CHAPTER

Effects of climate change on forested wetland soils

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Carl C. Trettin[a,*], **Martin F. Jurgensen**[b], **Zhaohua Dai**[a]

[a]*Southern Research Station, USDA Forest Service, Cordesville, SC, United States,*
[b]*School of Forest Resources and Environmental Science, Michigan Technological University, Houghton, MI, United States*
[*]*Corresponding author*

ABSTRACT

Wetlands are characterized by water at or near the soil surface for all or significant part of the year, are a source for food, fiber and water to society, and because of their position in landscapes and ecological structure help to moderate floods. They are also unique ecosystems with long-persistent flora and fauna. Because water is a driving factor for existence as a wetland, these systems are particularly vulnerable to climate change, especially as warming is accompanied by changes the quality and quantity of water moving through these systems. Because they are such diverse ecosystems, wetlands respond differently to stressors and, therefore, require different management and restoration techniques. In this chapter we consider forested wetland soils, their soil types, functions, and associated responses to climate change. Wetland processes are not well understood and therefore additional information is needed on these areas. In addition, more knowledge is needed on the interface between wetlands, uplands, and tidal waters.

Introduction

Wetlands are defined on the basis of saturated anaerobic soil conditions near the surface during the growing season and plants that are adapted to growing in anoxic soils (Cowardin et al., 1979). While specific definitions of wetlands vary by country or region, it is the presence of saturated soils and hydrophytic trees and understory plants that differentiate forested wetlands from upland forests. Non-forested wetlands typically have persistently high water table conditions that inhibit forest development. Additional considerations of wetlands are provided by Mitsch and Gosselink (2015), and wetland soil conditions and processes are thoroughly considered by Richardson and Vepraskas (2001) and Reddy and DeLuane (2008), while Moomaw et al. (2018) provide a global context for the consideration of climate change and wetlands.

Forested wetlands

Forested wetlands are critical for delivering a range of ecosystem services such as fiber, food, clean water, climate and flood regulation, coastal protection, and recreational opportunities (See Box 9.1). These wetlands exist in both salt and fresh water ecosystems, but threats from changing climatic

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BOX 9.1 Ecosystem services provided by wetlands

Ecosystem services provided by wetlands

Provisioning services: encompass products derived from the wetland, including: forest products, fisheries, animals for food, fur and products, crops, clean water.

Regulating services: reflect ecosystem functions that provide value, including: flood mitigation, water quality enhancement, shoreline protection, carbon sequestration and long-term storage.

Cultural services: reflect value derived from the existence of the wetland, including: recreation, aesthetics, education, wildlife viewing, and ecotourism.

Supporting services: encompass functions and processes that provide the foundation for the Provisioning, Regulatory and Cultural services, which include: plant production, soil processes that regulate carbon transformations and storage and water purification, water storage, flow regulation, groundwater recharge.

These four categories were developed in the Millennium Ecosystem Services report (2005), providing a functional categorization of wetland ecosystem services. The annual value of wetland ecosystem services ranges from \$25,600 ha^{-1} for swamps and floodplains to \$193,000 ha^{-1} for marshes and mangroves, which is much greater than upland forests (\$3800 ha^{-1}, Mitsch et al., 2015).

conditions especially extreme events, nutrient loading, and population growth and associated urban development threaten their sustainability and the valued ecosystem services they provide. Understanding where wetlands are located and how they function within the landscape is critical to developing strategies for mitigation and adaption strategies.

Description of wetland types

Wetlands are concentrated in the boreal and tropical zones (Fig. 9.1), with the type of wetland being determined by water regime, climate, and salinity (Mitsch and Gosselink, 2015). Tidal marshes occupy a zone of high salinity (>5 salinity parts per trillion (ppt)) that extends from the coastal fringe through the estuary to where water transitions to low salinity (<0.5 salinity ppt); typically freshwater tidal marshes persist in those reaches that experience daily inundation. Tidal freshwater forested wetlands are found upstream of freshwater marshes, in low-gradient coastal plains that exhibit regular soil saturation and periodic flooding. There is considerable variation in the composition and structure of these ecosystems. For example, in tropical coastal areas, mangrove forests can be the dominant vegetation, and encompass a variety of tree species that are adapted to growing in saline waters (Parida and Jha, 2010). Non-tidal wetlands also occur as marshes in areas with persistent saturation (e.g., lake margins) and as forests with hydric soils in a wide range of geomorphic positions.

Forested wetlands comprise a wide variety of forest types that occur across common geomorphic positions — depressions, riverine, slope, wet flats, lacustrine, and estuarine (Euliss et al., 2004). In the high-latitude boreal zone, forested wetlands are predominantly coniferous species from the genera *Pinus, Picea, Abies, Tsuga, Thuja, Larix*, with hardwood species (*Acer, Betula,* and *Tila* genera) common in the riparian zone (Dahl and Zoltai, 1997). Temperate forested wetlands are dominated by a large number of hardwood species, but coniferous species are also common (Brinson and Malvárez, 2002). Tropical forested wetlands include a wide variety of forest types reflecting hydrogeomorphic position and climate, including várzea and ígapó forests in South America (Klinge et al. 1990), peat swamp forests in central Africa and southeast Asia (Posa et al., 2011) and mangroves (Atwood et al., 2017).

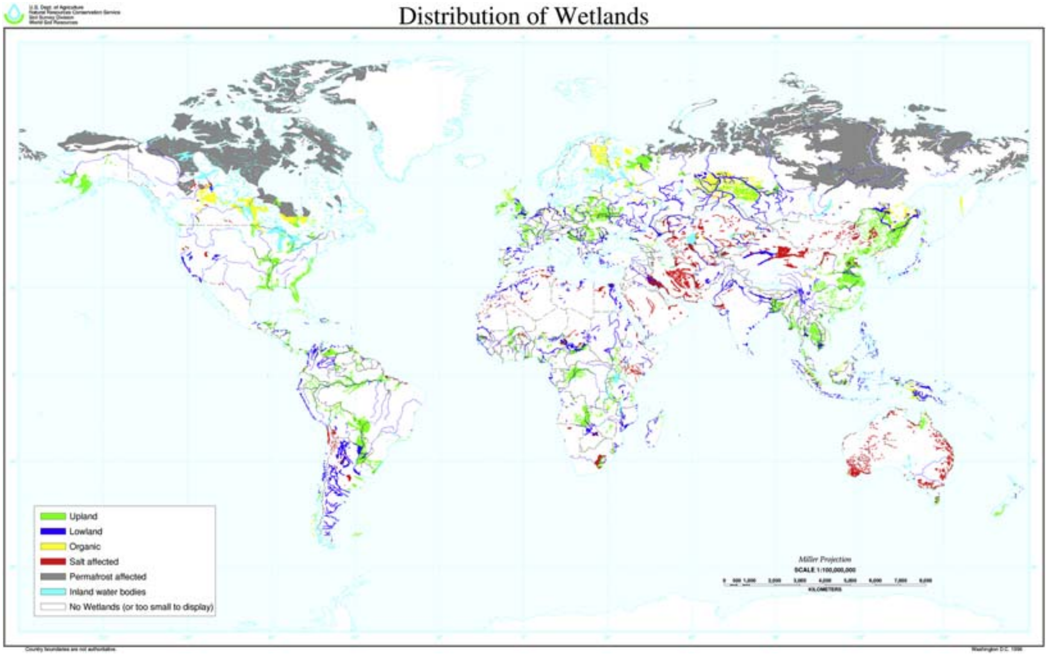

FIG. 9.1

Global distribution of wetlands.

Source: USDA-NRCS.

Estimates of the area of wetlands globally vary widely due differences in wetland definitions, available data and analyses (Table 9.1); commonly the global area of wetlands is reported as 4–5 $\times$ 10^6 km^2 (Mitra et al., 2005). Similarly, there is considerable variation in regional estimates of wetland area, and if those estimates are aggregated they will yield another estimate of global wetland area (Table 9.1). Correspondingly, the proportion of forested wetlands is difficult to determine, because some inventories do not distinguish them from non-forested wetland types. In North America, the proportion of forested wetlands is approximately 50% of the total wetland area (1.24 $\times$ 10^6 km^2, Kolka et al. 2018). In South America, there is an additional 2.12 $\times$ 10^6 km^2 of tropical forested wetlands (Klinge et al., 1990). Globally, the coastal mangroves comprise approximately 8.1 $\times$ 10^4 km^2 (Attwood et al., 2017).

Forested wetland soils

Mineral and organic soils

Hydrology is the factor that regulates wetland development, ecosystem function and associated ecosystem services (Euliss et al., 2004; Mitsch and Gooselink, 2015). In non-tidal terrestrial wetlands,

Table 9.1 Area of wetlands and peatlands globally, and areas of forested wetlands and peatlands within regions.

	Area (km^2)	Source
Global wetlands	0.54×10^6 to 18.42×10^6	Hu et al., (2017), Finlayson and Spiers (1999), Matthews and Fung (1987)
Area of forested wetlands		
North America	1.241×10^6	Kolka et al., 2018
South America	2.118×10^6	Klinge et al., 1990
Europe	2.570×10^6	Finlayson and Spiers (1999)
Asia	2.042×10^6	
Africa	1.213×10^6	
Mangroves	8.1×10^4	Attwood et al., 2017
Peatlands		
Global peatlands	3.524×10^6 to 4.232×10^6	Davidson and Finlayson (2018), Joosten (2010), Xu et al., (2018), Zoltai and Martikainen (1996), Zoltai et al., (1996)
Forested peatlands globally	0.696×10^6 to 0.969×10^6	Davidson and Finlayson (2018), Joosten (2010), Zoltai and Martikainen (1996)
Northern Europe	0.218×10^6	Vasander et al., 2003
Peat swamp forests—SE Asia	0.067×10^6	Posa et al., 2011
Peat swamp forests—central Africa	0.145×10^6	Dargie et al., (2017)
Peat swamp forest—Amazonia	0.035×10^6	

Note mangroves include both mineral and organic soils.

the sources of water sustaining the wetland include ground water and precipitation, which, in turn, interact with geomorphic position and climate to determine the type of wetland that develops (Winter, 1988). Some wetlands are sustained by precipitation alone and are called 'bogs' (Strack, 2008). In tidal wetlands, tidal hydraulics interact with groundwater, surface water, and precipitation to regulate ecosystem processes (Tiner, 2013). Accordingly, forested wetland soils are a function of the five factors of soil formation (e.g., parent material, climate, time, topography, and organisms; Jenny, 1941) analogous to upland forest soils, but the varying degrees of saturation near the soil surface cause wetland soil formation to be distinct from that of upland soils.

Soil saturation impedes the diffusion of oxygen into the soil, hence heterotrophic respiration within the soil matrix leads to depletion of oxygen, with the imposition of anaerobic conditions

FIG. 9.2

Redoxomorphic features in a mineral soil (Meggett series: fine, mixed, thermic Typic Albaqualf) from a bottomland hardwood forest in the lower coastal plain of South Carolina (USA).

forcing the use of other electron acceptors to support microbial respiration (Ponnamperuma, 1972, 1984). If present, nitrate is the first compound to be reduced (Ponnamperuma, 1972), which is a process that is beneficial from a water quality perspective (Hill, 1996). In mineral soils, the reduction of Fe and Mn minerals is the dominant reaction that produces distinct features within the soil profile that are diagnostic of the anaerobic conditions (Faulkner and Patrick, 1992). These redoxomorphic features are particularly prevalent within the zone that is periodically saturated (Fig. 9.2). During periods of low water table, air will diffuse into the soil restoring aerobic conditions. Accordingly, it is this alternating cycle of aerobic and anaerobic soil conditions within the active rooting zone that is characteristic of wetland soils (Richardson and Vepraskas, 2001), and the regulator of soil processes (Zedler and Kercher, 2005).

Both mineral and organic soils are common in forested wetlands. As noted above, mineral wetland soils are analogous to upland soils, except they contain redoxomorphic features (Faulkner and Patrick, 1992). In contrast, organic soils are characterized by an accumulation of partially decayed organic matter overlaying mineral sediments (Trettin and Jurgensen, 2003; Sahrawat, 2004). The thickness of the organic soil layers may vary from 30 cm to several meters. These organic soils, also termed peat soils or peatlands (Gorham, 1957; Belyea and Baird, 2006), occur in areas where the annual production of organic matter is only partially decomposed, resulting in the buildup of organic sediments over time. The rate of organic matter accumulation is highly variable, typically ranging from 0.5 to 2 mm yr^{-1} (Frolking et al. 2001).

In North America, approximately 50% of the non-tidal freshwater forested wetland area contains organic soils (Kolka et al., 2018); that proportion for global forested wetlands has been more difficult to determine. There are an estimated 3.5–4.2 $\times$ 10^6 km^2 of peatlands globally, with the area of forested peatlands ranging from 19 to 25% (Table 9.1). Peatlands comprise at least 60% of the global wetland area, acknowledging that there are considerable uncertainties in the global wetland area and the corresponding breakdown based on soil type (Table 9.1).

Soil processes

Carbon

Wetlands have a higher carbon density than uplands, hence they are recognized for their C storage and sequestration potential (Kayranli et al., 2010). Soils are the dominant ecosystem C pool in these systems, with forested peatlands containing significantly more C than mineral soil forested wetlands (1085 vs. 214 Mg C ha^{-1} respectively, Kolka et al., 2018). For comparison, upland forests contain approximately 70 Mg C ha^{-1} (Lajtha et al., 2018). Consequently, even though wetlands comprise only a small proportion of the global land area, they contain a disproportionate amount of the terrestrial C pool (Mitra et al., 2005). It is through the combination of high vegetation productivity and low decomposition rates due to saturated soils that these systems can accumulate such large amounts of soil carbon. Carbon dynamics in forested wetland soils are mediated by variable periods of aerobic and anoxic conditions that affect carbon storage and fluxes (Fig. 9.3, Trettin and Jurgensen, 2003). Changes in aeration are mediated primarily by the position of the water table (Ponnamperuma, 1984); gas diffusion rates through saturated soil are very slow. As a result, organic matter decomposition tends to be slower in hydric soils due to the lower efficiency of anaerobic respiration pathways (Reddy and DeLaune, 2008), with the decomposition rate in mineral soils generally exceeding that of peat. In addition to anoxia, temperature also affects the rate of organic matter decomposition (Neue et al., 1997). Low soil temperatures are a contributing factor to the widespread occurrence of peatlands in Alaska, Canada and Russia.

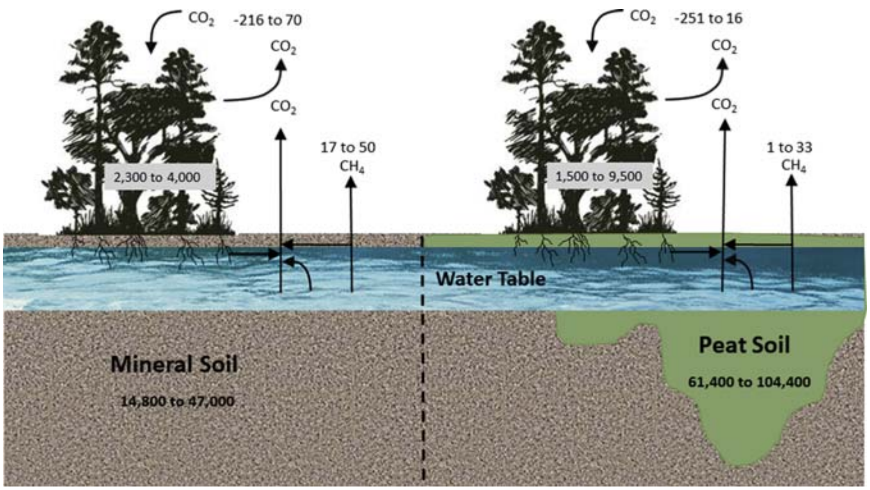

FIG. 9.3

Forested wetland carbon pools and fluxes on mineral and peat soils. Soil and vegetation carbon pools (g m^{-2}) are the range of carbon density (minimum to maximum) in North America. Annual carbon fluxes as CO_2 and CH_4 are the 95% confidence interval; a negative flux indicates a transfer from the atmosphere to the wetland.

Adapted from Trettin and Jurgensen, 2003 and Kolka et al., 2018.

As with most ecosystems, soil C content in forested wetlands represents the net long-term balance between organic matter inputs and decomposition losses (Trettin and Jurgensen, 2003). Forested wetlands are on average a sink for atmospheric C (Fig. 9.3), with little difference among North American peatlands and mineral soils (53 vs. 56 g C m^2 yr^{-1}, respectively; Kolka et al., 2018). However, sequestration rates can be quite variable, ranging from a net sink to a source of atmospheric carbon dioxide (CO_2) depending on the effects of weather conditions, forest health and land management (Nahlik and Fennessy, 2016; Kolka et al., 2018).

Another major difference with upland forest soils is that forested wetlands soils can produce large amounts of methane (CH_4), which is a greenhouse gas that exerts 20 times the radiative forcing of CO_2; if not metabolized in the soil profile, CH_4 may be emitted to the atmosphere (Trettin and Jurgensen, 2003; Krauss and Whitbeck, 2012). Methanogenesis requires a very low redox potential within the soil (Ponnamperuma, 1972), and the major factor affecting CH_4 emissions to the atmosphere is the water table position because CH_4 may also be oxidized within an aerated portion of the soil profile (Bridgham et al., 2013). When the water table is more than 15 cm below the soil surface, emissions may be greatly reduced due to CH_4 oxidation (Trettin et al., 2006). Conversely, when the water table is at or above the soil surface, CH_4 is not oxidized but rather CH_4 is release primarily through ebullition (Whalen, 2005). Soil micro-topography affects CH_4 emissions because it effectively changes the volume of soil that is aerated; as a result emissions tend to higher in hollows or depressions, and lower on hummocks or elevated positions (Jauhiainen et al., 2005). Vegetation also influences CH_4 release from wetlands, with aerenchyma providing a pathway to directly vent CH_4 produced in the subsoil to the atmosphere; this pathway is well documented in gramminoids but it has also been documented in trees (Bridgham et al., 2013). Methane emissions from mineral soil wetlands is greater than peatlands (Kolka et al., 2018), but there is considerable uncertainty in the emission rates due to few measurements.

Nitrogen

Soil N dynamics in forested wetlands are analogous to upland forests except for the regular occurrence of anaerobic conditions within the solum (Fig. 9.3) (Hefting et al., 2004). Nitrate (NO_3^-) is the first compound reduced once oxygen (O_2) becomes limiting as a result of denitrification (Bowden, 1987). This is the process that makes wetlands effective at reducing NO_3^- in runoff from upland areas; thereby limiting entry into aquatic systems (Hill, 1996). Correspondingly, the NO_3^- level in wetland soils is typically very low during periods of saturation. However, once a wetland soil drains and becomes aerated, nitrification may occur (Howard-Williams, 1985). Nitrous oxide (N_2O), a greenhouse gas that provides 200 times the radiative forcing of CO_2, may be produced as an intermediate product during denitrification; however, emissions to the atmosphere are difficult to detect, and are small relative to other land uses, especially agriculture (Tian et al. 2015).

Tidal forest soils

In contrast to terrestrial wetlands that have hydrologic regimes governed by precipitation, ground water, and evapotranspiration, tidal forested wetlands experience oscillations in soil moisture due to the tide, in addition to the aforementioned factors (Tiner, 2018). Tidal freshwater forested wetlands (TFFW) occur in low-gradient coastal plains between freshwater marshes and the non-tidal forested wetland riparian zone (Conner et al., 2007). The hydrology of the TFFW is regulated by sea level, tide and river or stream discharge (Day et al., 2007). The soils are typically developed in mineral substrates, and the hummock and hollow micro-topography, which is characteristic of TFFW, influences forest community composition (Duberstein and Conner, 2009) as well as biogeochemical processes that drive greenhouse gas emissions (Anderson and Lockaby, 2007). However, there is considerable variability among TFFWs; for example, in a comparative study among tidal and non-tidal forested wetlands, Verhoeven et al., (2001) did not find differences in N and phosphorous (P) mineralization. While the regular periods of soil saturation are conducive for CH_4 production in the TFFWs, the majority of the production is oxidized within the aerated soil volume (Megonigal and Schlesinger, 2002).

Mangroves are unique among forested wetlands because of the marine sediment biogeochemistry that is mediated by tidal hydrology (Alongi, 2009). Mangrove soils are typically mineral sediments that are high in organic C (median soil C = 2.2%; Kristentsen et al., 2008), which accounts for the large soil C stock (approximately 370 Mg C ha^{-1} in the upper 100 cm; Jardine and Siikamaki, 2014). Peat soils may also exist in mangroves (Hossain and Nuruddin, 2016), typically in areas that pond or have low flow, with root biomass likely as the primary source of the organic soil (Ezcurra et al., 2016). Mangroves commonly exhibit variable inundation cycles due to position on a landscape, tidal amplitude, lunar cycles and storm events. For example, some areas below the high tide mark will be flooded daily, where other areas may only be inundated on a monthly or seasonal basis. The extent and duration of the tidal inundation and the salinity of the water affect both the tree species distribution and their productivity (Alongi, 2009).

Effects of climate change

The effects of climate change on wetlands will be mediated through direct changes in the climatic regimes and by anthropogenic actions that are also linked to climate change. Context for climate change effects on wetland ecology are provided by Erwin (2009), Mitsch et al., (2013), and Mitsch and Gosselink (2015). The following section considers the principle drivers associated with climate change, including temperature, precipitation regime, extreme events, sea level rise and anthropogenic activities, and their associated effects on forested wetland soil processes. Table 9.2 provides a contextual framework linking the generalized response of selected ecosystem services to those drivers (See Box 9.1).

Atmospheric temperature

The doubling of atmospheric CO_2 concentration that is predicted to result in a warming of 2–4.5 °C globally (Rogelj et al., 2012) has implications for soil temperature since they are functionally linked (Zheng et al., 1993). Atmospheric warming may be expected to increase the rate of microbial mediated soil processes, which include organic matter decomposition (Davidson and Janssens, 2006) and greenhouse gas emissions (Reichestein et al., 2005). However it must be noted that those processes are also affected by plant production, soil aeration and other factors (Middleton and Souter, 2016), hence the net effect of increased temperature will depend on those interactions (Davidson and Janssens, 2006).

Higher atmospheric temperatures can change surface water hydrology by altering precipitation and runoff patterns, changing snow melt patterns, and increasing evapotranspiration (Nijssen et al., 2001). Similarly, forest productivity may increase slightly due to increased photosynthetic activity, but the

Table 9.2 Potential effects of climate change on ecosystem services derived from forested wetlands.

		Precipitation				
Ecosystems services	**Warmer temp**	**Drier**	**Wetter**	**Extreme events**	**SLR[a]**	**Human**
Forest products	+	+	–	–	–	–
Wildlife	nd[b]	+	–	–	–	–
Flood mitigation	nd	+	–	nd	–	–
Water quality	+	+	–	–	–	–
Shoreline protection	nd	–	–	–	–	–
Carbon sequestration	–	–	–	–	–	–
Recreation	nd	+	–	–	–	–
Aesthetics	nd	–	nd	–	–	–
Organic matter decomposition	+	+	–	–	+	+
Methane emission	+	–	+	nd	–	–
Denitrification	+	–	+	nd	+	–
Groundwater recharge	nd	–	+	nd	nd	–

[a]*SLR is sea level rise.*
[b]*nd = not determined.*

gains may be off-set by increased respiration. In the boreal region, warmer temperatures may expand the growing season and increase productivity (Moor et al., 2015), but that effect may not be sustained on sites where soil moisture is limiting (D'Organgville et al., 2018). The increase in productivity will increase the amount of C input into the soil in the form of increased litter fall and root productivity (Trettin and Jurgensen, 2003). The rates of biogeochemical property changes that are sensitive to temperature (e.g., organic matter decomposition, denitrification, methanogenesis) may also be expected to increase due to increases in soil temperature.

Precipitation

The effects of climate change on forested wetland soils will be mediated primarily through changes in precipitation because water regulates hydrologic processes in terrestrial wetlands, and these processes directly affect many biogeochemical processes. While there are considerable uncertainties in the projected changes in precipitation, there is consensus among projections that some areas will become drier and others will become wetter. Accordingly, consideration of the precipitation effects should address scenarios where wetlands may become either wetter or drier, depending on the precipitation response. Specific responses will also be dependent on the wetland type and geomorphic position.

Wetlands that are dependent on precipitation to sustain their water balance are particularly vulnerable to changes in precipitation patterns in contrast to those that are sustained primarily by ground water discharge (Winter, 2000). Assessing the sensitivity of precipitation-dependent wetlands to altered precipitation regimes, Fay et al., (2016) showed that the ratio of mean annual precipitation to potential evapotranspiration was an effective metric to predict wetland response to changing precipitation. Accordingly, forested bogs and ephemeral ponds will be particularly susceptible. A reduction in the water table within the wetland will increase the drained soil volume, thereby facilitating aerobic metabolism. These aerobic conditions, in turn, will increase organic matter decomposition (Griffis et al., 2000; Laiho 2006; Clair et al., 1995; Morrissey et al., 2014) and so alter greenhouse gas emissions from soil. The increased decomposition will cause a corresponding increase in soil CO_2 flux to the atmosphere (Field 1995; Moor et al., 2015). However, CH_4 emissions may be reduced significantly, especially once the water table is 15 cm or more below the soil surface (Trettin et al., 2006). The decrease in water input may also result in a reduction in water output from the wetland, and any increase in water storage within the wetland may enhance flood mitigation. For forested peatlands the consequence of a reduction in water table during the growing season is an increased risk of wildfire and consumption of the peat soil (Flannigan et al., 2009).

Increases in precipitation amount may be expected to raise the water table level within the wetland, effectively decreasing the aerated soil volume. The general effect on biogeochemical processes would be to reduce organic matter decomposition, and increase methanogenesis and denitrification. In temperate and tropical areas, the productivity response to climate change will also be affected by the changes in the precipitation regime. In areas that experience an increase in the frequency and duration of saturated soils, productivity may increase (Field 1995; Noe and Zedler, 2001) or decrease depending on the antecedent condition. For example, in tropical wetlands the productivity response coincides with change in precipitation (Barros and Albernaz 2015). In the case of subtropical coastal marshes, moderation of winter temperatures is facilitating the northward expansion of mangroves, which are displacing marsh ecosystems (Saintilan et al., 2014; Armitage et al., 2015; Simpson et al., 2017), changing the form of organic matter input to the ecosystem, which in turn may alter the faunal communities (Smee et al., 2017).

As a result of decreased water storage within the wetland, increased precipitation may also increase frequency and duration of flooding events. Similarly, extreme events may also increase flooding due to both the amount and intensity of the precipitation that impair the provision of ecosystem services (Talbot et al., 2018). However, specific changes may also be strongly affected by land use (Martin et al., 2017).

Extreme weather events

Extreme weather events may be characterized as those near or exceeding the historical ranges of variability, are usually severe or unseasonable, and can often be hard to predict. Climate change is recognized for causing an increase in the frequency of extreme weather events, which have been characterized by increased storm precipitation intensity and amount. Increased frequency of tornados or tropical storms may also occur. Again effects of extreme events on wetlands will be dependent on antecedent conditions, as well as the direct consequences of physical forces associated with the event.

Extreme precipitation events are characterized by storms that deposit a large volume of water in a short period of time. These storms may produce flash-flooding as infiltration capacity of the landscape is exceeded by the storm intensity. Wetlands typically provide some buffer capacity to flooding due to their inherently high internal storage, making them valuable for mitigating floods. However, flash flooding may overwhelm the storage capacity, and rapid flow associated with the flooding may cause erosion that alters the flow and storage characteristics, along with other ecosystem services (Talbot et al., 2018).

Extreme events that include strong winds (e.g., tornados, tropical storms) may result in the physical destruction of the forest vegetation with long-term consequences to wetland hydrology. Large-scale forest mortality that follows these events tends to result in a reduction in evapotranspiration that alters the hydroperiod and associated biogeochemical processes, an effect that can last for several years. For example, in 1989 Hurricane Hugo devastated the forests in the lower coastal plain near Charleston, South Carolina where the hydrology of a wetland-dominated watershed took 11 years to return to pre-storm characteristics (Jayakaran et al., 2014).

The role of shoreline protection by forested wetlands includes both terrestrial wetlands that border lakes and rivers, and mangroves in the coastal zone. Extreme events may compromise that function due to impacts from flooding, high flow and loss of vegetation (Zedler, 2010). The impact may include eroded shoreline, forest degradation and salinization (Morton and Barras, 2011). The recovery of the shoreline depends on the vegetation and climatic zone; in the tropics, recovery of the mangroves is rapid, in comparison to temperate or boreal settings.

Sea level rise

Sea level rise of 0.3 to 1.8 m by 2100 (Nicholls and Cazenave, 2010) will result in significant changes in the hydroperiod and salinity of coastal forested wetlands as well as shifts in freshwater, sediment, and nutrients due to changes in inputs from upland watersheds (Kirwan and Megonigal, 2013; Megonigal et al., 2016). However, in many areas of the world, the coastal land mass is subsiding; hence the net change in the hydroperiod of coastal wetlands will be dependent on local conditions. Tidal freshwater forested wetlands are particularly vulnerable to changes in flooding and salinity that may result in the mortality of freshwater wetland species, and the subsequent replacement by salt-tolerant species (Allen et al., 1998). For those wetlands, there will be significant changes in soil carbon and nutrient cycling. For example, the addition of salt water to a TFFW can increase P availability (e.g., Caraco et al., 1989), decrease ammonium absorption (Howarth et al., 1988), and depending on conditions either stimulate or inhibit organic matter decomposition (D'Angelo and Reddy 1999; Weston et al. 2006; 2011). For this last effect, the causes of the contrasting decomposition responses remain unclear (Sutton-Grier and Megonigal 2011), limiting certainty in the use of any mitigation actions.

Many of the TFFW are extremely vulnerable to changes in salinity due to their position at the terrestrial–marine interface (Krauss et al., 2018). A key factor to the sustainability of TFFW is sediment accretion to compensate for changes in sea level (Morris et al., 2002); recent research suggest that the current rate of sea level rise exceeds the sedimentation rates (White and Kaplan, 2017). Failure to keep pace with sea level and the associated intrusion of saltwater has significant effects on soil biogeochemical processes, which, in turn, affects the health of the forest, its rate of degradation and eventual conversion to marsh and potentially open water (Tully et al., 2019). These changes in wetland condition and type results in the degradation of ecosystem services, which begs the question regarding appropriate management responses. White and Kaplan (2017) provide useful perspectives on the management alternatives to address the threats posed by sea level rise to the TFFWs.

Mangroves, occurring in marine coastal zone, are subject to changes in sea level and the direct impacts from tropical storms. While tropical storms can alter the existence and condition of mangroves (Gilman et al., 2008), there will be relatively minor changes in soil processes within the mangroves because strong marine influences are inherent. Correspondingly, sea level rise may be expected to facilitate the landward migration of mangroves into TFFW, particularly in low gradient coastal plains.

Forested wetland loss

The rate of wetland loss is difficult to estimate due to inconsistencies among national inventories, the basis for reporting and assessment intervals (Finlayson et al., 1999). Davidson's (2014) recent assessment of wetland loss is particularly valuable in providing a global perspective, which shows that global loss of forested wetlands has likely been 54%–57% since AD 1900 or higher depending on the historic baseline used in the evaluation. It should be noted that the rate of loss has declined significantly in North America and Europe in recent decades. For example, the United States has seen approximately 53% of its wetland area lost (Dahl, 1990), but the rate has declined significantly in the past 30 years, exhibiting a net decline of 0.06% over the period of 2004–09 (Dahl, 2011). The historical drivers for wetland loss have been for agricultural and human developments. In contrast, wetland loss remains high in Asia (Davidson, 2014), with a principle driver being the conversion of tropical peatlands into oil palm plantations (Koh et al., 2011). The conversion of the tropical peatlands has serious ramifications to regional biodiversity, a host of ecosystem processes (e.g., water storage, water quality, food), and ecosystem C storage (Rehman et al., 2014).

Development in and around forested wetlands is the biggest threat to forested wetlands globally, and much of that pressure is a result of direct and indirect effects of climate change. As climate change alters agriculture production in terrestrial uplands, the need for new land may encourage the development of wetlands. The pressures on coastal wetlands are particularly apparent, where activities such as aquaculture or salt production are seen as alternative livelihoods. Deforestation is another threat to forested wetlands; however if it is solely deforestation, it should not change the wetland into non-wetland status, but it will result in a change of ecosystem services.

While mangroves occupy a very small proportion of the forested wetlands (Table 9.1), the threats to mangroves highlight challenges to all forested wetlands (Alongi, 2008), and a focus to address those threats is needed to sustain their highly-valued ecosystem services. Conversion to agriculture is the largest threat to mangroves, resulting in large-scale C emissions. Similarly, conversion to salt production or aquaculture have had similarly large consequences to the carbon stocks. Erosion, exacerbated by extreme events, is another cause of mangrove loss, but new sediment deposition also provides land for the development of new mangroves (Shapiro et al., 2015). There remains considerable uncertainty about the effects of conversion of mangroves on forest carbon pools and fluxes, due to paucity of baseline data and manipulative studies. Similarly, restoration of mangroves is an attractive response to recoup wetland functions, but there is little information regarding changes in the C emissions, for example.

Perspectives

The basic ecological, hydrological and biogeochemical principles governing forested wetlands are well established, and hydric soil processes are well understood. But this understanding has been derived from information obtained in studies across an array of individual sites and wetlands. Given that forested wetlands exist in all the major landform settings and across most bioclimatic zones, there remains considerable uncertainty in the manifestation of specific processes (e.g., CH_4 emissions). Predicting the responses of forested wetlands to climate change is further complicated by the uncertainties of future climatic conditions at any given location. Hence, forested wetland soil response requires some information about the current conditions and projections of future climatic conditions, especially with respect to changes precipitation amount, intensity, and temporal distribution including periodicity. Based on basic principles, we can make general statements about the potential effects of climate change on forest soil processes, but they remain difficult to generalize across a region because of the locally determined and highly variable hydrologic influences and the concomitant uncertainties in the future climatic conditions.

Mechanistic models, operating within a spatial framework, provide a means for assessing forested wetland responses to projected environmental change. In this manner, the hydrologic setting, forest and soil conditions and climatic factors can be incorporated into a landscape-scale assessment of wetland soil processes. The issue of scale is an important consideration with respect to any forested wetland soil assessment (Cao et al., 2019). At the plot to watershed scale, explicit consideration of wetland hydrology is paramount, and may be accomplished through the use of empirical models (Zhu et al., 2017) or linking hydrology and soil biogeochemical models to assess forested wetland responses (e.g., Dai et al., 2010). Since wetlands are often a small proportion of the watershed or landscape area, considerations of appropriate scale where wetlands can be accurately represented is important. For example, Dai et al., (2010) found that representation of wetland at a 15 m × 15 m pixel was necessary to assess methane fluxes from wetlands within a southern pine-hardwood flatwoods watershed. However, at regional to global scales, simplified representations of wetland processes provide an effective means for assessing wetland responses to climate change (Mitsch et al., 2013; Zhang et al., 2018).

Although wetlands comprise a relatively small portion of the terrestrial landscape, they provide many ecosystem services (e.g., water purification, carbon sequestration) that are mediated by soil processes. These services are dependent on the hydrologic regime and so any changes in hydrology caused by land clearing, climate change, drainage, or other disturbances are likely to alter or even eliminate these function. Furthermore, although wetland restoration activities can help off-set loss of natural wetlands, restoration activities must be informed by local site factors.

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