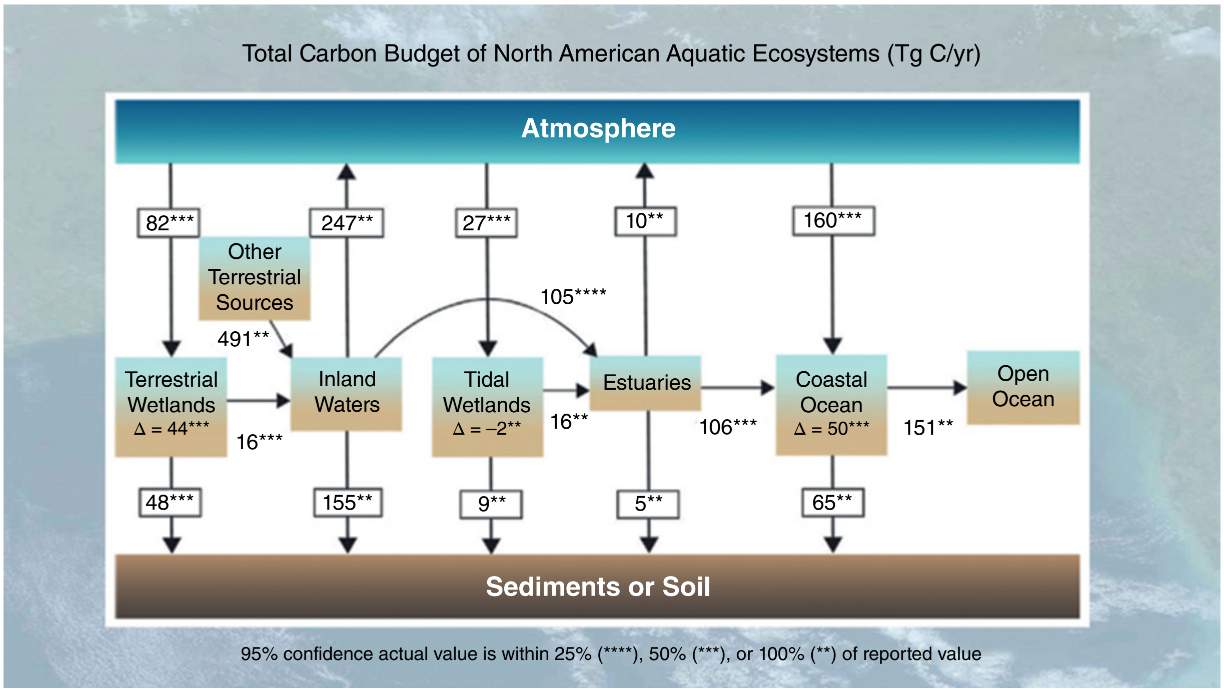

Figure 24.4 Total carbon budget of the terrestrial to aquatic continuum in North America. Fig. 24.2 represents the Terrestrial Wetlands component of Fig. 24.4 and Fig. 24.3 represents the Tidal Wetlands component of Fig. 24.4 (compilation from SOCCR2 chapters by Butman et al., 2018; Fennel et al., 2018; Kolka et al., 2018; Windham-Myers et al., 2018).

into soils and sediment (i.e., burial) along the aquatic continuum is also an important process when considering the overall balance from wetlands to inland waters through estuaries and into the ocean (Fig. 24.4). For tidal wetlands, DIC is the dominant export to estuaries and coastal waters. For North America, terrestrial wetlands are a sink of 82 Tg C/yr of CO_2 with approximately 48 Tg C/yr of CO_2 of that C buried and 16 Tg C/yr of CO_2 transported downstream to inland waters. Inland waters have considerable upland inputs (491 Tg C/yr of CO_2) in addition to wetland inputs, emitting 247 Tg C/yr of CO_2 leading to a burial of 155 Tg C/yr of CO_2, and contributing 105 Tg C/yr of CO_2 to coastal estuaries. Tidal wetlands are also connected to coastal estuaries providing 16 Tg C/yr of CO_2, following the uptake of 27 Tg C/yr of CO_2 and burial of 9 Tg C/yr of CO_2 with a net change of 2 Tg C/yr of CO_2. Aquatic components (subtidal, open water) of estuaries are a small source of carbon to the atmosphere (10 Tg C/yr of CO_2) and have small burial rates (5 Tg C/yr of CO_2) but are a large source of lateral C transport to the coastal ocean (106 Tg C/yr of CO_2). Coastal oceans are large sinks for C from the atmosphere (160 Tg C/yr of CO_2) and have high burial rates (65 Tg C/yr of CO_2) but are also large sources of C (151 Tg C/yr of CO_2) to the open oceans (Fig. 24.4).

24.3. MANAGED WETLANDS AND THE CARBON CYCLE

Wetlands may be managed for one or more of their inherent ecosystem services, as well as for the provision of new services, for example crop production. Management actions that change the hydrology, soils, or vegetation will affect C dynamics, and can often lead to enhanced decomposition, decreased CH_4 flux, and less C sequestration, particularly when wetlands are drained. In contrast, restoration of drained wetlands may increase C sequestration and the production of GHGs, as is observed in impounded tidal wetlands. Here we synthesize the principal interactions of management activities, which encompass direct uses including agriculture, forestry, urbanization, and restoration, on the wetland C cycle, emphasizing the relevance to societal values.

24.3.1. Agriculture

Once drained, wetland soils are often well poised to support crop production because of their high C and nutrient content, and thus agriculture has been the cause of the majority of the wetland losses in the US. The soil pool of those converted wetlands is reduced significantly as a result of the drainage and cropping systems (Bedard-Haughn et al., 2006; Bridgham et al., 2006; Euliss et al., 2006; Gleason et al., 2008, 2009; Tangen et al., 2015). Here we consider the effects of agricultural use on the C balance of intact wetlands that are either used for cropping or grazing, or those that are inextricably linked to agricultural activities adjoining or surrounding them.

Wetlands within or adjacent to agricultural fields receive sediment, nutrients, and agrichemicals through wind and water erosion. Runoff from adjacent crop lands tends to be greater than grass lands (Euliss & Mushet, 1996; van der Kamp et al., 1999, 2003). Sedimentation into the wetland is a primary effect of the agricultural runoff, which may result in the burial of wetland surface soils (Martin & Hartman, 1987; Tangen & Gleason, 2008). Nutrient-enriched runoff from agricultural lands may also increase GHG emissions in receiving wetlands (Hefting et al., 2003; Helton et al., 2014). With respect to cropping within the wetland, production of rice is a significant source of atmospheric CH_4 (Minarni, 1994; Shearer & Khalil, 2000) due to extended periods of inundation and the application of fertilizers. Similarly, sugar cane production may reduce soil C pools due the use of drainage systems (Baker et al., 2007).

Livestock grazing can directly impact wetlands through biomass removal, soil compaction, erosion, introduction of exotic species, and deposition of waste (Morris & Reich, 2013). However, whether grazing increases or decreases soil C and nitrogen (N) varies considerably, and in most cases depends on the location of the study and the grazing management system (Li et al., 2011). For example, in coastal salt marshes, livestock grazing reduces plant biomass, but reported effects on soil C differ with no effect reported for Europe and reductions reported in North America (Davidson et al., 2017). Studying the effects of grazing and burning on a northern peatland, Ward et al. (2017) indicated reductions in aboveground biomass for both treatments, but the soil C content to a depth of 1 m was not affected.

24.3.2. Forest Management

Forested wetlands may be managed for a variety of commercial and non-commercial uses. Those management practices may affect wetland functions and the C cycle as a result of harvesting, roads, site preparation, and minor drainage. Most of the disturbances associated with silvicultural practices occur infrequently (e.g., once in a rotation, 30–100+ years), and effects on wetland functions are commonly either benign or ephemeral. Changes in the hydrologic regime and vegetation composition of the managed forest are the two principal factors affecting the C pools and turnover in managed wetlands.

Harvesting results in the removal of the forest overstory vegetation, hence a direct and immediate reduction in the aboveground C pool. However, that change is temporary, with recovery rates depending on the inherent productivity of the site and management practices. A comparison of logging practices in bottomland hardwoods did not show a deterioration of hydric soil processes (Aust et al., 2006; McKee et al., 2012). In another bottomland hardwood study, the regenerating forest exhibited higher levels of productivity than the mature stand (Lockaby et al., 1997).

Site preparation involves practices to facilitate regeneration of stands. In forested wetlands, site preparation may include treatments that reduce wetness in the upper soil such as bedding or mounding. In both those treatments, an elevated planting site is created either by the formation of a continuous bed using a plow, or a mound using an excavator. The objective of the prescription is to increase the aerated soil volume, which improves seedling survival, but it also results in an increase in organic matter decomposition and nutrient cycling (Grigal & Vance, 2000; Trettin et al., 1996). However, studies in the cold-temperate zone of Michigan (Trettin et al., 2011) and subtropical coastal plain of South Carolina (Neaves et al., 2017a) have shown the disturbance effect on soil C to be relatively short (e.g., < 20 yr). A factor contributing to the recovery of the soil C pool is that tree productivity tends to be higher on the bedded sites (Neaves et al., 2017b).

Minor drainage may also be used as a means to mitigate soil disturbance during logging operations and to improve seedling survival and tree growth. However, a stipulation for use of minor drainage for silvicultural operations in the US is that it should not compromise the jurisdictional status of the wetland; hence, wetland hydrology should remain intact, albeit modified. As the hydrologic controls on C cycling in forested wetlands are well established (Trettin & Jurgensen, 2003), changes in mean water table depth and periods of high-water table or inundation do have an effect on C pools, as well as GHG emissions. The lowering of the water table typically results in an increase in organic matter decomposition and a reduction in the soil C pool (Laiho, 2006), especially if drainage is improperly installed or maintained. Correspondingly, CO_2 emissions increase and CH_4 emissions decline (Couwenberg et al., 2009; Moore & Knowles, 1989; Nykänen et al., 1998). Seedling survival and early stand growth may be improved through the use of minor drainage (Fox et al., 2004; Skaggs et al., 2016), and enhanced productivity on boreal peatlands has been shown to increase the soil C pools (Minkkinen et al., 1999).

Forested coastal wetlands include mangroves (saline tidal forest) and swamps (tidal freshwater). Mangroves are confined to saltwater while tidally influenced swamps have forest communities similar to their non-tidal counterparts. There is a lack of information on the effects of harvesting of tidal forested freshwater wetlands; the area of actively managed land is small given the overall area of this wetland type. However, these lands are sensitive to salinity, which can change as a result of sea-level rise resulting in the conversion of forest to marsh (Krauss et al., 2018). In contrast, mangroves are expanding poleward due to climate warming, effectively displacing salt marshes (Saintilan et al., 2014).

24.3.3. Urbanization and Development Activities

Urbanization and development activities were the primary cause of wetland loss in the US during 1992–1997 (NRCS, 2001). The loss is typically a result of draining and filling the wetland such that the wetland conditions no longer prevail. As a result, the C-related functions cease, and their removal from the landscape adversely affects water quality, plant and animal diversity, and water storage (Faulkner, 2004; Lee et al., 2006). Wetlands within or adjoining urban areas provide important functions that are inextricably linked to the C dynamics within the wetland, which may be valued higher in the urban setting than in an undeveloped landscape (Ehrenfeld, 2000). Those wetlands may provide storage for runoff waters thereby reducing flooding, and they support biogeochemical processes that help to mitigate chemical contaminants (e.g., hydrocarbons, fertilizer, pesticides) and sediments in the runoff, thereby functioning to improve downstream water quality (Mitsch & Gosselink, 2015). Urban wetlands may also be degraded due to the volume or toxicity of chemical constituents entering the system.

The blue carbon of coastal wetlands is inextricably linked to other ecosystem services. Those wetlands are threatened by a range of factors from development to disturbances from coastal storms (Dahl & Stedman, 2013). Wetlands that are poised at elevations within the tidal frame may experience tidal restrictions both actively (dikes, levees) and passively (transportation corridor and culverts). Approximately 30% of saltwater wetlands are estimated as having an altered hydrologic regime, either drier (drained) or wetter (impounded), which influences biogeochemical processes and C dynamics (Kroeger et al., 2017).

24.3.4. Restoration

Wetland restoration is a means to re-establish or enhance wetland functions. It is fundamental to achieving the no net loss goal of the US, and it is widely used elsewhere as well. Adaptive management is the key to successful wetland restoration projects (Stelk et al., 2017). Wetland restoration practices typically focus on modifications of the site hydrology and re-establishment of hydrophytic vegetation, actions with direct implications to the wetland C cycle.

Restoring wetlands has been found to reverse the loss of soil C from drainage (Järveoja et al., 2016). For instance, restored mineral soil wetlands in central New York accumulated 0.74 Mg C/ha/yr over a 55-year period (Ballantine & Schneider, 2009). Guidelines from the Intergovernmental Panel on Climate Change for mineral soil wetlands state that cultivation leads to losses of up to 71% of the pre-cultivation soil organic C pool in the top 30 cm of soil over 20 years and that restoration increases post-cultivation depleted soil C pools by 80% over 20 years, and by 100% (i.e., relative to pre-cultivation pools) after 40 years (Wickland et al., 2014). An increase in soil C is also regularly measured after restoring organic soil wetlands (Lucchese et al., 2010). In contrast, restoration of drained wetlands may increase production of GHGs, as is observed in impounded tidal wetlands (Windham-Myers et al., 2018).

Wetland restoration typically lowers DOC export from wetlands (Strack & Zuback, 2013); however, there may be an initial flush after restoration activities. Rewetting or creating freshwater wetlands may increase CH_4 emissions (Badiou et al., 2011; Strack & Zuback, 2013), although some studies have found that restoration did not increase CH_4 emissions (Richards & Craft, 2015). Methane emissions appear to be especially high in restored wetlands located in agricultural settings or in deep water areas with emergent vegetation (Schrier-Uijl et al., 2014; Strack & Zuback, 2013). In the long term, the climate benefits of increasing soil C sequestration through restoring degraded wetlands appears to be a positive for GHG mitigation (Strack & Zuback, 2013), especially in saline tidal wetlands where the presence of sulfate in floodwater suppresses CH_4 production (Poffenbarger et al., 2011). However, it should be noted that most of the assessments of C dynamics in restored wetlands have been done within a few years following restoration (e.g., < 5 yrs), hence the rate of change of C sequestration in restored wetlands may be expected to change as the ecosystem matures (Anderson et al., 2016). At the landscape-scale in Florida, an increase in wetland area significantly enhanced soil C storage (Xiong et al., 2014), reflecting the inherently higher C density in wetlands as compared to uplands and/or drained areas.

Mapping these changes, whether it is degradation or restoration, is extremely difficult and yet required for C balance accounting. Dynamic maps, whether decadal or annual, are helpful to track the relative changes in hydrology and related C fluxes, such as net primary productivity.

24.4. CLIMATE CHANGE AND WETLAND CARBON DYNAMICS

Although wetlands provide many ecosystem services (Pindilli, 2021) the sequestration and storage of C from the atmosphere may be the most important as we consider mitigation strategies and adapt to climate change. There are considerable uncertainties in how wetlands will respond to changing climatic conditions as we note here; hence we share several case studies that provide examples of promising new research to reduce those uncertainties and thereby improve the basis for considering how other important societal ecosystem services may be affected (e.g., Box 24.1).

Climate change is affecting ecosystems across the globe, but maybe most notably in wetlands (Junk et al., 2013). From permafrost and boreal peatlands in the northern hemisphere to high-elevation peatlands in the Andes Mountains in South America to coastal wetlands that are threatened by rising sea level, wetland water balance and subsequent C balance are very susceptible to warming and changes in precipitation regimes (Moomaw et al., 2018). Future global warming will likely result in these changes being accelerated, especially in the northern hemisphere where warming is predicted to increase 2 ˚C to 7 ˚C by 2100 (IPCC, 2014). From the early 1960s to the present in northern Minnesota, temperatures have already risen by 2.5 ˚C with a growing season that is approximately 30 days longer than in 1960 (Kolka et al., 2011).

Although the implications of elevated CO_2 on wetland C pools is a bit ambiguous, generally, elevated CO_2 leads to enhanced emissions of CH_4 (Kirwin and Blum, 2011). However, warming studies have demonstrated that increasing temperatures will have important impacts that will affect humans and ecosystems globally. Warming generally leads to higher evapotranspiration, which lowers water tables and extends the aerobic zone. Because aerobic decomposition is more efficient than anerobic decomposition, warming directly leads to higher decomposition rates. Changing precipitation regimes will also have dramatic impacts on wetland water tables and subsequent decomposition rates.

Wetland hydrology is the principal attribute affected by climate change, primarily by changes in precipitation, including its periodicity, storm intensity, and recurrence of extreme events (Winter, 2000). Changes in evapotranspiration may interact with altered precipitation regimes, exacerbating the effects on wetland hydrology (Zhu et al., 2017). A reduction in the water table within the wetland will alter the biogeochemical process (e.g., C cycle), subsequently increasing organic matter decomposition, reducing CH_4 production, and altering the rate of C sequestration. Higher incidences of drought are especially problematic because of the potential change in plant communities where the C building blocks in many peatlands, *Sphagnum* moss, can be outcompeted by sedges and other plants that are not as productive (Hanson et al., 2020). The drying of peatlands increases the susceptibility to wildfire resulting large losses of soil C (Reddy et al., 2016: Turetsky et al., 2011).

24.4.1. Case Studies

There are several notable studies that have assessed some aspect of climate change on wetland ecosystems including SPRUCE, PEATcosm, the Seney Wildlife Refuge, APEX, Biotron, and SMARTX experiments. Maybe the most notable is the Spruce and Peatland Responses to Changing Environments (SPRUCE) experiment (Hanson et al., 2017). SPRUCE is a whole ecosystem warming by elevated CO_2 experiment in a northern bog in Minnesota, USA. SPRUCE has 10 chambers with 5 temperature treatments ranging from 0 ˚C to 9 ˚C with and without elevated CO_2 at 900 ppm, a little over twice current CO_2 concentrations in the atmosphere. The warming treatments are both aboveground and belowground and a continuous differential from the ambient conditions. Although early in the experiment, warming has lowered growing season water tables and led to much higher CO_2 and CH_4 fluxes (Gill et al., 2017; Hanson et al., 2020). The combination of warming and lower water tables have led to dramatic changes in the plant communities with the black spruce (*Picea mariana*) and eastern larch (*Larix laricina*) trees, *Sphagnum* species, and lichens (Smith et al., 2018) not faring well, but some ericaceous shrubs such as blueberry (*Vaccinium angustifolium*) and Labrador tea (*Rhododendrom groenlandicum*) thriving (McPartland et al., 2020). Plant phenology is also changing with green-up occurring earlier in the season, and plants remaining greener throughout the growing season and maintaining greenness later in the season (McPartland et al., 2019; Richardson et al., 2018). One result of earlier green-up, especially by the trees, is that they have become susceptible to false springs where the trees are cued to green-up when temperatures are relatively high but get severely frostbitten when colder temperatures occur again (Richardson et al., 2018).

PEATcosm is an experiment in Houghton, Michigan, USA, that manipulated undisturbed peat monoliths in 1 m^3 boxes. In its first iteration, PEATcosm simulated drought and changes in plant communities by lowering water tables with rain-out shelters and removing ericaceous shrubs or sedges. Results indicated that *Sphagnum* species had highest productivity in the high-water table treatments and vascular plants had the highest production in the low-water table treatments (Potvin et al., 2015). Peat accumulation rates decreased when ericaceous shrubs were removed (Potvin et al., 2015). Overall, the results from PEATcosm suggested that drought and changes in plant communities in peatlands will affect peat accumulation rates and the cycling of C.

As a result of levee construction in the 1930s, a long-term water table change occurred at the Seney National Wildlife Refuge in the Upper Peninsula of Michigan, USA. The levee construction led to higher peatland water tables on one side of the levee and lower water tables on the opposite side of the levee (Chimner et al., 2017). By comparing both higher and lower water table areas to a relatively undisturbed control peatland, Chimner et al. (2017) found that the undisturbed control had the greatest plant productivity followed by the drier site and then the wetter site. All sites were net CO_2 sinks annually with the greatest sink being the wet site and the dry site being the lowest sink. However, the wet site also had the highest CH_4 emissions (Chimner et al., 2017). The Seney lower water table treatments may be a good example of what the future might hold given warmer temperatures and increased potential for drought.

Somewhat similar to the Seney manipulation, the Alaskan Peatland Experiment (APEX) located on the Bonanza Creek Experimental Forest in central Alaska, USA, has lower (drained), higher (flooded), and control water table treatments in an open peatland. Also, growth chambers were positioned in each of the treatments which led to surface warming of about 1˚C. The combination of flooding and warming led to the highest CH_4 emissions with drainage and no warming leading to the lowest CH_4 emissions (Turetsky et al., 2008). For CO_2, the results are not as clear, as the lower water table treatment was a weak sink or small source while the control was a moderate sink, and the flooded treatment was a weak to strong sink (Chivers et al., 2009).

The Biotron experiment at Western University, Ontario, Canada, used mesocosms somewhat similar to PEATcosm and experimented with differences in water tables (high and low), temperature (+0–8˚C), and elevated CO_2 (ambient and 750 ppm). With increasing temperature and CO_2 and lower water tables the vegetation tended to shift away from *Sphagnum* and towards graminoid based communities, likely leading to lower C accumulation rates (Dieleman et al., 2015). Increasing temperatures also increased decomposition rates and led to higher concentrations of DOC in pore water, again likely indicating lower C accumulation rates (Dieleman et al., 2016). Overall, it is apparent that the combination of warming, elevated CO_2, and increasing variable water tables from more stochastic events as predicted from climate change will lead to wetlands becoming either lesser sinks for C or potentially becoming sources.

Tidal marshes are undergoing experimental research to assess global change influences as well. Following a long history of mesocosm experiments (Global Climate Research Wetland Project or GCREW) that tested the effects of elevated CO_2 (Erickson et al., 2007), and also the combination of elevated CO_2 and nitrogen fertilization (White et al., 2012) on tidal marshes, an important long-term study was established. SMARTX (Salt Marsh Accretion Response to Temperature eXperiment) was initiated in 2016 within the Global Change Research Experimental Wetlands at the Smithsonian Environmental Research Center in Edgewater, Maryland, USA. SMARTX is designed with active warming of the brackish tidal marsh soils and air (up to 5.1°C), as well as treatments with elevated atmospheric CO_2 (750 ppm). It is unique in being the first replicated terrestrial ecosystem experiment with both temperature and CO_2 enhancements. Early results suggest non-linear responses to warming in productivity and marsh elevation responses, and significant enhancement of CH_4 emissions due to longer growing seasons in the warmer plots (Megonigal et al., 2019).

24.4.2. Future Prediction of Net C Balance in Wetlands

Modeling of future conditions on wetlands shows variable results but in general wetlands are predicted to either become lesser sinks for C or even flip to become sources (Mitsch et al., 2013). Methane emissions are predicted to increase (Shindell et al., 2004) and the net sink of CO_2 is expected to decrease (Mitsch et al., 2013). The overall sink/source balance of C is a result of changes in water tables and plant communities (Walker et al., 2016). Simulations of precipitation vary widely but even if there are small changes in precipitation (positive or negative) it is anticipated that evapotranspiration will increase, leading to lower water tables and higher rates of aerobic decomposition and either lower rates of C accumulation or greater net C emissions. Mitigation strategies are two-fold. First, wetland preservation is key to maintaining the ecosystem services that wetlands provide society. Although predictions generally indicate lesser C sequestration, wetlands are still our largest C sink in global terrestrial systems. Second, restoring and creating wetlands will provide new opportunities for potential C sinks. Restoring or creating wetland functions that lead to higher plant production and/or slower decomposition rates will lead to lower C emissions and possibly C sequestration. Lowering of water tables and associated changes in plant communities could lead to higher C sequestration in some wetlands that currently have water tables at or near the surface (Walker et al., 2016). In temperate or boreal climates, lowering of water tables would likely result in a change from *Sphagnum*/graminoid-dominated ecosystems to *Sphagnum*/forested ecosystems with overall higher productivity and decomposition, but with plant productivity increasing relatively more than decomposition rates.

24.5. PERSPECTIVES

Wetlands generally have a higher soil C density than upland ecosystems, due primarily to limited decomposition of organic matter. However, soil C densities are often overestimated due to mapping limitations and methodological issues in measuring bulk density (Köchy et al., 2015). However, the soil C pool in wetlands is often larger than reported because most of the measurement data is obtained from within the upper meter of soil. Peatlands are known to have organic soil depths ranging from 2–10 m; unfortunately, there is very limited peat depth data globally, thereby precluding estimates of the total peatland C pool. Similarly, blue carbon soils (tidal freshwater forests and marshes, salt marshes, and mangroves) can extend 2–10 m below the surface (blue carbon ecosystems also include sea grass beds, which are not considered here); again, these C pools are not typically reported due to the lack of widespread inventory data and an emphasis on surface soils that are most susceptible to human and climate influence. It is not practical to routinely sample wetland soils to depths of 2–3 m or to probe peatlands or blue carbon to refusal. However, regional studies of specific wetland types to characterize the uncertainties in deep soil (> 1 m) C pools would greatly improve the accuracy of regional and continental scale inventories, and provide a much improved basis for considering the implications associated with climate change and management activities.

There are too few measurements of C fluxes in wetlands, in comparison with upland settings and in representing variability of wetland conditions. Accordingly, there is a high degree of uncertainty when aggregating measurements to characterize particular wetland types or wetlands within a specified geographic area. The inadequate database on C fluxes from wetlands is also an impediment to the development and testing of mechanistic models, as measurement data for model calibration and validation are fundamental to developing better tools for simulating the wetland C cycle and considering the effects associated with changing climatic conditions and management regimes. Many types of wetlands have a heterogeneous surface micro-topography – a mosaic of low areas (e.g., hollows or depressions), elevated areas (e.g., hummocks or mounds), and flat or sloping areas (e.g., lawns). This micro-topography affects biogeochemical processes principally because it defines the elevation relative to water table and the anoxic soil layers. For example, hummocks may exhibit very low CH_4 emissions or even oxidize atmospheric CH_4, while adjoining hollows emit CH_4. Accordingly, to scale soil gaseous emissions, measurements are needed from the various micro-topographic positions along with detailed information about the distribution of the micro-topography within the wetland. This information is generally lacking, thereby contributing to the uncertainties in estimates of emissions from wetlands. Estimates of emissions from wetlands derived using eddy covariance measurements effectively integrate the heterogeneity in the wetland surface conditions; however, that technique is data intensive and relatively expensive, and also does not support fine-scale analyses of specific processes and mechanisms at the micro-topography scale. While eddy covariance is data intensive and relatively expensive, the networks are expanding and being used to estimate larger scale ecosystem wetland productivity (Feagin et al., 2020) and contribute to regional and global studies on methane emissions (Knox et al., 2019).

Although wetlands are recognized as a major source of atmospheric CH_4 globally, there remain considerable uncertainties in estimating emissions. This is because controls on CH_4 dynamics are inadequately described in current models and there is an inadequate basis for scaling among the varied wetland conditions (Bridgham et al., 2013). Exasperating the complexity of CH_4 dynamics, recent advances in soil microbiology found that methanogenesis is not constrained to anaerobic soil layers (Angle et al., 2017), underscoring our incomplete understanding about the wetland C cycle and suggesting that we have underestimated the uncertainties in how wetlands may respond to perturbations and changes in ambient conditions.

Wetlands are defined on the basis of the interactions among hydrologic conditions, soils, and hydrophytic plants. Accordingly, they are expected to be sensitive to changes in abiotic conditions, hence vulnerable to changes in temperature, precipitation, and storm intensity. However, there is considerable uncertainty in how individual wetlands may respond given current uncertainties in climate predictions for local areas. Functional linkages between global circulation models and mechanistic biogeochemical models are needed to provide a basis for assessing the complex interactions that comprise the wetland C cycle. Those improved tools would then provide a basis to (a) prioritize wetlands that are particularly vulnerable, and (b) understand the consequences of the changes that are unavoidable as warming continues and precipitation regimes change.

Wetlands can be sustainably managed and restored, but they are sensitive to changes in abiotic conditions and nutrients. Forested wetlands can be sustainably managed and the C pools unaltered, especially if the hydrology is not modified. Conversely, improperly installed and managed silvicultural drainage systems may result in desiccation and the concomitant loss of C. Practices that minimize drainage and nutrient inputs and manage residues can reduce the global warming potential of wetlands managed for agriculture. Conversion to non-wetland results in diminished C functions but restored or created wetlands can lead to increases in functions and potential C sequestration. There are too few long-term studies that provide the basis for quantifying the effects of ongoing silvicultural and agricultural practices on wetland C cycling.

Terrestrial and coastal wetlands provide a variety of ecosystem services to society including moderating fluxes and pools of C. Although wetlands are sources of CH_4 to the atmosphere, historically they have been large sinks of CO_2, and overall sinks for C. The wetland C sink mitigates the effects of C source activities, such as the burning of fossil fuels. The rate of climate warming would increase without the wetland C sink. Although great strides have been made in C accounting we need to continue to standardize techniques and better utilize new technology to refine estimates of C pools and fluxes, especially as those pools and fluxes are influenced by climate change. Also, wetlands are generally not well represented in global circulation models that predict our future climate, even though they are disproportionately important for understanding global C cycling. Coupling data from undisturbed and disturbed wetlands, restored and created wetlands, and manipulation experiments with models will be essential for predicting the impact climate change will have on society.

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