# Chapter 5
# Carbon Sequestration in Wetland Soils

## Contents




**Abstract** Crops adapted to wetland conditions such as rice (*Oryza sativa* L.) have been cultivated on waterlogged anoxic soils for millennia. Grazing of livestock is another important agricultural activity in wetlands. Wetlands including peatlands may cover up to 26.9 million $km^2$ globally, and wetlands may contain up to 158 Pg soil organic carbon (SOC) to 1 m depth, but knowledge on wetland distribution, extent, and volume needs to be strengthened. Peatlands are organic-rich wetlands and cover $\sim$4 million $km^2$ with large areas in the Northern Hemisphere. Over centuries to millennia, organic soils of peatlands have accumulated globally >750 Pg of carbon (C) as peat, sometimes to several meters depth as decomposition rates are greatly reduced under wet and acidic soil conditions. Thus, peatlands while covering only 3% of the global ice-free land area store more than one-fourth of the global SOC stock. Stocks in northern peatlands alone may store >600 Pg C, and their utilization for agriculture may release large amounts of carbon dioxide ($CO_2$) by peat oxidation. For example, $\sim$1 Pg $CO_2$ is emitted annually from drained peatlands (including emissions from fire), with high emissions especially from drained organic soils in tropical regions often for cultivation of oil palms (*Elaeis guineensis* Jacq.). Further, cumulative net emissions from global peatland use have been estimated at 6 Pg C for the period 1850–2015. Wetlands are also among the major biogenic methane ($CH_4$) sources contributing to about 30% of the total $CH_4$ emissions and will increasingly contribute to the projected climate change. Further,


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C losses from wetlands may also increase in the future because of the projected climate change. Thus, sustainable intensification (SI) should be applied to reduce $CH_4$ and C losses from wetlands. Options include, for example, restoring drained agricultural land-use types to flooded conditions, improved fertilizer and water management of paddy fields, and breeding of new crop cultivars better adapted to anoxic wetland soil conditions. In this respect, paludiculture of wetlands is promising as a suitable agricultural practice with the cobenefit of C sequestration. This chapter begins with a general overview on wetlands and peatlands. Then, the peatland C balance is discussed in more detail. Agricultural use and management of wetland soils are presented in the following section. The final section discusses options for more 'climate-friendly' agriculture in wetlands.

**Keywords** Organic soils · Paludiculture · Tropical peatlands · Oil palm cultivation · Rice cultivation · Methanogenesis · Sustainable intensification
Fire · Gaseous emissions

## 5.1 Introduction

Wetlands are far more important in the biosphere than their extent suggests. Unmanaged wetlands are hot spots of biological diversity (Junk et al. 2006), ecosystem productivity (Rocha and Goulden 2009), and economic activity (aquaculture, tourism, timber; Mitsch and Gosselink 2015). Wetlands are key regulators of biogeochemical cycles, including water flows and associated nutrients (C, N, P), pollutants and sediments, coastal erosion, and land stabilization (Junk et al. 2013). Wetlands also play fundamental roles in climate change regulation and mitigation (Gumbricht et al. 2017). Wetlands act as long-term soil organic carbon (SOC) reservoirs dating back to the Holocene. However, humans have practiced crop cultivation and livestock grazing in wetlands since thousands of years. For example, crops were cultivated 6,000 years ago in the floodplains of Mesopotamia (Verhoeven and Setter 2010). The cultivation has affected the large amounts of organic matter (OM) which have accumulated in wetland or hydromorphic soils (Jungkunst et al. 2012), and, especially, organic-rich peatland soils as oxygen deficiency results in far less efficient microbial decomposition (Schlesinger and Bernhardt 2013). OM enters wetlands through net primary production (NPP) and transport in both solid and dissolved forms, while decomposition and transport out of wetlands result in OM loss (Keller and Medvedeff 2016). NPP is the ultimate source for soil organic carbon (SOC) of wetlands, and aboveground NPP for wetlands varies widely from 10 to 4,600 g C $m^{-2}$ $y^{-1}$ with <1,000 g C $m^{-2}$ $y^{-1}$ for northern peatlands (Keller and Medvedeff 2016; Gopal and Masing 1990). However, it is generally recognized that belowground NPP can be significant but is rarely measured similar to that in the non-wetland soils.

While extensive agricultural practices have less severe effects on wetland ecosystems, complete conversion of wetlands for intensive agriculture by drainage, nutrient and water management, and vegetation clearance, and peat extraction may severely affect carbon (C) sequestration and stocks. For example, world resources of peat are estimated at 1,900 Pg of which 0.026 Pg peat was produced in 2012 (Mitsch and Gosselink 2015). However, a major issue is that wetlands and peatlands are defined differently across countries and scientific disciplines (Jackson et al. 2017). In the following section, wetlands and the main subclass peatlands will be characterized with a focus on peatland soils as those dominate wetland soil C stocks.

### *5.1.1 Wetlands*

Wetlands are distinct ecosystems with intermittent to permanent waterlogged soil conditions resulting in anoxia because oxygen consumption exceeds the rates of its delivery (Schlesinger and Bernhardt 2013). However, there is no universal definition of wetlands as the characteristics distinguishing wetlands from other ecosystems make them also less easy to define (Mitsch and Gosselink 2015). Specifically, depth and duration of flooding vary considerably from wetland to wetland and from year to year. Wetlands may be located at the margins between deep water and uplands and be influenced by both. Wetland species include obligate species adapted to wet environments and facultative species adapted to either wet or dry conditions. Wetlands vary widely in size, and their location can also vary greatly. Also, wetland condition or the degree of modification varies greatly from region to region and from wetland to wetland (Mitsch and Gosselink 2015). The Ramsar Convention considers wetlands to be 'areas of marsh, fen, peatland or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish or salt, including areas of marine water the depth of which at low tide does not exceed six meters.' (Davis 1994). Under this definition, wetlands include lakes and rivers, swamps and marshes, wet grasslands and peatlands, oases, estuaries, deltas and tidal flats, nearshore marine areas, mangroves and coral reefs, and human-made sites such as fish ponds, rice (*Oryza sativa* L.) paddies, reservoirs, and salt pans. Ramsar (2013) differentiates the five major wetland types estuarine, lacustrine, marine, riverine, and palustrine wetlands. However, a wetland definition satisfactory to all users has not been developed (Mitsch and Gosselink 2015).

Estimates for the global wetland area vary fourfold based on modeled area simulations (7.1–26.9 million km$^2$) and threefold (4.3–12.9 million km$^2$) in observational mapping (Melton et al. 2013). Variability in areas and volumes are related to definition issues discussed previously and to the temporality of the inundation patterns which complicate comparisons among estimates (Page et al. 2011). Also, wetland ecosystem boundaries can be difficult to assess, and detailed inventories for the major regions South America, Africa, and Russia are missing (Schlesinger and Bernhardt 2013). Major wetland regions are located in the Neotropics (4.15 million km$^2$), Europe (2.58 million km$^2$), North America (2.42 million km$^2$), and Asia (2.04 million km$^2$) (Finlayson et al. 1999). Further, about

**Table 5.1** Estimated global area (million km$^2$) of wetland ecosystem types (modified from Mitsch et al. 2009)

| Type of wetland | Area |
|---|---|
| Tidal salt marshes | 0.1 |
| Tidal freshwater marshes | 0.02 |
| Mangrove wetlands | 0.24 |
| Freshwater marshes | 0.95 |
| Freshwater swamps and riparian forests | 1.09 |
| Peatlands | 3.5 |

1.63 million km$^2$ under paddy rice cultivation on flooded mineral soils was harvested in the year 2014 (http://www.fao.org/faostat/en/#home). The largest wetland types are peatlands followed by freshwater swamps/riparian forests, and freshwater marshes (Mitsch et al. 2009; Table 5.1).

Recently, improved estimates for tropical and subtropical wetland areas and volumes were reported by Gumbricht et al. (2017). Estimates were based on an expert system approach to estimate wetland and peatland areas, depths, and volumes, which relies on three biophysical indices related to wetland and peat formation (i) long-term water supply exceeding atmospheric water demand; (ii) annually or seasonally waterlogged soils; and (iii) a geomorphological position where water is supplied and retained. Tropical and subtropical wetland area covers an estimated 4.7 million km$^2$. The American continent is the major contributor (45%), and Brazil, with its Amazonian interfluvial region, contains the largest tropical wetland area (800,720 km$^2$) (Gumbricht et al. 2017). Historically, the global wetland area was much larger as humans have been draining, in-filling, and converting both coastal and inland wetlands for many centuries. Wetland area conversion and loss were in excess of 50% of the original area and are continuing in all parts of the world, particularly rapidly in Asia (Davidson 2014).

Storage of large amounts of organic C is among the major benefits of wetlands. Decomposition rates in wetland soils are low due to anoxic conditions resulting in accumulation of dead plant material. However, data on SOC stocks are uncertain as the volume of wetland soils (Gumbricht et al. 2017), and its C mass is generally not known as there is no single definition of wetland. Globally, SOC stocks to 1 m depth were estimated at 82 Pg C for permanent wetlands and 158 Pg C for all wetlands except open water (Köchy et al. 2015). Tropical wetlands alone may contain 38.3–39.9 Pg SOC in the top 1 m. Köchy et al. (2015) suggested that the areal extent of organic soils, their depth, and the bulk density at different depths should receive the greatest focus of future soil mapping activities to improve those estimates. In contrast to peatlands, wetlands such as salt marshes can accumulate SOC quite rapidly (Kirwan and Mudd 2012). Specifically, blue carbon, i.e., C sequestered by vegetated coastal ecosystems for long-term storage (Howard et al. 2014), is a potential sequestration component for atmospheric $CO_2$. Tidally influenced wetlands are attributed with 36% of the total sequestration by all wetlands and 18% of the total C sequestration of all ecosystems in the conterminous USA (Bridgham et al. 2007). Across the continental USA, there are 1,153–1,359 Tg of

SOC in the upper 100 cm of soils across a total of 24,945.9 $km^2$ of tidal wetland area (Hinson et al. 2017).

Wetland soils not only sequester C but simultaneously emit both carbon dioxide ($CO_2$) and methane ($CH_4$) as respiratory by-products under waterlogged conditions (Bridgham et al. 2013). Globally, $CO_2$ emissions from wetlands are estimated at 2.1 Pg C $yr^{-1}$ (Aufdenkampe et al. 2011). The complete mineralization of OM in wetlands requires a complex community of microbes. Specifically, $CH_4$ is produced by fermentation processes of methanogenic microbes. Among the controls of $CH_4$ emissions from wetland are soil temperature, water table, and vegetation, but these relationships are modified depending on wetland type (bog, fen, or swamp), region (subarctic to temperate), and disturbance (Turetsky et al. 2014). Microbial controls also regulate wetland $CH_4$ cycling (Bridgham et al. 2013). Globally, soils of wetlands are among the major biogenic $CH_4$ sources contributing to about 30% of the total $CH_4$ emissions with natural wetlands emitting 177–284 Tg $CH_4$ $y^{-1}$ for the decade of 2000–2009 (Ciais et al. 2013). Unmanaged wetlands are the largest and most uncertain natural sources of $CH_4$ and also the presumed drivers of the inter-annual variations in $CH_4$ atmospheric growth rates (Petrescu et al. 2015).

Increasing temperature, which increases wetland area by causing high-latitude permafrost thawing (Tchebakova et al. 2011), drives a long-term trend in wetland $CH_4$ emissions (McNorton et al. 2016). Over several centuries, sustained $CH_4$ emissions in natural northern latitude wetland ecosystems are typically offset by $CO_2$ uptake (Petrescu et al. 2015). In contrast, high rates of NPP and litter decomposition in tropical wetlands result in both high $CO_2$ and $CH_4$ emissions. For example, Sjögersten et al. (2015) estimated that 4,540 and 90 Tg $CH_4$ $y^{-1}$ are released from tropical wetlands globally. Under the 'worst-case' RCP8.5 scenario with no climate mitigation, boreal $CH_4$ emissions are enhanced by 18.1–41.7 Tg, due to thawing of inundated areas during the cold season and rising temperature (Zhang et al. 2017). By 2099, tropical $CH_4$ emissions are projected to accelerate with a total increment of 48.4 Tg to 87.4 Tg. Thus, climate mitigation policies must consider mitigation of wetland $CH_4$ feedbacks to maintain average global warming below 2 °C (Zhang et al. 2017). Wetlands may also emit nitrous oxide ($N_2O$) due to their high potential for denitrification. For example, under favorable hydrological conditions, undisturbed wetlands are reported to act as moderate $N_2O$ sources (Frolking et al. 2011. However, there are no reliable estimates of the contribution of wetlands to global $N_2O$ emissions (Schlesinger and Bernhardt 2013).

Aside from the utilization for agriculture, the wetland SOC stock is also under threat from the climate change. Specifically, increases in temperature changes in the total amount and distribution pattern of precipitation, and, in the case of coastal wetlands, sea level rise may increasingly alter wetland soil C stock and sequestration (Settele et al. 2014). However, there is currently no consensus on the impact of climate change on greenhouse gas (GHG) emissions and the SOC stock of global wetlands (Meng et al. 2016). For example, less frequent inundation is likely to foster oxidation of OM, but the loss of OM may be coupled with enhanced NPP and reduced $CH_4$ emissions (Schlesinger and Bernhardt 2013). Some freshwater wetlands may become brackish or saline as a result of saltwater intrusion into coastal wetlands from sea level rise or through drought-induced evaporative concentration of salts resulting in enhanced OM decomposition. Predicting the effect of rising temperatures on wetland biogeochemistry is, particularly, difficult due to complex positive and negative feedbacks. For example, permafrost melting may lead to wetland drainage, and OM decomposition in many boreal wetlands may be enhanced. Another threat to wetland C stocks from the climate change is the higher susceptibility to wildfire. Further, elevated $CO_2$ concentrations may stimulate enhanced rates of OM oxidation and $CH_4$ production in wetlands (Schlesinger and Bernhardt 2013).

### *5.1.2 Peatlands*

Forested and non-forested peatlands are among the wetland types differentiated by the Ramsar Convention. Peatlands may broadly be defined as organic-rich wetlands and include Histosols, bogs, and fens (Keller and Medvedeff 2016). Peatland refers to a peat covered landscape. All peats are formed in wetlands, but not all wetlands are associated with peat because not all wetlands have the conditions to form peat (Rydin and Jeglum 2006). However, classification of peatland types is challenging because of gradients in alkalinity, pH, nutrient availability, and vegetation among peatland types. Thus, similar to wetland, there is no agreed definition of peatland. Most definitions require a minimum peat thickness of 30 cm for an ecosystem to be considered a peatland (Loisel et al. 2017). Boreal, temperate, tropical, high-altitude, and low-altitude peatlands can also be differentiated, and common characteristic is that they contain a large SOC stock (Biancalani and Avagyan 2014). Peat is the OM formed in situ which has been accumulated in soil over long periods of time as a result of incomplete breakdown of dead plant remains due to oxygen deficiency usually caused by waterlogging (Osvald 1937). For example, tropical peatlands may have been involved in the global C cycle (GCC) prior to the last glacial maximum (26,500–19,000 years ago) and, thus, are older than subarctic and temperate peatlands (Page et al. 2004).

Similar to peatland, there is also no universal definition for peat. For example, peat can be defined as sedentarily accumulated material consisting of at least 30% (dry mass) of dead organic material (Joosten 2010). If the plant remains are still recognizable in wetlands, organic materials are called peat (Schlesinger and Bernhardt 2013). While peat in tropical regions is largely composed of trees, mosses and sedges are major components of peat in boreal and temperate regions (Biancalani and Avagyan 2014). To differentiate organic from mineral soils, an OM content of >20–30%, together with a depth criterion, is often used to characterize organic soils. Peatland soils with a thick layer of strongly decomposed acidic organic material, 70 cm thick, with continuous rock at 80 cm and in an environment with a large excess of precipitation qualify as Histosols (FAO 2015). The total extent of Histosols in the world is estimated to be 3.25–3.75 million km$^2$. In comparison, the global peatland area has been estimated at 4.23 million km$^2$ based on the improved global peatland map PEATMAP (Xu et al. 2018). Major peatland complexes are located in the circum-arctic zone, particularly the Western Siberian Lowlands in Russia, and the Hudson and James Bay Lowlands in Canada. Other important concentrations at lower latitudes include extensive peat-dominated wetland or swamp forest landscapes such as the Congo and Amazon Basins, and those of Southeast Asia (Xu et al. 2018). However, estimates are uncertain as there is a basic lack of consistent peatland mapping at the global scale (Biancalani and Avagyan 2014). Further, global wetland and soil datasets are poorly suited to estimating peatland distribution (Xu et al. 2018). Thus, previous global peatland inventories likely underestimated peat extent in the tropics and overestimated it in parts of mid- and high-latitudes of the Northern Hemisphere.

Large peatland areas are found with estimated 1.38 million $km^2$ in Russia, 1.13 million $km^2$ in Canada, 0.27 million $km^2$ in Indonesia, and 0.22 million $km^2$ in the USA (Joosten 2010). Based on PEATMAP, different numbers were reported with 0.15 million $km^2$ for Indonesia and 0.20 million $km^2$ for the USA (Xu et al. 2018). Recently, Tanneberger et al. (2017) provided the most accurate representation of current peatland distribution across the whole of Europe that was possible on the basis of available national data, using a consistent definition of peatland. The overall area of peatland in Europe was estimated at 593,727 $km^2$ (528,337 $km^2$ based on PEATMAP; Xu et al. 2018). Mires, i.e., peatlands where peat is being formed, covered >320,000 $km^2$ (54% of the peatland area). If shallow peatlands (<30 cm peat) in European Russia were also taken into account, the total peatland area in Europe was >1,000,000 $km^2$, which was almost 10% of the total surface area (Tanneberger et al. 2017).

Aside from the northern peatlands, other peatland-rich areas include the Amazon River Basin, Congo, Indonesia, the Tibetan Plateau, and southern Patagonia with peat often occurring under swamp forests (Dargie et al. 2017; Loisel et al. 2017). For example, lowland peatlands in Southeast Asia may cover 0.248 million $km^2$ (Biancalani and Avagyan 2014). However, unprecedented extents and volumes of peatland in the tropics (1.7 million $km^2$ and 7,268 (6,076–7,368) $km^3$) were reported by Gumbricht et al. (2017) based on an expert system model. South America and not Asia may contribute the most to tropical peatland area and volume (44% for both), partly related to some yet unaccounted extended deep deposits but mainly to extended but shallow peat in the Amazon Basin. About 38% of both tropical peat area and volume may be located in Asia with Indonesia having the deepest and most extended peat areas in the tropics. Further, Africa may have more peat than previously reported but climatic and topographic contexts leave it as the least peat-forming continent. The results suggest large biases in current understanding of the distribution, area, and volumes of tropical peat and their continental contributions (Gumbricht et al. 2017).

Soil information systems are the most complete source of georeferenced information on the occurrence of organic soils for large areas at the continental or global scale. For example, the majority of Histosols are located in the boreal, subarctic, and low arctic regions of the Northern Hemisphere. Only one-tenth of all Histosols are found in the tropics. Extensive areas of Histosols occur in the US and Canada, Western Europe and northern Scandinavia and in the West Siberian Plain (FAO 2015). The peatland C balance will be discussed in more detail in the following section as peatland soils dominate wetland soil C storage.

## 5.2 Peatland Carbon Balance

Globally, most soils are unsaturated and oxic, so $CO_2$ is the main respiration flux (Gougoulias et al. 2014). However, in waterlogged anoxic or hydromorphic soils of peatlands, $CO_2$ is reduced to $CH_4$ by hydrogenotrophic archaea in methanogenesis. The net flux of $CH_4$ produced depends on the relative activity of methanogens versus the activity of aerobic $CH_4$-oxidising bacteria (methanotrophs) residing in the surface, oxic layers of peatland soils. Further, the microbial anaerobic oxidation of $CH_4$ in anoxic layers of peatlands may also contribute to the $CH_4$ balance (Gougoulias et al. 2014). However, the current paradigm that microbial methanogenesis can only occur in anoxic habitats has been challenged. For example, Angle et al. (2017) reported $CH_4$ production in well-oxygenated soils of a freshwater wetland. In this wetland, up to 80% of $CH_4$ fluxes could be attributed to methanogenesis in oxygenated soils.

The drivers of the C balance of peatlands differ across different scales (Fig. 5.1; Limpens et al. 2008). At the local scale, the depth of the unsaturated zone together with vegetation composition is predictors for soil respiration through their effects on the availability of electron donors (i.e., OM) and electron acceptors. At the ecosystem scale, the vegetation composition is controlled by water level, nutrient availability, and pH. At the landscape and regional scales, the percentage of peatlands (land cover) and their connections to other ecosystems through subsurface and surface hydrology and topography affect C export to water and atmosphere. However, full integration of C flux across spatial scales is difficult. Yet, a large part of the drivers is related to hydrology, i.e., surface wetness, depth of the unsaturated layer, water table depth, and water movement. Integrating these aspects into a single parameter (e.g., by calculating water residence times) and relating it to biogeochemical processes and vegetation shifts may be promising. To sum up, the main drivers controlling C fluxes in peatlands are largely scale dependent and most are related to hydrology. Despite high spatial and annual variability in net ecosystem exchange (NEE), the differences in annual NEE are more a function of broad-scale geographic location and physical setting than internal factors. Thus, there may be strong feedbacks. In contrast, $CO_2$ and $CH_4$ emissions may be mainly controlled by local factors (Limpens et al. 2008).

The waterlogged anoxic, cold, and acidic conditions limiting microbial decomposer activity together with plant and peat recalcitrance may be main reasons for accumulation of dead plant material in boreal and temperate peatlands (Rydin and Jeglum 2006). The same factors, except for low temperatures, may contribute to peat accumulation in tropical peatlands. There, trees are the main peat-forming plants, whereas those are less important in northern latitude peatlands. Here, peat

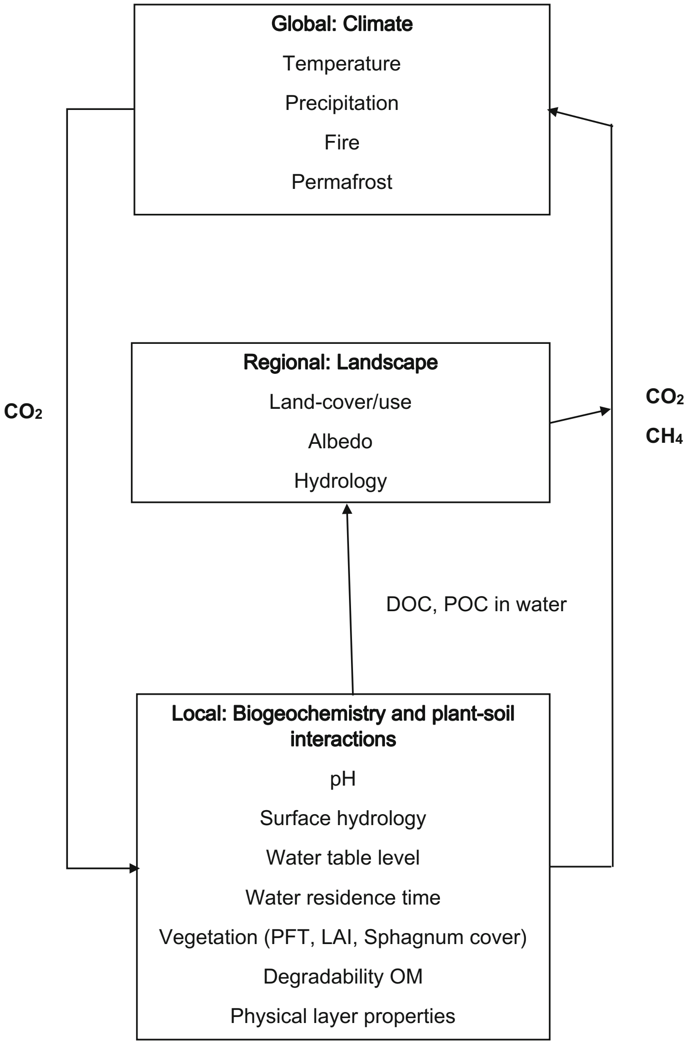


**Fig. 5.1** Biogeochemical and biophysical drivers of the peatland carbon balance at different spatial scales (PAR = Photosynthetically active radiation, DOC = Dissolved organic carbon, POC = Particulate organic carbon, PFT = Plant functional type, LAI = Leaf area index; redrawn from Limpens et al. 2008)

mosses (*Sphagnum*), 'brown' mosses (Amblystegiaceae), and sedges (*Carex*) are the major peat-forming plants (Biancalani and Avagyan 2014). However, plant NPP and peat accumulation rates vary widely among peatlands. For example, the 'efficiency' of peatlands (i.e., the ratio between peat accumulation and NPP) varies between 1 and 20% depending on geographical location, age, and type of the peatland (Biancalani and Avagyan 2014). Gaseous C losses occur by $CO_2$ and $CH_4$

emissions from tropical peatlands, whereas boreal and temperate peatlands are net emitters of $CH_4$ only.

Peatlands dominate the wetland soil C storage and are very efficient at sequestering C relatively slowly over millennia (Loisel et al. 2017). However, the C sequestration rates have been highly variable during the Holocene (the past 12,000 years after the last glaciation) with peak C accumulation rates occurring during warmer climate intervals (Yu 2012). Nevertheless, C sequestration in peatlands during the Holocene is on the same order as the total land C sink with the majority occurring in northern peatlands (Stocker et al. 2017). Current total peat C stocks are estimated at 547, 50, and 15 Pg C for northern, tropical, and southern peatlands, respectively (Yu 2011). Peatland C stocks are much higher than those for wetlands as peatland soils can be much deeper than wetland soils. The range of Indonesia's total peat C stock alone is within 13.6 and 40.5 Pg C, with a best estimate of 28.1 Pg C (Warren et al. 2017). This is more than the estimated C stock in above- and belowground biomass of all Indonesian forests. When field measurements from one of the world's most extensive regions of swamp forest, the Cuvette Centrale depression in the central Congo Basin, are included, global tropical peatland C stocks increase substantially to $\sim$105 Pg C (Dargie et al. 2017). Peat layers in tropical regions are typically thicker than those in temperate and boreal peatlands explaining the relatively large SOC stocks in tropical peatlands despite relatively low areal extent (Page et al. 2011). However, similar to the spatial extent and peat volume, there is high uncertainty of C stocks for tropical peatlands (Draper et al. 2014).

Joosten (2010) computed slightly different numbers and estimated that the largest peat stocks are located in Canada (155 Pg C) followed by those in Russia (138 Pg C), Indonesia (54 Pg C), and the USA (29 Pg C). However, the greatest source of uncertainty is the lack or insufficient representation of data, including peat depth, bulk density, and C accumulation data, especially from the world's large peatlands (Yu 2012). For example, using the peat volume approach, the C density approach and the time history approach, estimates of C stock in northern peatland are 180–455, 125–466, and 473–621 Pg C, respectively (Yu 2012). Further, major uncertainty is linked to the area covered by high-latitude peatlands (published estimates vary between 1.2 and 2.7 million $km^2$), which alone results in a range of 94–215 Pg SOC (Köchy et al. 2015). Scarcity of information about SOC at depths >1 m and variation in definitions of 'peatland' need to be addressed, in particular, to improve the estimates of peat C stock. Yu (2012) proposed to include the estimates of C stocks by regions and further utilization of widely available basal peat ages to reduce uncertainty. Improved extent and volume mapping of peat are required to reduce data uncertainty (Robinson et al. 2017). Nevertheless, peatlands cover only $\sim$3% of the global ice-free land area of 130 million $km^2$ (Hooke et al. 2012), but store more than one-fourth of the global SOC stock of 3,000 Pg C (Köchy et al. 2015). Thus, drainage and utilization of peatlands for agriculture may release large amounts of C.

The properties of organic soils of peatlands have major effects on the SOC balance. The major difference in C flow between organic soils and those of drier soils outside of peatlands is that lateral transfer of C with water plays a more prominent role. Peatland C inputs occur by (i) plant photosynthesis; (ii) dissolved inorganic C (DIC) and dissolved organic C (DOC) deposition with rainwater; and (iii) intake of inorganic C from the weathering of underlying strata and lateral inflows of organic and inorganic C from other sites (Biancalani and Avagyan 2014). Peatland C release follows also lateral pathways as C is transported to streams in the form of particulate organic C (POC), DOC, or DIC (Biancalani and Avagyan 2014). Generally, DOC is the main component of the lateral peatland C flux (Roulet et al. 2007). The export of DOC is a key part of the peatland C cycle and has also important implications for downstream water chemistry (Biancalani and Avagyan 2014).

Similar to wetlands, the future of the peatland C stock is uncertain even without agricultural use. During the Holocene, northern peatlands have probably been a persistent atmospheric $CO_2$ sink and a $CH_4$ source, with an overall negative radiative impact or cooling effect on the global climate (Loisel et al. 2017). For example, $CH_4$ emissions from northern peatland are estimated to range from 31 to 106 Tg $CH_4$ $yr^{-1}$ (Zhuang et al. 2004). The future of the large C stock of northern peatland is particularly uncertain under the projected climate change (Loisel et al. 2017). Specifically, there is no consensus on the direction and magnitude of the impact of climate change on the peatland C sink capacity. On the one hand, enhanced peat decomposition and subsequent C emission to the atmosphere may occur in a warmer and drier world. Otherwise, warmer temperatures would prolong the growing season for peatland vegetation and, thus, increase the NPP and C accumulation in peatlands that are not water-limited. However, not well understood are the impact of drought on fire severity, intensity, and recurrence on $CO_2$ emissions and the impact of warming temperature on permafrost thaw and subsequent peatland collapse on $CH_4$ emissions (Loisel et al. 2017). While Ise et al. (2008) simulate quick losses of labile SOC from northern peatlands during dry periods to the expected warming in the twenty-first century, Wang et al. (2015) propose that the projected 'positive feedback loop' between C emission and drought in peatlands may not occur in the long term. Rather, historically accumulated SOC stock in northern peatlands will potentially be protected by a vegetation shift from low-phenolic *Sphagnum*/herbs to high-phenolic shrubs. However, the functional identity and functional redundancy of European peat bog communities as a whole remain unchanged (Robroek et al. 2017). This strongly suggested that species turnover across environmental gradients is restricted to functionally similar species. Thus, plant taxonomic and functional turnover are decoupled, which may allow these peat bogs to maintain ecosystem functioning including C sequestration when subject to future climate change (Robroek et al. 2017).

The effects of permafrost thawing on peatland C accumulation are uncertain (Limpens et al. 2008). The SOC in permafrost may be protected by microscale structures and higher aggregate stability (Mueller et al. 2017; Oztas and Fayetorbay 2003). Thawing may accelerate the decomposition of this previously frozen and protected SOC pool (Walz et al. 2017). However, the amount of C released and whether C will be released as $CO_2$ or as $CH_4$ will largely depend on in situ thaw and hydrological conditions. For example, anaerobic environments created by melting ice may replace freezing temperatures as a mechanism for SOC stabilization, keeping $CO_2$ and $CH_4$ emissions lower than they would otherwise be (Zona et al. 2012). Otherwise, the water released by melting ice may increase water residence time, promote peat formation and, thus, local C accumulation but increase $CH_4$ emissions (Limpens et al. 2008). For example, the C sequestration rate of collapse scars is among the highest reported for peatlands (Meyers-Smith et al. 2008). Otherwise, if the permafrost layer acts as a water impermeable layer, thawing will lead to drainage and stimulate decomposition processes. Further, in peatlands with discontinuous permafrost, severe fire events may contribute to permafrost thawing leading to more permanent vegetation changes, potentially increasing C accumulation in the long term (Limpens et al. 2008). Chaudhary et al. (2017) applied a model accounting for feedbacks between hydrology, peat properties, permafrost, and dynamics of vegetation across a heterogeneous peatland landscape. This model was able to reproduce broad, observed patterns of peatland C and permafrost dynamics across the pan-Arctic region. Under a business-as-usual future climate scenario, Chaudhary et al. (2017) showed that non-permafrost peatlands may become a C source due to soil moisture limitations, while permafrost peatlands gain C due to an initial increase in soil moisture, which suppresses decomposition while enhancing plant production.

Tropical peatlands may increasingly lose C by fire following drought periods. As fire frequency is expected to increase with climate change, understanding the interactive effects of altered hydrology and fire on SOC and GHG emissions from peatlands is crucial (Keller and Medvedeff 2016). In the following section, the consequences of utilization of wetland soils for agriculture will be discussed.

## 5.3 Management and Use of Wetland Soils for Agriculture

Soils of wetlands and, in particular, peatlands contain large amounts of C. Thus, their conversion for agriculture and intensive agricultural use such as high degree of soil amelioration, drainage, and fertilizer use may result in a significant increase of the atmospheric radiative forcing (Petrescu et al. 2015). For example, total net emissions by peatland use have been estimated at 6 Pg C during the period 1850–2015 (Houghton and Nassikas 2017). Any management plans for these complex ecosystems should, therefore, carefully account for the potential biogeochemical effects on climate.

Aside from utilizing wet and peatlands as croplands and pastures, wetland SOC and peat have been used historically by local communities for many purposes, including as growing media and soil improvers, building material, livestock bedding, and fuel for generating heat and energy (Biancalani and Avagyan 2014). Naturally forested peatlands are also a valuable source of timber. However, when disturbed by agricultural practice, the large peat C stock accumulated over millennia may rapidly subside. Peat C is lost, in particular, because agricultural land use alters the balance between C inputs of dead plant OM to the soil, and the mineralization and export of SOC (Biancalani and Avagyan 2014). Today, $\sim$0.26

million $km^2$ of organic soils is utilized for agriculture, i.e., 0.18 million $km^2$ for cropland and 0.08 million $km^2$ for pastures (FAO 2014). Between 2000 and 2006, the area of organic soils being cropped increased across EU-25, but no further increase occurred after 2006 (Robinson et al. 2017).

Large areas of flooded mineral soils, especially in Asia, are under rice cultivation. Globally, the wetland category 'Rice Paddy and Field' covers 2.14 and 1.06 million $km^2$ in tropical regions (Köchy et al. 2015). SOC stock estimates to 1 m depth are 17.1 Pg C globally with tropical wetland soils under flooded rice storing 8.4 Pg C. Rice paddies are a substantial $CH_4$ source and have emitted 33–40 Tg $CH_4$ $y^{-1}$ for the decade of 2000–2009 (Ciais et al. 2013). Bridgham et al. (2013) reported a median value of 53 Tg $CH_4$ $y^{-1}$ based on both bottom-up and top-down methods. Rice paddies share the same fundamental set of controls over $CH_4$ emissions as natural wetlands. However, the dominant groups of methanogens in rice paddies differ from those observed in natural sites. Specifically, biogenic $CH_4$ is produced in rice fields by anaerobic methanogenic archaea (Alpana et al. 2017). Variations in the ecophysiology of methanogens are likely factors accounting for differences in $CH_4$ production (Bridgham et al. 2013). Rice paddies emit also $N_2O$ as plant residue-N and N fertilizer application enhance $N_2O$ emissions. In the year 2000, 42–122 Gg $N_2O$ was emitted from flooded rice soils (Gerber et al. 2016).

### *5.3.1 Wetlands*

Wetlands do not necessarily be drained for agricultural use (Mitsch and Gosselink 2015). For example, some use is made of more or less undisturbed wetlands such as harvesting salt marsh hay (*Spartina patens*) as bedding and fodder for cattle. Further, there is a renewed interest in the ancient Mexican practice of marceno (Mitsch and Gosselink 2015). Specifically, during the dry season, small areas in freshwater wetlands are planted in corn (*Zea mays* L.) tolerant to flooding. After harvest, the marshes are naturally flooded, and native grasses reestablished until the next dry season (Mitsch and Gosselink 2015). Major examples of agricultural uses of wetlands are crop fields on river floodplain soils and rice fields (Verhoeven and Setter 2010). Some of the agricultural wetlands may be converted to some degree but maintain a modified range of ecosystem services (Ramsar Convention on Wetlands; FAO; International Water Management Institute 2014). Examples are small seasonal wetlands in Africa, rice paddies, and coastal grazing marshes. However, the use of salt marshes for livestock production affects multiple ecosystem properties, creating trade-offs and synergies with other ecosystem services (Davidson et al. 2017).

Floodplains in many parts of the world are managed by flood recession agriculture. The higher parts of the floodplains are highly suitable for growing crops, while the lower parts are wetter but are often suitable for grazing (Verhoeven and Setter 2010). Floodplain soils are nutrient-rich and are naturally 'fertilized' as a result of flooding events. Other wetlands are dependent on continued agricultural activities to maintain their ecological character, such as mowing and grazing in wet grasslands (Ramsar Convention on Wetlands; FAO; International Water Management Institute 2014). Some wetlands are maintained in a natural state for production and harvesting of specific products such as wild rice (*Zizania* spp.) in the USA. Other wetland systems are constructed or managed solely for agricultural purposes such as cranberry (*Vaccinium macrocarpon*) bogs in North America (Ramsar Convention on Wetlands; FAO; International Water Management Institute 2014). In the long term, however, intensification of land use with high levels of fertilizer and pesticide use may result in non-sustainable wetland agriculture (subsidence) and be devastating for wetland biodiversity such as observed in the Dutch wetlands (Verhoeven and Setter 2010).

Some more or less undisturbed wetlands are also used for aquaculture, the farming of fish and shellfish (Mitsch and Gosselink 2015). Aquaculture is the fastest growing food sector and continues to expand with an annual rate of 7.8% worldwide between 1990 and 2010 (Troell et al. 2014). Currently, aquaculture provides half of the fish consumed worldwide. However, aquaculture is increasingly dependent on terrestrial crops and wild fish for feeds, draws on freshwater and land resources for a large portion of its aggregate production, and can be damaging to aquatic ecosystems and fisheries. Thus, it is unclear whether aquaculture adds resilience to the global food system (Troell et al. 2014).

A total of 0.255 million $km^2$ of organic soils has been drained for agriculture, with about 60% of total drained organic soils in boreal and temperate cool, 34% in tropical, and 5% in warm temperate regions (Tubiello et al. 2016). Further, more than 90% of drained organic soils are under cropland. Total GHG emissions are 0.91 Pg $CO_2$ eq. $y^{-1}$ with 0.86 Pg $CO_2$ eq. $y^{-1}$ from drained cropland and 0.05 Pg $CO_2$ eq. $y^{-1}$ from drained grassland. Emissions of $CO_2$ alone represent more than 85% of the total of $CO_2$ and $N_2O$ emissions (Tubiello et al. 2016). Globally, $CO_2$ emissions from drained organic soils represented one-third of net $CO_2$ emissions from the 'agriculture, forestry, and other land-use' (AFOLU) sector, on average over the period 1990–2010. Further, the $CO_2$ component alone represented more than one-fourth of the net $CO_2$ emissions from the AFOLU sector. A few key countries, largely in Southeast Asia, make the largest contribution to the global trends as rates of C loss are much higher under tropical climates because biological decomposition is a temperature-dependent process (Tubiello et al. 2016).

### *5.3.2 Peatlands*

In their natural state, peatlands capture C as $CO_2$ but may also act as $CO_2$ sources in some years (Strack 2008). In contrast, agricultural peatlands commonly act as sources for C and GHGs (Schrier-Uijl et al. 2014). Contemporary agriculture techniques, in particular, heavily impact peatlands through land clearance, drainage, and fertilization, a process that too often involves fire (Wijedasa et al. 2017). When only slightly disturbed, anthropogenic modification of peatlands may result in reduced biomass production, resulting in a decreased input of OM to the peat C store (Biancalani and Avagyan 2014). For example, grazing of temperate peatlands by livestock and the production of hay may lead to changes in the composition of plant species and affect C inputs. Overgrazing, in particular, contributes to degradation of the Qinghai–Tibetan Plateau peatlands in China (Yang et al. 2017). Here, soil drainage facilitated earthworm invasion and subsequent SOC loss (Wu et al. 2017).

Compared to grazing, crop cultivation is a more intensive form of peatland use that disturbs the ecosystem to much greater extent (Biancalani and Avagyan 2014). For example, 80% of all Indonesian peatlands are less than 300 cm deep and are, thus, allowable for conversion under current regulations (Warren et al. 2017). The past, ongoing, and eventual conversion of these shallower peatlands may release 10.6 Pg C to the atmosphere assuming total peat loss. Drastic changes occur also by conversion of tropical peatlands for oil palm (*Elaeis guineensis* Jacq.) cultivation in Southeast Asia. Palm oil is used for bioenergy among other things. Converting peatlands to produce food crop-based biofuels create a 'biofuel carbon debt' (Fargione et al. 2008). For example, converting tropical peatland rainforest in Indonesia and Malaysia to palm biodiesel would result in a biofuel carbon debt of 3,450 Mg of $CO_2$ $ha^{-1}$ that would take 423 years to repay. The required drainage of peatland causes a sustained emission of 55 Mg of $CO_2$ $ha^{-1}$ $yr^{-1}$ from oxidative peat decomposition. Until the carbon debt is repaid, producing and using palm biodiesel from this land would cause greater GHG release than would refining and using an energy-equivalent amount of petroleum diesel (Fargione et al. 2008).

Drainage changes peatlands from long-term C reservoirs to net sources of GHG emissions (Wijedasa et al. 2017). Drainage results in a rapid increase in decomposition rates, leading to increased emissions of $CO_2$, and some $N_2O$ from microbial breakdown of peat, and vulnerability to further emissions of $CO_2$, CO, $CH_4$ and black carbon (BC) emissions by fire (Smith et al. 2014). For example, in 1997, about 24,000 $km^2$ of peatland was burned in Southeast Asia, which released an estimated 0.81–0.95 Pg C (Page et al. 2002). Further, during the summer of 2010, an extreme period of high temperatures and low rainfall resulted in widespread and prolonged fires on abandoned drained peatlands in the Russian Federation (Biancalani and Avagyan 2014).

The type of land use, including the management type, and not SOC content may be a major control of $CO_2$ emission from drained peatland soils. For example, lower NEE values (−6 to 1707 g $CO_2$–C $m^{-2}$ $yr^{-1}$) were found at arable sites and higher values (1354–1823 g $CO_2$–C $m^{-2}$ $yr^{-1}$) at grassland sites on drained organic soils independent of differences in SOC contents (Eickenscheidt et al. 2015). Further, cultivated peatlands are not necessarily net C sources to the atmosphere as were shown in a study from Sweden (Hadden and Grelle 2017). Young and old SOC may respond differently to peatland use. For example, in strongly degraded organic soils, drained and managed since decades, fresh OM additions suppressed the decomposition of old SOC, i.e., peat (Bader et al. 2018). Fresh OM addition induced positive SOC decomposition in grassland soils, but no priming effects were observed for cropland soils (Bader et al. 2018).

There is a clear relation between $CO_2$ emissions and the water table depth below the peat surface. Emissions increase proportionally as water tables are lowered below 0.5 m (Biancalani and Avagyan 2014). The $CO_2$ emission factors are the highest for drained tropical peatlands and as high as 14 Mg C $ha^{-1}$ $y^{-1}$ for croplands and 9.6 Mg C $ha^{-1}$ $y^{-1}$ for grassland (IPCC 2014). Hooijer et al. (2012) estimated $CO_2$ emissions of >73 Mg C $ha^{-1}$ $yr^{-1}$ from tropical peatlands under plantation agriculture. In total, 355–855 Mg $CO_2$ may be released annually more than 5 years after drainage of tropical peatlands in Southeast Asia (Hooijer et al. 2010). Emissions are predicted to rapidly increase until at least 2020, and emissions from fire and decomposition in the first years after drainage may be of similar magnitude (Biancalani and Avagyan 2014).

In contrast to other GHGs, $CH_4$ emissions from drained peat soils may be absent (Biancalani and Avagyan 2014). However, drainage ditches and canals typically represent hot spots for $CH_4$ emissions. Hot spots may also occur at interfaces between different microtopographic units and may be responsible for the bulk of a site's $CH_4$ emissions. Drainage also causes an increase in $N_2O$ emissions from peat decomposition. The application of fertilizers contributes to increasing the overall negative balance of GHG emissions from drained peatlands. $N_2O$ fluxes from agricultural peatlands may be dominated by emission pulses which are often observed after fertilization or thaw events. Generally, there is no thorough understanding of the magnitudes and drivers of $N_2O$ emissions (Butterbach-Bahl and Wolf 2017). However, in terms of $CO_2$ equivalents, peatland $N_2O$ emissions are much lower than $CO_2$ or $CH_4$ emissions (IPCC 2014).

About a quarter of the global peatland area drained for agriculture is located in the former Soviet Union (Biancalani and Avagyan 2014). Drained fen peatlands have provided some of the most fertile soils for crops and support high levels of production. In undrained fens, precipitation exceeds evapotranspiration. In contrast, precipitation equals evapotranspiration in natural bogs. Drained bogs have problems of high acidity and waterlogging producing low yields of grasses and crops. Thus, the agricultural use of peatlands has changed over time with many areas having been abandoned. For example, while peatland drainage and conversion to agriculture have virtually ceased in boreal and temperate zone countries, it is increasing in the tropics. The palm oil and paper industries, in particular, are fueling rapid agricultural development of tropical peatlands. In contrast, paludiculture is now receiving increasing attention (Joosten et al. 2015). Paludiculture uses biomass from wet and rewetted peatlands under conditions that maintain the peat mass, promotes peat accumulation, and provides some of the ecosystem services (ESs) associated with natural peatlands (Biancalani and Avagyan 2014).

To sum up, C stocks in tropical peatland are particularly affected by drainage as tropical areas account for 34% of all drained organic soils, which is about three times greater than their share of total organic soils (Biancalani and Avagyan 2014). Besides the effects on GHG emissions, drained peatlands also release more DOC than undrained peatlands (Moore et al. 2013). As older C from deeper in the peat profile is also mobilized, drainage severely diminishes C sequestration and climate change mitigation in peatlands as a C that has accumulated over millennia is returned to the atmosphere. Any management practices that lower the water table lead to losses of C and N from peatlands (Biancalani and Avagyan 2014).

The subsidence components and their impacts on oxidation, compaction, shrinkage, and consolidation need to be addressed separately to assess drainage effects on the peatland C balance (Hooijer et al. 2012). As build-up of peat generally requires that water levels be near the soil surface, drainage is required to allow biomass production by non-adapted vegetation such as major upland crops. The drop in the water table ranges from 0.4 m for grasslands to 1.2 m for crop production. Following lowering the water table, the soil surface subsides by consolidation as the soil moisture content declines and the peat volume contracts. While the peat surface may fall up to 1 m in the first year, the rate of surface lowering varies afterward from 1–2 cm per year for temperate peatlands to 3–5 cm per year for tropical peatlands drained for agricultural production (Biancalani and Avagyan 2014). For example, apparent subsidence rates for the Willard muck agricultural area in Ohio ranged from 33 to 216 cm over a 40-year period (Hidlebaugh 1982). Elder and Lal (2008) estimated subsidence rates (cm $yr^{-1}$) of 1.34 under moldboard plow, 1.07 under no-till and 1.25 under bare management in the same region. Thus, within 214–268 years, the deepest organic soil deposits will be completely depleted assuming current subsidence rates. Further, all tillage treatments were persistent sources of $CO_2$ of 18.9–22.5 Mg $CO_2$–C $ha^{-1}$ $yr^{-1}$, but emissions did not differ between treatments (Elder and Lal 2008). Effects of tillage practices on emissions of GHGs from organic soils are generally less well studied.

When drainage exposes underlying mineral deposits unsuitable for agriculture, agricultural production in some peatlands may end (Biancalani and Avagyan 2014). Some examples of improved management of agricultural wetlands are discussed in the following section.

## 5.4 Improved Management of Wetland Soils

Wetlands have been reclaimed for agriculture in many parts of the world with ever more effective drainage and land amelioration measures (Verhoeven and Setter 2010). Much of their original character may have been lost leading to reduced biodiversity and reduced performance of functions other than crop productivity. In contrast, extensive agriculture, i.e., activities in a more or less intact wetland ecosystem without the use of machinery and chemicals, may have less severe effects than intensive agricultural use of wetlands (Verhoeven and Setter 2010). Emissions of $CH_4$ from paddy fields, for example, can be reduced by improved management. Specifically, increased availability of mineral fertilizer contributed to a decrease in use of organic amendments in paddy rice fields, and this resulted in a decrease in $CH_4$ emissions (van der Gon 1999). Reductions in water use contributed also to a reduction in $CH_4$ emissions (Frolking et al. 2004). Managed peatlands can also be turned back into GHG and C sinks within 15 years of abandonment and rewetting if appropriate measures (i.e., reductions in management intensity) are taken (Schrier-Uijl et al. 2014).

Drainage of organic soils results in soil subsidence and contributes to increased atmospheric $CO_2$ concentrations as OM decomposition is accelerated. Rewetting drained peatland soils is considered an important climate change mitigation tool to reduce GHG emissions and create suitable conditions for SOC sequestration. However, knowledge about the exchange of $CO_2$ and $CH_4$ following rewetting during restoration of disturbed peatlands is limited. Restoring drained agricultural land-use types to flooded conditions is among the options to reduce C losses from drained peatlands. For example, converting a drained agricultural area in the Sacramento-San Joaquin Delta, California, back to flooded conditions for rice agriculture created an atmospheric sink for $CO_2$ (Knox et al. 2015). Simultaneously, $CH_4$ emissions from the rice paddy increased. Although the rice paddy was a net C source, it lost 230–454 g C $m^{-2}$ $y^{-1}$ less than the drained agricultural peatlands under corn and pasture. However, wetland restoration provided the most GHG benefit, with the potential of converting drained peatlands from GHG sources to GHG sinks (Knox et al. 2015). Emissions for a rewetted industrial cutaway peatland in Ireland were −104 g $CO_2$–C $m^{-2}$ $yr^{-1}$ (i.e., $CO_2$ sink) and 9 g $CH_4$–C $m^{-2}$ $yr^{-1}$ (i.e., $CH_4$ source), while $N_2O$ emissions were not detected nearly a decade after rewetting (Wilson et al. 2016). However, although the GHG balance was reduced noticeably (i.e., less warming) in comparison with a drained site, it was still higher than comparative intact peatland sites. In contrast, during the eighth year following rewetting, a restored disturbed peatland ecosystem in British Columbia, Canada, was almost $CO_2$ eq. neutral ($CO_2$ eq. (g) = −103.1 g $CO_2$ eq. $m^{-2}$ $yr^{-1}$) over a 100-year time horizon (Lee et al. 2017). To sum up, disturbed peatlands may become net annual C sink after restoration by rewetting. However, long-term monitoring of new rewetted ecosystems is needed to provide critical information for land managers, policymakers, and other stakeholders (Wilson et al. 2016).

The major cereal, legume, and fiber crops are domesticated dryland plants (Biancalani and Avagyan 2014). Cultivating them in wetlands is only possible after drainage which severely affects C sequestration and other ecosystem services. In contrast to conventional drainage-based agriculture, low-intensity agricultural activities in wetlands may be associated with a high biodiversity and other ecosystem services (Verhoeven and Setter 2010). Many traditional agricultural systems have resulted in a very diverse landscape with high species densities in a human-created setting. Examples are the moist grasslands and herbaceous fens in Europe, which have been used for grazing and hay-making. Salt marsh grazing management can be based on local context and desired ecosystem services (Davidson et al. 2017). For example, grazing leads to reductions in blue C, i.e., long-term C storage in coastal vegetated habitats in the Americas but not in Europe. Further, grazing may compromise coastal protection and the provision of a nursery habitat for fish while creating provisioning and cultural benefits through increased wildfowl abundance (Davidson et al. 2017). Rice has traditionally been cultivated in strongly modified landscapes with rice paddies on hill slopes or lowlands with a human-controlled water regime (Verhoeven and Setter 2010).

Another strategy is to make major crop species suitable for growth in wetland environments (Verhoeven and Setter 2010). A range of crop varieties is known that have better waterlogging tolerance than the regular cultivars. However, waterlogging tolerance needs to be further enhanced in new cultivars by crop breeding. In the future, more flood-tolerant and salt-tolerant crops may be cultivated in wetlands (Verhoeven and Setter 2010).

Paludiculture has the cobenefit of C sequestration (Joosten et al. 2015). Spontaneous vegetation on natural sites is used for paludiculture, or crops are artificially established on rewetted sites. Peat is generally formed by roots and rhizomes in the temperate, subtropical, and tropical zones, and peatlands hold vegetation of which aboveground parts can be harvested without substantially harming peat conservation and formation (Joosten et al. 2015). Traditional agricultural products such as food, feed, fiber, and fuel are derived from paludiculture. For example, common reed (*Phragmites australis* (Cav.) Trin. ex Steud.) may be used as construction material, for paper production and as bioenergy feedstock. Cattails (*Typha* spp.) and reed canary grass (*Phalaris arundinacea* L.) may also be feedstock for bioenergy production. In addition, reed canary grass provides fodder and can be used for pasture and the production of silage and hay. Further, peat moss (*Sphagnum* spp.) can replace fossil peat in horticultural growing media. Aside temperate peatlands, opportunities may also exist for paludiculture in degraded Indonesian peatlands. However, more research is needed on the suitability of paludiculture to replace drainage-based agriculture. Similar, the potential to establish sustainable grazing practices on peatlands is less well known (Biancalani and Avagyan 2014). Paludiculture can often compete effectively with drainage-based peatland agriculture. However, technical and political constraints hamper large-scale implementation of this type of peatland use (Joosten et al. 2015).

## 5.5 Conclusions

Wetlands are characterized by high C density similar to permafrost soils. Wetlands have been used by humans for growing crops and grazing animals for millennia. Due to the anoxic conditions, large amounts of C have accumulated particularly in peatland soils as decomposition rates are greatly reduced. Thus, draining wetlands and converting them for growing crops under non-flooded conditions releases large amounts of C and enhance climate change, and this may be exacerbated by increasing incidence of fire. Wetland $CH_4$ emissions will increasingly contribute to the projected climate change. Otherwise, climate change may further enhance wetland C losses. Thus, less intensive agricultural practices such as paludiculture should be promoted to reduce C losses while benefiting from other ecosystem services of wetlands.

## 5.6 Review Questions

1. Describe traditional agricultural practices in wetlands.
2. What are the principal differences between wetlands and peatlands?
3. How can wetlands and their use be adapted to climate change?
4. Biochar has been proposed to replace peat in horticultural media—describe the differences between both and the challenges to realize this potential.
5. Describe 'wetland-friendly' agricultural practices.
6. What are the major challenges in plant breeding for adapting crops to wetland conditions?
7. Rewetting of wetlands has been proposed as a climate change mitigation strategy—discuss its potential in relation to 'negative emission technologies'. What are the implications for the three dimensions of sustainability?

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