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Management of organic soils to reduce soil organic carbon losses

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1 Introduction

Peatlands are ecosystems where soil organic matter (SOM) forms under water-saturated conditions, leading to anoxic conditions, which slow down microbial decomposition processes. The corresponding organic soils, Histosols, are important soil organic carbon (SOC) stores that accumulated 600 Pg SOC during the Holocene (Yu et al., 2010). Organic soils spread over almost 500 Mha worldwide, of which the majority, around 400 Mha, is situated in boreal and temperate regions, and 40–70 Mha is located in the tropics (Frolking et al., 2011; Leifeld and Menichetti, 2018; Page et al., 2011). Correspondingly, these soils have high SOC densities of often 1000–2000 t SOC ha^{-1} (Leifeld and Menichetti, 2018; Page et al., 2011). The quasi-continuous SOC accumulation rate of intact peatlands is estimated at 0.22 (northern) and 0.45 (tropical) t C ha^{-1} yr^{-1} (Leifeld et al., 2019; Loisel et al., 2014), resulting in a net accumulation of annually 0.41–0.51 Pg CO_2-eq, mostly as SOC (Gallego-Sala et al., 2018; Leifeld et al., 2019). As a result of differences in accumulation rates and attributed areas, about 75% of total SOC in Histosols is stored in boreal and temperate peatlands, and 25% in tropical regions.

SOM accumulation under anoxic conditions has trade-offs, as it implies redox processes that lead to the formation of methane, a greenhouse gas, which is, over a 100 years' time scale, around 28 times more potent per kg than CO_2 (Myhre et al., 2013). Indeed, current methane emissions from peatland ecosystems are estimated at 0.03 Pg CH_4 yr^{-1} (0.84 Pg CO_2-eq. yr^{-1}; Frolking et al., 2011). This is similar to the fluxes attributed to global emissions from paddy rice production (Chapter 17 of this book) and corresponds to 20% of

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the CH_4 flux from all natural wetlands (Saunois et al., 2020). Despite of being a substantial methane source, the long-term GHG effect of intact peatlands is negative with a global cooling of -0.2 W m^{-2} to -0.5 W m^{-2}, mostly owing to the long-lasting C sink and the much longer atmospheric lifetime of CO_2 over CH_4 (Frolking and Roulet, 2007). Emissions of nitrous oxide, the third important biogenic GHG, are very small in intact peatlands but raise after drainage.

Glossary of peatland terms.

Peat	Fibric organic sedentarily accumulated material with virtually all of the organic matter allowing the identification of plant forms; consists of at least 30% (dry weight) of dead organic material (Parish et al., 2008).
Peatland	An area with or without vegetation with a naturally accumulated peat layer at the surface of at least 30 cm depth (Parish et al., 2008).
Mire	Synonymous with any peat-accumulating wetland. A peatland where peat is currently forming and accumulating (Parish et al., 2008).
Bog	Mire raised above the surrounding landscape and only fed by precipitation (Parish et al., 2008).
Fen	Peatland receiving inflow of water and nutrients from the mineral soil. Distinguished from swamp forest by a lack of tree cover or with only a sparse crown cover (Parish et al., 2008).
Swamp	Usually forested minerotrophic peatland (minerotrophic peatland receiving nutrients through an inflow of water that has filtered through mineral soil). Separate from wooded fens due to a denser tree canopy. Also called peat swamp forest (Parish et al., 2008).
Acrotelm	Upper peat producing layer of mire with a distinct hydraulic conductivity gradient in which water level fluctuations and most of horizontal water flow occur (Parish et al., 2008).
Catotelm	The lower permanently water-saturated layer in a peatland, with relatively low hydraulic conductivity and rate of decay (Parish et al., 2008).
Mesotelm	The intermediate zone, through which carbon is transferred from acrotelm to catotelm (Clymo and Bryant, 2008).
Blanket Bog	Bog in a very humid climate, which forms a blanket-like layer over the underlying mineral soil (Parish et al. 2008).
Cutover	Peatland area where peat was extracted, but where still economically useful peat is left (Price and Ketcheson, 2009).

Peat extraction	The excavation and drying of wet peat and the collection, transport, and storage of the dried product (Parish et al., 2008).
Wetland	Areas of marsh, fen, peatland, or water, whether natural or artificial, permanent or temporary, with water that is static, flowing, fresh, brackish, or salt, including areas of marine water, the depth of which at low tide does not exceed 6 m (definition according to Ramsar convention).
Histosol	Histosol is a soil type that comprise soils formed in organic material accumulating as groundwater peat (fen), rainwater peat (raised bog) or mangroves (WRB, 2015).

Climatic factors, particularly temperature and moisture, mostly controlled the rates of peat accumulation since the last glacial maximum (Charman et al., 2015). With ongoing climate change, it is discussed controversially whether intact peatlands will continue being a C sink throughout the twenty-first century. Gallego-Sala et al. (2018) argued that the net CO_2 uptake of peatlands is going to slightly increase during the next decades, while Huang et al. (2021) found that, owing to lowered water tables with a warming climate, peatlands will release more CO_2 by 2100 through microbial decomposition. In addition, fire-induced losses may increase when peatlands are drained (Turetsky et al., 2015). An expert assessment indicated that ongoing and expected future changes in the peatlands' carbon balance will be more strongly driven by land-use change, fire and permafrost thaw than in the past (Loisel et al., 2021). Land-use change of peatlands towards agriculture, forestry, or peat extraction usually implies drainage of the peat deposit. Drainage for management induces a variety of system changes, of which biodiversity loss, loss of water storing capacity, soil subsidence and compaction, leaching and SOM decomposition owing to peat aeration with subsequent release of CO_2 and N_2O, but reduced CH_4 emissions, are the most important ones.

Losses of SOC and, more generally, changes in the GHG balance of organic soils after drainage are pronounced and well documented (Hooijer et al., 2012; Kasimir-Klemedtsson et al., 1997; Tiemeyer et al., 2020). The area of peatlands drained for management has been estimated at 43–51 Mha, representing 10% of its total area (Joosten, 2010; Leifeld and Menichetti, 2018), of which almost half is situated in the tropics. Because tropical peatlands contribute less than 20% to the total peatland biome, this number indicates that they are exploited disproportionately. In consequence, the global peatland biome, representing managed and unmanaged organic soils, has shifted from a persistent GHG sink to a strong GHG source since 1850. The annual global GHG sink of intact peatlands contrasts with a GHG source from drained and managed organic soils in the order of 0.8–1.91 Pg CO_2-eq. yr^{-1} (Evans et al., 2021; Humpenöder et al., 2020; Joosten, 2010; Leifeld et al., 2019; Leifeld and Menichetti, 2018).

According to the same authors, tropical organic soils managed by agriculture contribute 59–82% to these emissions. By source, CO_2 is the most important GHG emitted from organic soils drained for agriculture, contributing between 75–78% (boreal and temperate) and 88–90% (tropical) to total GHG emissions (i.e. sum of CO_2, CH_4, N_2O; IPCC, 2014). These annual SOC losses from a relatively small area of only 50 Mha are higher as compared to the total SOC loss, which occurred in the last century from 5000 Mha mineral soil under crop- and grassland (estimated at 0.48 Pg CO_2-eq yr^{-1}; Sanderman et al., 2017). Addressing these losses is thus of major importance in the context of the soils' role in climate change.

In this chapter, we give an overview on the assessment of C losses and processes occurring after peatland drainage in different environments. We will also present management strategies to reduce SOC loss and GHG emissions from managed peatlands.

2 Soil properties and carbon dynamic following peatland drainage

2.1 Methods for estimating C loss after drainage

The SOC loss from drained peatlands can be determined by two different approaches: (i) inventory methods and (ii) direct flux measurements (Simola et al., 2012). By integrating the SOC loss over long periods (several years to decades, up to several hundred years), inventory studies deliver robust estimates of long-term changes in SOC stocks (e.g. Krüger et al., 2016; Minkkinen et al., 1999). They incorporate gaseous and aqueous carbon losses as well as erosion. Detailed information on the drainage history is necessary to calculate annual carbon loss rates. Inventory studies can be broadly classified into (a) repeated inventories that allow to compare historic SOC stocks with present day or (b) profile methods, where peat profiles of drained and undrained (reference) sites are compared, following the space-by-time approach. As natural reference sites are often unavailable, the subsoil is – as an approximation – taken as a reference for peat properties before drainage. The assumptions for this approximation are that the subsoil is not affected by drainage and that changes in carbon stock in the topsoil derivate from peat decomposition. Recent annual changes can only be investigated by tracing the gas exchange of the soil or the whole ecosystem. Flux measurements provide insights into processes and drivers of the GHG exchange. Mechanistic understanding of the underlying carbon cycling that is needed for modelling can be gained as well. However, these methods are time and cost-intensive.

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2.1.1 Soil inventory methods

2.1.1.1 Repeated soil sampling

Repeated inventories of SOC stocks can detect carbon losses over time. As the volume of peat is decreasing upon drainage, it is crucial to consider the entire carbon stock down to the mineral subsoil or to compare the carbon stock above a layer of known radiocarbon age. The challenge of this method is to detect a relatively small annual carbon loss within a large carbon stock. Moreover, the small-scale spatial heterogeneity of carbon stock adds uncertainty to this method. Yet, it has been successfully applied in a repeated inventory in Finland where at 37 drained forest sites an average C loss of 7.4 kg C m^{-2} was observed over a 30-year period (Simola et al., 2012).

2.1.1.2 Ash method

The ash method is based on the simplified assumption that the carbon to ash content is constant during peat accumulation. Changes in the ratio are attributed to the decomposition of peat and subsequent loss of SOC. Accordingly, the occurrence of mineral layers – as often found in fens – restricted this approach to organic peatlands without variable mineral input, in the absence of a natural reference site (e.g. Krüger et al., 2016). The method can provide reasonable estimates of past drainage-induced C losses in bogs (e.g. Rogiers et al., 2008; Wüst-Galley et al., 2016) but has been criticized because of large errors due to the fact that conditions for its application are often not matched (Laiho and Pearson, 2016).

2.1.1.3 Bulk density method

The peat surface subsides over time due to physical compaction as well as peat decomposition (Section 2.3.1). As only decomposition leads to a carbon loss, the contribution of this process to the overall soil subsidence has to be determined (Schothorst, 1977). For this method, depth profiles of peat bulk density and carbon concentrations before and after drainage must be available. An example of this approach by Kluge et al. (2008) indicated average losses of 0.7 kg C m^{-2} yr^{-1} for drained organic soils in Germany. Soil subsidence together with changes in soil bulk density were used by Hooijer et al. (2012) to estimate C losses from drained tropical peatlands. More simplified approaches that derive drainage-induced losses just from the difference in actual soil bulk density and carbon content in degraded topsoils and intact deeper peat layers have also been applied (e.g. Leifeld et al., 2011; Egli et al., 2021). Yet, the latter methodology suffers from limitations similar to the ash method (see above). New approaches to monitor soil subsidence with satellite data and modelling offer the opportunity to assess peat carbon loss at a large scale (e.g. Hoyt et al., 2020).

2.1.1.4 Radiocarbon method

Because drainage-induced peat decomposition proceeds from the surface to deeper layers and is accompanied by continuous loss of surface peat, it comes along with an increasing peat age at the surface as compared to the native state because younger material previously situated at the surface of the profile is lost first (Section 2.2). Whereas in an intact peatland the uppermost peat layers are of recent age, strongly degraded peatlands that were drained for many decades reveal topsoil radiocarbon ages of centuries to millennia (Bader et al. 2018b; Leifeld, 2018; Mrotzek et al., 2020), providing strong evidence for the loss of young peat. The age difference of degraded versus intact topsoil has been used to estimate carbon losses from drainage *a posteriori* (Krüger et al., 2016; Leifeld et al., 2018). There are two principal approaches to use ^{14}C for estimating previous C losses. First, the SOC stock above a reference layer of same radiocarbon age can be measured for the drained situation and compared with the stock of a nearby natural peatland to infer SOC losses. This might be hampered by fact that peatland drainage often concerns larger areas and intact reference sites are not available. A second approach uses the age-depth gradient of intact peat below the drained horizon to infer the site-specific SOC accumulation rates of the peatland and to calculate expected SOC stocks that are compared to the actual ones. With this approach, Leifeld et al. (2018) showed that drained European peatlands lost on average 50% of their former C stock, corresponding to 56 kg C m^{-2}. A recent application by Wang et al. (2021) extended the approach to managed organic soils with mineral cover. However, limitations of this approach are due to the dilution of the older topsoil peat by recent assimilates (Leifeld et al., 2021), which leads to an overestimation of the remaining peat if not quantitatively considered, and to varying peat accumulation rates over time.

2.1.2 Net ecosystem carbon balance approach

The net ecosystem carbon balance (NECB) approach is determined by direct measurement of all carbon fluxes entering and leaving the investigated compartment (soil-vegetation to atmosphere). Generally, the data are reported as positive and negative fluxes with negative values indicating a net downward flux (i.e. a gain of carbon from the atmosphere = carbon sink). Positive values indicate a loss (e.g. harvested biomass or through oxidation/decomposition = carbon source). Fluxes include the exchange of gaseous (CO_2, CH_4) and, ideally, waterborne fractions (dissolved organic carbon (DOC), particulate organic carbon (POC), and dissolved inorganic carbon (DIC)) and, in case of a managed system, any carbon import (e.g. organic fertilizer) or export (harvest). The NECB approach is equivalent to the change in SOC storage. However, in systems where substantial biomass is accumulating during a year, such as tree growth in young plantations, it is essential to account for this storage change; otherwise, the SOC loss is underestimated (e.g. Hommeltenberg et al., 2014; Minkkinen et al., 2018). Long-term time series for peatlands generally include not more than several years. In addition, for a complete NECB approach, fluviatile carbon losses have to be quantified, which is complicated and time consuming. As the major part of carbon is released as CO_2, numerous studies do not quantify waterborne carbon losses (e.g. Tiemeyer et al., 2016). Thus, in many peatlands, carbon losses may have been underestimated by this approach. The direct gas exchange between surface and atmosphere can be measured by two different techniques: Eddy-Covariance (EC) and chambers.

2.1.2.1 Eddy-Covariance

EC measures the undisturbed gas exchange of the net ecosystem exchange (NEE), that is, the sum of CO_2 uptake by plants (gross primary production, GPP) and ecosystem respiration (ER) over an area of about 0.5–5 ha. The ER consists of autotrophic and heterotrophic respiration. NEE is partitioned into GPP and ER by a modelling approach. The advantage of EC is that the CO_2, and also CH_4, exchange can be measured continuously in the free atmosphere without disturbing the investigated ecosystem. As there is no limitation of vegetation height, EC can be used for forests as well. However, NEE is integrated over an area, which is variable in its extension and direction. Thus, capturing the small-scale heterogeneity of mosaic structures, as often found in intact peatlands, is impossible. Limitations of the method under specific micrometeorology conditions generate gaps within the continuous data set, which have to be filled. Furthermore, the equipment is expensive, and expert knowledge is needed to perform flux measurements (e.g. Minkkinen et al., 2018; Peacock et al., 2019; Paul et al., 2021).

2.1.2.2 Chambers

The most widely used method to measure gas exchange is the chamber method. Briefly, this method consists of putting a closed chamber on the soil and measuring the gas, which is accumulating over a specific time, often covering an area of 0.01–1 m^2. Gas samples are taken to detect the change in gas concentration *in situ* or offline. The most common chamber systems operate manually, which limits the frequency of flux measurements. Automatic chamber systems can increase the temporal resolution, especially covering

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the whole day and night-time measurements (Wang et al., 2022). However, these systems are expensive and electricity is needed. Using a transparent chamber, NEE can be measured, and opaque chambers, excluding the light, are used for ER. Advantages are that specific vegetation communities and the small-scale variability can be captured; however, the use is restricted to small vegetation fitting within the chamber. Artefacts due to the creation of closed spaces (changes in temperature, humidity, or pressure) may influence the measurements (e.g. Alm et al., 2007; Leiber-Sauheitl et al., 2014).

2.2 Peat profile changes following drainage

Drainage is a prerequisite for the agricultural use of peatlands, which became widespread in the twentieth century, particularly in Europe and North America (Davidson, 2014). Agricultural practices on drained peatlands range from extensively used pastures and highly productive agriculture for food production to forestry. Drainage-induced mineralization transforms the original fibric to hemic structure of the undrained peat to a new material called mursh, which is characterized by a moderate to strongly granular or blocky structure, or the structure is even pulverized (WRB, 2015). Mineralization of peat decreases the organic carbon content and C/N ratio while bulk density increases (Fig. 1). The most severe shift in the pore structure of peat occurs within the first 20 years of drainage (Liu et al., 2020). The decline in total porosity is accompanied by a shift in pore size distribution (Wallor et al., 2018), and the altered pore structure reduces saturated hydraulic conductivity (Kechavarzi et al., 2010). The infiltration rate may decrease while the runoff from drained peatlands may increase, thereby enhancing soil erosion (Li et al., 2018; van Seters and Price, 2002).

Surface desiccation can lead to an increased hydrophobicity of peat (Eggelsmann et al., 1993). A dry surface crust is often developed, which is susceptible to shrinkage and cracking and thus may enhance superficial flooding. Dry peat becomes flammable. Peat fires generally are dominated by smouldering combustion, and these flameless fires can burn for long periods. In natural peatlands, fires occur as well, but deeper peat layers are protected from burning due to a high water table (Turetsky et al., 2015). Drainage and also climate change increase the frequency and extent of peat fires thereby also affecting the mineralization of old carbon (Konecny et al., 2016; Turetsky et al., 2015; Wiggins et al., 2018). In natural peatlands, the age of SOC linearly increases with depth (Fig. 1). Drainage and fire increase the mineralization of the peat, and the age of the carbon at the soil surface increases successively (Fig. 1). The contribution of old peat-derived carbon to total SOM decreases, while the contribution of the fresh litter input from current non-peat vegetation increases over time (Fig. 1).

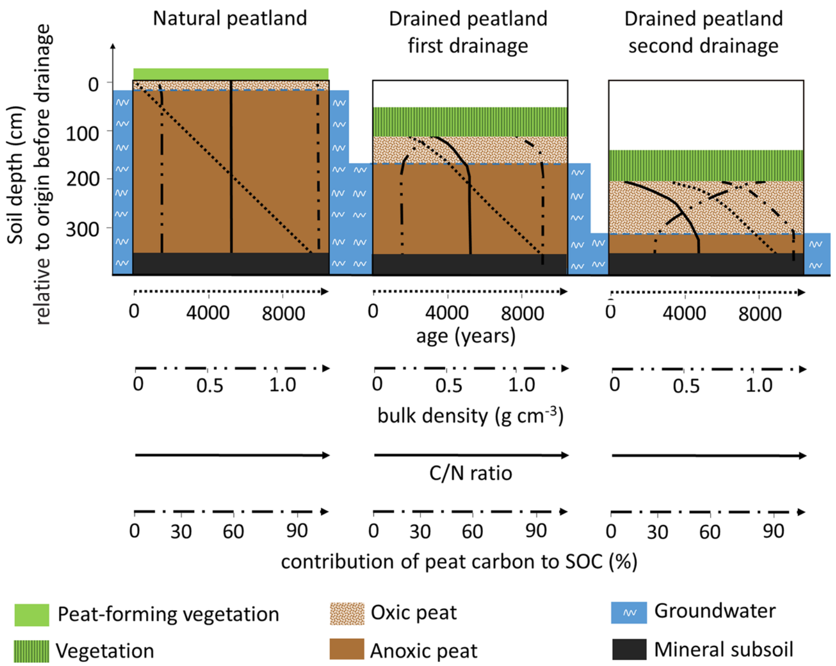

Figure 1 Changes of soil properties through drainage over 100 years in a temperate peatland.

2.3 Soil subsidence and carbon loss from drained peatlands

2.3.1 Soil subsidence

In drained peatlands, SOC loss will continue as long as decomposable organic material is available in the aerated soil horizons. The loss of peat by wind and water erosion, burning, and peat extraction also contributes to the loss of organic material and thus soil subsidence. After 40 years of drainage, Schipper and McLeod (2006) found average subsidence rates of 3.4 cm yr^{-1} in a New Zealand under grassland. The authors attributed about 63% of the soil subsidence to consolidation and shrinkage, with the remainder (37%) attributed to losses of OM by peat mineralization. For longer periods (1000 years), the estimated contribution of decomposition to soil subsidence was 85% (Schothorst, 1977).

After strong initial soil subsidence due to rapid consolidation and shrinkage, the rate of subsidence decreases because peat decomposition processes occur on longer timescales (Fig. 2). For the first 10 years after drainage, average soil subsidence rates of -1.6 ± 1.0 cm yr^{-1} to -6.2 ± 2.2 cm yr^{-1} were found for subarctic/boreal, temperate, and tropical peatlands (Evans et al., 2019; Liu et al., 2020). Subsidence was the strongest in the first 5 years in tropical systems, with rates up to 28 cm yr^{-1} (Hooijer et al., 2012). The primary thickness

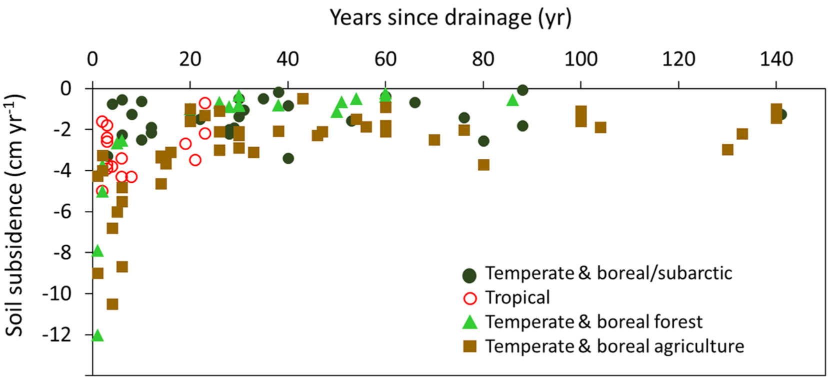

Figure 2 Soil subsidence rates over time (data of temperate and boreal forest and temperate and boreal agriculture from Liu et al., 2020: data of temperate and boreal/subarctic and of tropical peatlands from Evans et al., 2019 and references therein).

of peat, drainage intensity, and physical properties of the peat are important factors influencing soil subsidence rates in the initial phase (Oleszczuk et al., 2020). Thereafter, soil subsidence rates decreased with lower subsidence rates in boreal and temperate forest (-0.67 ± 0.3 cm yr^{-1}) compared to boreal and temperate croplands (-2.1 ± 0.9 cm yr^{-1}, Liu et al., 2020) and to tropical forest (-2.3 ± 1.0 cm yr^{-1}; Evans et al., 2019). The considerably shallower water table of the forest sites compared to the arable sites (35 cm vs. 87 cm) might explain the low subsidence rates for forests (Liu et al., 2020).

Besides time since drainage, the rate of subsidence varies strongly with drainage depth. As the groundwater table determines the volume of exposed peat material, the drainage depth is a key factor: A linear increase in subsidence rate with decreasing ground water table was found for high-latitude and tropical peatlands (Evans et al., 2019). A decrease in drainage depth from 40 cm to 80 cm resulted in a threefold increase in subsidence rates from -0.42 cm yr^{-1} to -1.27 cm yr$-^{1}$ in temperate and boreal/subarctic peatlands. In tropical peatlands, a similar decrease in water table seems to result in higher subsidence rates as an increase from -2.8 to -4.5 cm yr^{-1} was recorded. However, this might also have resulted from the shorter drainage time of 8.7 years in the tropical system, while the temperate peatlands had been drained for 37 years.

Peat properties such as density and thickness add to the observed variation in subsidence rates. For the Sacramento Delta, a strong relationship was found between the content of organic matter (OM) and subsidence rates. After 100 years of initial drainage, subsidence rates were less than 1 cm yr^{-1} in areas where organic matter content in the topsoil was below 10% but considerably higher (3.5 cm yr^{-1}) in areas with high organic matter content (60%; Deverel and Leighton, 2010). In agreement with this finding, the spatial variation in

SOM content, ranging from 4% to 60%, explained over 55% of the variation in average subsidence rates from 1978 to 2006 in the same area (Rojstaczer and Deverel, 1995; Deverel and Leigthon, 2010). Similarly, thicker peat deposits showed higher subsidence rates than shallow peat deposits over a 38-year observation period in Poland's peatlands (Grzywna, 2017).

With ongoing subsidence, the soil surface (relative to the ground water table) decreases and the peatland becomes wet again when subsidence depth approaches the groundwater table. To maintain an aerated zone in the organic soils for agriculture, repeated drainage is necessary, thus becoming a self-perpetuating process. In temperate regions, the drainage system has to be renewed every 40–50 years (Ferré et al., 2019). Long-term drainage will finally lead to a loss of the whole peat deposit – converting the organic into a mineral soil. The subsequent long-term agricultural suitability depends strongly on the properties of the underlying mineral material. Coastal regions with substantial soil subsidence over the last century of up to 8 m lead to levee instability or failure, island inundation, and increased seepage penetration into the groundwater (Deverel and Leigthon, 2010; Erkens et al., 2016). In addition, high mineralization rates of the peat cause enhanced nutrient mobilization and thus the risk of water eutrophication (Kløve et al., 2017, 2010; Prévost et al., 1999).

2.3.2 Pathways and forms of C loss in drained peatlands

During the aerobic decomposition processes of peat in the aerated zone, most of the carbon is lost as CO_2 (Fig. 3; IPCC, 2014). Only a minor part is released as CH_4 via the ditch waters. Besides gaseous carbon loss, carbon may be transported in the liquid phase. Losses of particulate organic carbon (POC) are often connected to water erosion processes. Moreover, POC can also be eroded by wind, especially when the peat is not covered by vegetation. Peat fires represent another substantial source of carbon release. Occasionally they are man-made fires for management purposes, but fires may also occur naturally in drained peatlands after prolonged drought periods. In addition to CO_2, CO, and CH_4 emissions from these fires, smoke is released reducing light and growth conditions and consequently CO_2 uptake by plants (Turetsky et al., 2015). The most direct carbon loss in peatlands is peat extraction for horticulture and energy use — still practised in several northeastern European countries. It is estimated that global emissions from peat mining activities amount to 0.066 Pg CO_2-eq yr^{-1}, in contrast to about 1 Pg CO_2-eq yr^{-1} released by peat fires and 1.5–2 Pg CO_2-eq. yr^{-1} by microbial decomposition of peat (Leifeld et al., 2019).

For estimation of SOC losses, default values as compiled by IPCC (2014) can be used if no detailed country-specific information is available (see Chapter 9 of this book). These default values were derived from studies of CO_2 and CH_4 emissions from land and ditches waters, DOC loss and carbon loss from peat fires. Emission

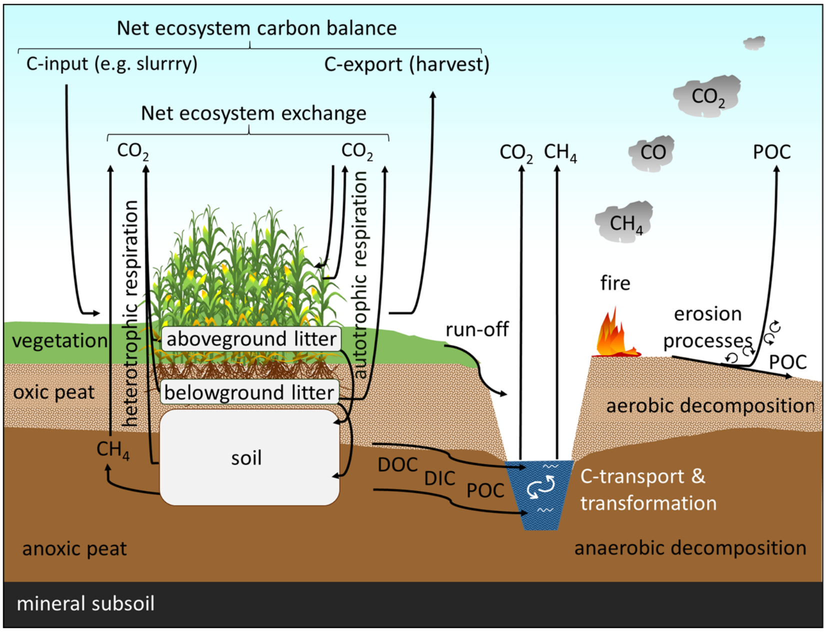

Figure 3 Summary of main carbon fluxes from drained organic soils contributing to SOC stock changes and additional carbon exchange fluxes due to management of agricultural systems on organic soils (see Section 2.1.1; modified after IPCC, 2014 and Jauhiainen et al., 2019).

factors were assigned to different climate zones and land use (Fig. 4). Depending on the available number of studies, more specific subclasses such as nutrient status or drainage depth are given for some categories. Integration of new data will be used to update and improve the estimated emission factors. Generally, total carbon losses from drained peatlands are dominated by CO_2 emissions, contributing to the total C loss by 93% in the tropics, 94% in the temperate zone, and 96% in the boreal zone, followed by DOC losses (IPCC, 2014). CH_4 emissions amount for <0.1% of the total SOC losses, but are more relevant when fluxes are expressed as CO_2-equivalents. However, the contribution of these fluxes may differ considerably across peatland types: In drained blanket peat in the UK, DOC fluxes were the dominating pathway, followed by DIC, POC, CH_4, while the net CO_2 balance was positive (Rowson et al., 2010).

2.3.3 Drivers of CO_2 emissions in drained peatlands

Following Fig. 4, temperature is a key driver for CO_2 emissions from drained peatlands, causing particularly high emissions in the tropics. In addition, in all climate zones, the groundwater level strongly determines the CO_2 emission (Evans et al., 2021; Huang et al., 2021). Independent of the climatic region,

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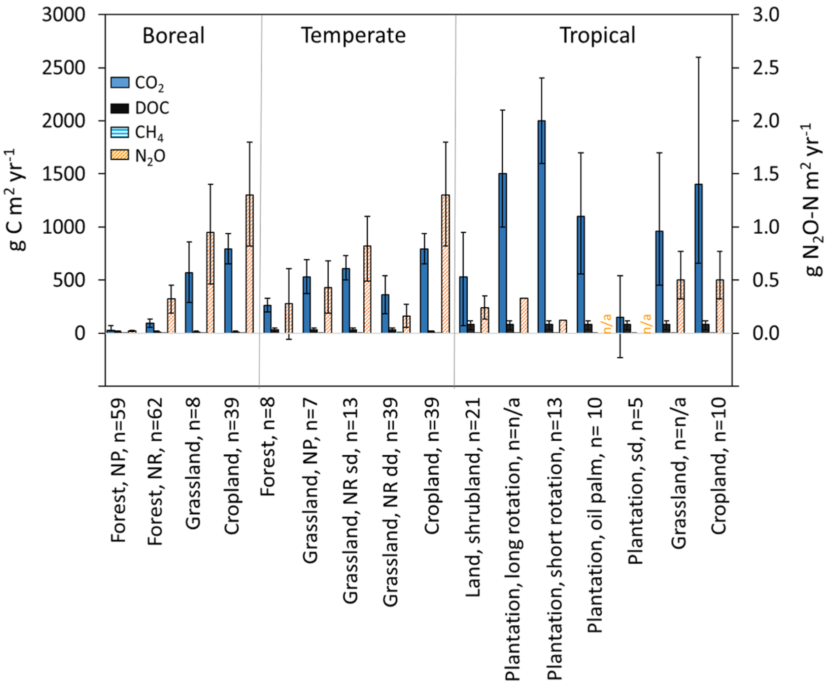

Figure 4 Summary of IPCC default values of drained peatlands (IPCC, 2014; 95% confidence interval; NR, nutrient-rich; NP, nutrient-poor; sh, shallow-drained (<30 cm); dd, deep-drained (>30 cm); n, number of studies for emissions factor of CO_2). No differentiation was made for the emission factor of temperate and boreal cropland. CH_4 includes direct emissions from land and from ditch waters. CH_4 fluxes are all <7 g C m^{-2} yr^{-1} and consequently too small for visualization.

considerably higher emissions are found at a drainage depth of −80 cm compared with −40 cm (Table 1). Generally, CO_2 emissions increase linearly with falling water table: per 10 cm water table drawdown by 188–267 g C m^{-2} yr^{-1} in tropical forests, by 134 in temperate peatlands in the UK and Ireland and by 16–33 g C m^{-2} yr^{-1} in boreal forests (Evans et al., 2021; Hooijer et al., 2012; Ojanen and Minkkinen, 2019; Table 1). In contrast, German peatlands are more sensitive and showed high site-to-site variability in CO_2 emissions at the same water table (Tiemeyer et al., 2020). Studies with water tables below 60 cm are scarce (Ojanen and Minkkinen (2019).

In some cases, the relationship between water table depth and CO_2 emissions is not linear (e.g. Tiemeyer et al., 2020; Couwenberg et al., 2010). In dry summers, the mineralization of peat in the upper layer may be limited by soil moisture (Glatzel et al., 2006; Hahn-Schöfl et al., 2011; Tiemeyer et al., 2016). Moreover, in shallow peat, no additional carbon may be mineralized with further lowering of the water table (Paul et al., 2021). While groundwater levels

Table 1 Change of carbon emissions per 10 cm water table drawdown (ΔWT_{10cm}) and resulting carbon emissions at a water table of 80 cm (WT_{80}) and 40 cm (WT_{40}) estimated from the relationship of water table and emission derived from studies of different peatlands, climatic region and land use.

Peatland	Land use	ΔWT_{10cm} (g C m^{-2} yr^{-1})	Drainage depth (cm)	Emission WT_{80} (g C $m^{-2}yr^{-1}$)	Emission WT_{40} (g C m^{-2} yr^{-1})	Method	*N*	Reference
Boreal NR, Finland	Forest	33	1–91		100	EC (NECB) SR + L	37	Ojanen and Minkkinen (2019)
Boreal NP, Finland	Forest	16	0–60		−5	EC (NECB) SR + L	33	Ojanen and Minkkinen (2019)
Temperate+ boreal NR + NP	Natural, rewetted cropland, grassland peat extraction	92	3–100	570	200	EC (NECB)	65	Evans et al. (2021)
Temperate NR + NP, UK, Ireland	Natural, rewetted cropland, grassland	134	1–91	900	360	EC (NECB)	16	Evans et al. (2021)
Temperate NR + NP, Germany	Rewetted, semi-natural, sphagnum farm, grassland	140-486	0–20[a]		543 WT_{20}	CH (NECB)	44	Tiemeyer et al. (2020)
Temperate NR + NP, Germany	Rewetted, grassland cropland	360-111	20–40[a]		960	CH (NECB)	34	Tiemeyer et al. (2020)
Temperate NR + NP, Germany	Grassland, cropland peat extraction	32-9	40–60[a]			CH (NECB)	19	Tiemeyer et al. (2020)
Temperate NR + NP, Germany	Grassland, cropland	0–2.5	60–120[a]	1000		CH (NECB)	15	Tiemeyer et al. (2020)
Tropical	Forest	189	3–64	1190	435	EC (NEE)	6	Evans et al. (2021)

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Tropical	Forest Indonesia	267	0–110	2138	1069	SS	51[b]	Hooijer et al. (2012)
Tropical	Plantation Indonesia	188	30–120	2078	1325	SS	167[b]	Hooijer et al. (2012)

NR = nutrient rich peatland, NP = nutrient poor peatland, Methods: EC, Eddy-covariance; CH, chamber; NECB, net ecosystem carbon balance; SR+L, soil respiration and litter production; SS, soil subsidence; EC (NEE), only NEE was measured, the tree stand biomass increment was not considered, therefore, NECB may be higher than NEE, all approaches of NECB did not include fluvial carbon losses, *N* = number of site. References Ojanen and Minkkinen (2019), Evans et al. (2021), and Hooijer et al. (2012) used linear models.

[a]Tiemeyer et al. (2020) fitted non-linear models, therefore, an average of emissions for a change in drainage depth was calculated to different 20-cm depth intervals.

[b]Number of measurement points, which were distributed over 16 transects with a longitude of 0.5–12 km.

can have a high predictive power for the CO_2 emissions in consecutive years for specific sites (Fortuniak et al., 2017; Hirano et al., 2012), it is unclear whether this is a general pattern that also holds for deeply drained peatlands.

In a newly established palm oil plantation, McCalmont et al. (2021) revealed that respiration from the upper soil horizons is more sensitive to water table changes than that from deeper layers. A 30-cm water table decrease from −50 to −80 cm resulted in 7% higher soil respiration compared to 37% when the water table fell from −10 cm to −40 cm (McCalmont et al., 2021). In line with this observation, Hirano et al. (2012) found a significantly stronger response of a tropical peat swamp upon water table lowering during an El Nino event, compared to a drained peatland. Recent EC measurements in an oil palm plantation revealed a very high carbon loss rate of −3.6 kg C m^{-2} in the first year following plantation establishment, which was reduced to −2.9 kg C m^{-2} in the second and third year and even further in the mature phase (−1.3 kg C m^{-2} yr^{-1}, McCalmont et al., 2021). One part of this considerable loss was explained by a higher water table in mature plantations compared to newly established ones (−0.25 cm vs. −0.60 cm). In addition, different peat qualities, originating from the antecedent decomposition of peat may be one factor explaining differences in mineralization rates with depth (Section 2.3). Also differences in vegetation induce different peat qualities which was found to affect decomposability (e.g. Duval and Radu, 2018; Maie et al., 2019; Moore and Dalva, 1997; Scanlon and Moore, 2000). Whether peatland type also affects CO_2 release after drainage is not yet clear. Whereas two studies found higher CO_2 mineralization rates for bogs than for fens (e.g. Alm et al., 1999a; Deppe et al., 2010), other studies reported only small differences in CO_2 emission between different peatland types (e.g. Moore and Dalva, 1993; Updegraff et al., 2001). Moreover, a meta-analyses by Huang et al. (2021) found a considerably stronger increase in CO_2 emissions with increasing drainage depth for fens than for bogs.

It is well established that peat decomposition increases with soil temperature in all climatic regions (Carter et al., 2012; Wilson et al., 2016a). Laboratory incubations indicated that Q10 values for CO_2 release under aerobic conditions are in the range of 1.4–2.5 for tropical peatlands (Girkin et al., 2020; Jauhiainen et al., 2019; Maie et al., 2019; Inglett et al., 2012; Ali et al., 2006), 1.8–4.2 for temperate peatlands (Bader et al., 2017; Hardie et al., 2011; Lund et al., 2007; Moore and Dalva, 1993), and 1.2–5.0 for boreal peatlands (Bubier et al., 1998; Yavitt et al., 2000). Noticeably, the boreal peatlands showed the highest variability in temperature sensitivity. The variability for CH_4 production rates was even more pronounced in boreal and temperate peatlands, which could be related to vegetation composition. CH_4 emissions from Sphagnum-dominated peatlands were more temperature sensitive with an average Q10 of 8.0 (at single sites Q10 values of up to 30) than wetter minerotrophic sedge-dominated peatlands (Q10 = 4.3, Lupascu et al., 2012). Q10 values of 1.9–5.8 were measured for an arctic permafrost layer in Sweden (Lupascu et al., 2012).

Determining the temperature dependency of CO_2 emission rates using field measurements is very challenging as land-use history, time since drainage, water table depth, management intensity, and other confounding factors differ between peatlands and regions. While in the tropics major drainage systems are relatively young, drainage is ongoing since centuries in the boreal and particularly in the temperate zone.

As the emitted CO_2 results from the combined effect of various controlling factors, including the composition of peat, climatic conditions (soil moisture and temperature), nutrient availability, land-use history, the influence of a single parameter on the CO_2 emissions is site-specific. It is up to now impossible to predict the emissions for a specific peatland and concurrently estimate the reduction potential of GHG emissions of management strategies for a particular site. However, on a broader scale, it can be concluded that the deeper the water table, the lower the latitude, and the nutrient richer the peatland is, the higher the CO_2 emissions and consequently the SOC loss.

2.3.4 Waterborne carbon losses in form of dissolved organic carbon (DOC) and particulate organic carbon (POC) in drained peatlands

Waterborne C losses of peatlands comprise different carbon species of various origins. DOC is released through biological activities during decomposition processes and plant growth. It is transported horizontally and vertically and can be exported from the peatland (Fig. 4) with DOC fluxes ranging from 5 g C m^{-2} yr^{-1} in subarctic and boreal peatlands to 100 g C m^{-2} yr^{-1} in tropical peat swamps (Evans et al., 2016; Moore et al., 2013). Temperature was found to be a major driver for DOC fluxes. Also within single sites, peat temperature may control the baseline flux of DOC (Rosset et al., 2020). In addition, drainage is usually considered as a driver for DOC production as it aerates the peat volume and consequently accelerates the decomposition rate and induces a shift in vegetation (e.g. Billett et al., 2006; Ritson et al., 2017; Strack et al., 2008). For example, Evans et al. (2016) found an overall higher DOC mobilization of 67% in drained as compared to natural peatlands. A positive correlation of water table depth with DOC fluxes was observed in drained oil palm plantations (Cook et al., 2018). Drainage depth was found to influence the age of DOC with deeper drainage leading to mobilization of older peat-derived carbon than shallow-drained peatland. Dean et al. (2019) analysed ^{14}C data from studies that measured DOC export to aquatic systems from drained peatland catchments. As a general feature, DOC from arctic, boreal, temperate, and tropical undisturbed and only slightly disturbed peatlands (n = 179) often had modern ages (87–90%), while 70% of disturbed peatlands (n = 30) had DOC, which showed older ages (Dean et al., 2019). This implies that drainage may trigger mobilization of older peat in form of DOC.

The contribution of DOC to the total carbon balance depends on the ecosystem type, disturbance degree, and climatic region and may be a quantitatively important component of the carbon balance. DOC fluxes of about 60 g C m^{-2} yr^{-1} in an undrained and about 100 g C m^{-2} yr^{-1} in a drained peat swamp were reported by Moore et al. (2013). In the latter case, DOC contributed 16–24% to the total carbon loss, when compared to CO_2 fluxes from the previous years (Hirano et al., 2012). In contrast, DOC fluxes of only 0.53 g C m^{-2} yr^{-1} contributed less than 2% to the total carbon loss of a temperate drained peat under grassland (Tiemeyer and Kahle, 2014), and in intensively used peatland in northern Germany, DOC fluxes of 4.28 g C m^{-2} yr^{-1} contributed about 8% to total carbon loss (Tiemeyer et al., 2016; Frank, 2016). In contrast to temperate or tropical systems, DOC is more relevant for the overall carbon budget in northern peatlands of the cool temperate or boreal zone with moderate disturbance (IPCC, 2014). For example, adding the DOC fluxes of 13 g C m^{-2} yr^{-1} in a peatland in northern Sweden to the gaseous exchange reduced the total carbon sink considerably from −37.5 g C m^{-2} yr^{-1} to −23.3 g C m^{-2} yr^{-1} (Nilsson et al., 2008). In a raised bog in Southern Scotland, waterborne fluxes of 30 ± 6.2 g C m^{-2} yr^{-1} even equalled CO_2 fluxes (28 ± 2.5 g C m^{-2} yr^{-1}, (Billett et al., 2004).

In comparison to DOC, POC contributes significantly only in disturbed peatlands to waterborne carbon fluxes (Billett et al., 2010; Cook et al., 2018; Dawson et al., 2002; Shuttleworth et al., 2015; Yupi et al., 2016). As particles are primarily detached and transported during erosion, relevant POC fluxes occur for bare peat during runoff situations (Billett et al., 2010; Holden, 2006; Pawson et al., 2012). In strongly disturbed blanket peatlands in the UK with eroding sites, POC fluxes of 13–79 g C m^{-2} yr^{-1} were recorded (Pawson et al., 2012). Evans et al. (2019) calculated that POC fluxes in these systems increased by around 4 g C m^{-2} yr^{-1} per 1% of exposed bare peat area. For tropical peat swamps, Moore et al. (2013) also found higher POC fluxes in disturbed (10 g C m^{-2} yr^{-1}) than in intact peat swamp forests (1 g m^{-2} yr^{-1}), and a high ratio DOC/POC was found to indicate erosion processes (Moore et al., 2013).

2.4 OM composition in intact and drained peatlands

The composition of OM in peat profiles is mostly related to (i) vegetation changes which occur following variations in climate and/or hydrology during millennial peat formation and (ii) ongoing decomposition of the formed peat

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(Moore, 2002). The latter is of particular relevance at it provides insight into mechanisms of OM transformation in the absence of minerals. Although intact peatlands almost steadily accumulate new carbon (Yu et al., 2010), peat formation is coupled to ongoing decomposition both in the acrotelm and in deeper peat layers (Clymo, 1984), albeit transformation rates in the permanently water-saturated zone are thermodynamically strongly constrained (Worrall et al., 2018).

Accordingly, deeper and older peat should reveal a stronger degree of transformation and a more pounced microbial signature at the expense of degradable plant remains. Further, lignins as vascular plant markers might be better preserved owing to the anoxic conditions below the acrotelm. This was confirmed by a range of studies from different peatland ecosystems. Zaccone et al. (2008) studied a cool temperate bog in Switzerland and not only showed that phenols were relatively enriched in the catotelm as compared to the mesotelm but also indicated that lignin decomposition, as revealed from a significant demethoxylation and higher degree of lignin oxidation, continues with depth, albeit at low rates. In general, anoxic conditions prevent lignin decomposition, requiring high amounts of oxygen. Indeed, preferential loss of carbohydrates and selective preservation of alkyl-C and aromatic C with increasing depth, including lignin, was found for tropical and temperate peatlands in many studies (Preston et al., 1987; Upton et al., 2018; Leifeld et al., 2012; Moody et al., 2018; Schellekens and Buurman, 2011; Broder et al., 2012).

In contrast, peat C/N ratios as a possible indicator for the microbial imprint show no clear pattern. Declining peat C/N ratios with depth have been interpreted as to represent the declining rate of peat decay over time with a concurrent preservation of nitrogen in the deeper peat layers (Kuhry and Vitt, 1996), but this pattern cannot be considered universal. For example, C/N ratios decreased from 50 to 70 in the upper 3.5 m of the profile to around 30 below 5.5 m (Schellekens and Buurman, 2011). Yet, within the upper 3.5 m, C/N ratios increased with depth. Peat bogs studied by Broder et al. (2012) showed no clear trend of the C/N ratio with depth. A compilation of data from dozens of northern peatlands by Loisel et al. (2014) confirms that peat C/N ratios do not follow a systematic trend with increasing peat age (Fig. 5). The diverse sources of N inputs from fixation, atmospheric deposition, and nitrogen inflow, outputs via denitrification and runoff and their variation over longer timescales makes the N content of peat highly variable within a profile (Limpens et al., 2006).

2.4.1 Changes in OM composition of peat after drainage

In drained peatlands, new vegetation is established following agricultural practices, which provides new litter input. For understanding the change in

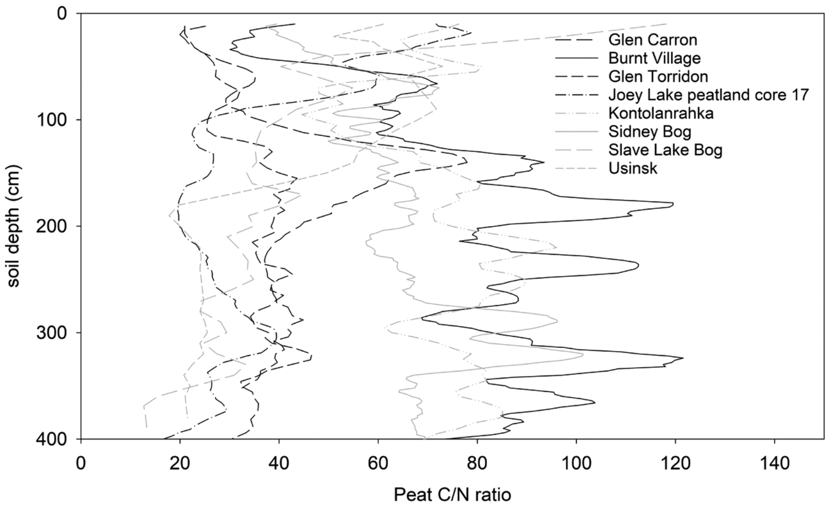

Figure 5 Depth trends of peat C/N ratios for selected sites from Loisel et al. (2014). Data are 20 cm running averages. Site description follows Loisel et al. (2014).

OM composition, peat inherent decomposition processes and incorporation of new inputs are of particular relevance. The contribution of new inputs to topsoil organic carbon (0–30 cm) in degrading peatlands was reported to be in the range of 14–24% after 2 decades, based on results from natural ^{13}C labelling when C3 vegetation was changed to C4 vegetation (Leifeld et al., 2021). The C4 accumulation rate in these topsoils was not significantly different to that of mineral topsoils, suggesting an overall similar dynamics of incorporation of new C. The new assimilates are relatively labile and contribute more to the CO_2 as the peat OM was from managed organic soils (e.g. Bader et al., 2017; Fig. 1).

After drainage, microbial transformation of the peat deposit accelerates, leading to an increase in the N content of OM. This increase can be substantial and is a major distinction between intact and drained peatland soils (Leifeld et al., 2020). A study from North-western Germany indicated that decadal drainage and grassland management reduced C/N ratios from 40–60 to around 20 in the topsoil (0–20 cm). Importantly, the trend was similar at fertilized and unfertilized sites, indicating that C/N in drained peatlands is related to OM mineralization and is a relative enrichment of nitrogen (Krüger et al., 2015).

With ongoing decomposition, exposure of older peat changes the overall peat composition towards a higher contribution of aryl-C and alkyl-C at the expense of O-alkyl originating from polysaccharides in boreal and temperate peatlands (Leifeld et al., 2018). In the same study, a relative enrichment of pyrogenic C with ongoing degradation was observed. A higher contribution of aromatics and other recalcitrant moieties was also measured for drained tropical peatlands (Könönen et al., 2016; Gandois et al., 2013; Cooper et al., 2019).

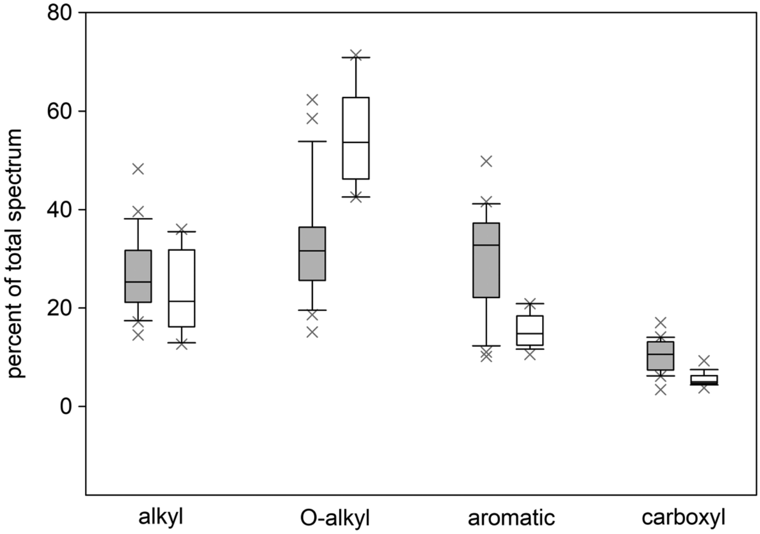

Figure 6 Chemical shift regions of ^{13}C-NMR spectra from tropical (grey) and northern (white) peat samples (alkyl-C: 0–45 ppm or 0–50 ppm; O-alkyl-C: 45–110 ppm or 50–110 ppm, aromatic 110–160 ppm, carboxyl 160–210 ppm or 160–220 ppm). Tropical data are from Purwanto et al. (2005), Sangok et al. (2017, 2020), Wright et al. (2011), northern are from Leifeld et al. (2018). All samples were derived from intact peatlands and were taken in the upper 1 m of the profile. Tropical $n = 22$, northern $n = 17$.

An increase in alkyl-C and a corresponding decline in O-alkyl-C is generally related to microbial decomposition of organic material, which is characterized by the accumulation of recalcitrant aliphatic plant compounds and/or microbial material (Baldock et al., 1997).

2.4.2 Latitudinal gradients in peat composition

Lowland tropical peats often form in peat swamp forests, where not only broadleaf trees but also palm trees (palm forest swamps) and sedges can be peat building. In addition, the vegetation in temperate and boreal peatlands is highly variable, including sphagnum spp sedges, broadleaf, and coniferous species, providing organic material for peat formation. The different conditions regarding vegetation, hydrology, and climate result in different peat properties. Tropical peatlands show lower contribution of polysaccharides and higher aromaticity than northern peatlands (Fig. 6; Hodgkins et al., 2018) because they contain higher contribution of woody species than temperate and boreal peatlands. In addition, tropical peatlands show on average lower C/N ratios (35.0 ± 17.8 SD; Leifeld and Menichetti, 2018) than northern peatlands (average 42.9 ± 18.8; Loisel et al., 2014). For the latter, sphagnum bogs contain the least nitrogen, resulting in wide C/N ratios of 81.0 ± 49.2.

2.4.3 Implications of peat composition for C loss of managed peatlands

The accumulation of peat is strongly related to the anoxic conditions of the peat deposit. The low abundance of oxygen directs microbial transformation towards other terminal electron acceptors, which implies a lower energy return and, finally, lower rates of decomposition (LaRowe and Van Cappellen, 2011). In addition, even reactions with a negative Gibbs energy might not occur owing to the energy requirement for ATP production and an efficiency of reaction of below 100% (Worrall et al., 2018). With drainage, these limitations are released and peat decomposition by microbes occurs at high rates.

In line with the thermodynamic constraints of water-logged peatlands, CO_2 fluxes from drained peatlands very much depend on climate and water table depth and thus oxygen availability as discussed above. Yet, these factors leave a large unexplained variability, and it has been hypothesized that the chemical composition of different peats might be a determining factor for CO_2 emissions from managed organic soils (Glatzel et al., 2004; Leifeld et al., 2012). In addition, also the abundance or absence of key controlling molecules such as polyphenols or tannins and their effect on OM decomposition might guide peat decomposition under drier conditions (Fenner and Freeman, 2020).

In many studies, peat samples from different soil depths were incubated in the laboratory and their CO_2, CH_4, or N release was monitored. Such an approach might be useful also to disentangle the role of peat composition on decomposability. Almost all of these studies found higher CO_2 release rates under aerobic conditions (Brake et al., 1999; Hardie et al., 2011; Hogg, 1993) and also under anaerobic conditions (Glatzel et al., 2004; Sjögersten et al., 2016) for surface samples, although depth effects were not always pronounced (Bader et al., 2018a). The van Post degree of humification, a widely used but unspecific field indicator of peat degradation (Malterer et al., 1992), was found to be positively related to the CO_2 release, that is, more decomposed peat, typically situated in the aerated topsoil of managed organic soils, was more labile (Glatzel et al., 2004; Säurich et al., 2019). However, simple indicators such as nutrient content or peat stoichiometry failed to explain peat decomposability (Bader et al., 2018a; Säurich et al., 2019). A higher degradability of surface peat might not be related to peat composition but to changes in microbial community composition and higher soil microbial activity in aerated topsoil peats (Brake et al., 1999), and to a high contribution of labile OM that derives from the current vegetation on former peatlands which may also induce priming effects (Bader et al., 2018b).

More specific approaches confirmed a regulatory role of OM composition for peat decomposability. An incubation of various tropical peats revealed a reduced nitrogen mineralization rate with increasing alkyl/O-alkyl-C ratio, that is,

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with increasing microbial transformation of the OM, as measured by solid-state ^{13}C-NMR spectroscopy (Purwanto et al., 2005). More specifically and also for tropical peat, Hoyos-Santillan et al. (2016) showed that both in anoxic and oxic environments, lignin moieties, longer-chain fatty acids, and polysaccharides, all analysed by TMAH-pyrolysis-GC-MS, were positively related to CH_4 and CO_2 release. For the anaerobic incubation of temperate peat, Reiche et al. (2010) showed that the peats' thermal stability, depending on the amount of carbohydrates in peat, explained 71% (CH_4) and 49% (CO_2) of the release of these two GHGs. Importantly, the relationship was non-linear. Leifeld et al. (2012), Sjögersten et al. (2016), and Normand et al. (2021) found a positive relationship between the O-alkyl-C content, as measured by ^{13}C NMR, and CO_2 release from various peatland soils. Together, these findings indicate that (i) peat composition varies steadily within the peat profile, (ii) peat quality changes systematically after drainage, and (iii) peat quality is one relevant factor for the release of CO_2 after drainage.

3 Management strategies to reduce soil organic carbon loss from drained peatlands

The most efficient ways to prevent further SOC loss from peatlands is to protect existing intact peatlands and to raise the water table of managed (drained) ones. Rewetting is then accompanied by either restoration of the peatland with the loss of the agricultural production function or with cropping of water-tolerant crops. In addition, engineering measures with continued management at deeper water tables were suggested.

3.1 Restoration and rewetting of peatlands without use

Rewetting of peatlands means restoring the water table or hydrological regime towards a condition where the new groundwater level is close to the surface of the peatland, with the aim of a partial or total reversal of drainage. However, the original ecosystem functions may be only partly restored. Restoration of peatlands aims at revitalizing the peat accumulation process. This includes the rehabilitation of original fauna and flora and thus the original ecosystem functions including carbon sequestration. In addition to rewetting measures, active restoration techniques that reintroduce peatland plant species are important (Landry and Rochefort, 2012), while the term 'rehabilitation' refers to natural succession after the abandonment of active management.

The continuous loss of SOC from degraded and drained peatlands can be reversed by rewetting as the main trigger for carbon release – the aerobic mineralization of the peat – is prevented (Evans et al., 2021; Huang et al., 2021). A common observation is that while CO_2 and N_2O fluxes decrease CH_4

emissions increase (Figs 4 and 7; Wilson et al., 2016a). After the successful recovery of the original vegetation, the carbon fluxes match those of undisturbed peatlands (IPCC, 2014; Nugent et al., 2019; Wilson et al., 2016b). Natural peatlands are characterized by a mosaic of habitat structure, including small-scale differences in water tables resulting in specific plant communities and GHG emission patterns (e.g. Lai et al., 2014; Laine et al., 2007; Maanavilja et al., 2011). Small-scale heterogeneity is often restored by rewetting (Wilson et al., 2013; Strack et al., 2014). However, carbon fluxes may differ in the initial phase after rewetting. Observed responses include (i) a stronger carbon sink due to higher carbon uptake during vegetation establishment, (ii) a consistent large carbon source due to high CO_2 emissions, or (iii) high CH_4 emissions (Valach et al., 2021; Waddington et al., 2010; Wilson et al., 2013). These short-term rewetting effects are often attributed to a transition period before a new equilibrium is established. Restoring the original ecosystem functions may be difficult and is a long-term objective (Vanselow-Algan et al., 2015; Renou-Wilson et al., 2019; Samaritani et al., 2011; Wilson et al., 2016b). The restoration success also depends on the state of degradation of the peatland and restoration techniques. In some cases, restoration may not be possible due to hydrological restriction or altered peat properties (Kløve et al., 2017). In some cases, restored wetlands may shift to stable states that differ from their original condition (Moreno-Mateos et al., 2012).

3.1.1 CO_2 flux changes after rewetting of drained peatlands

Strong reductions of CO_2 emissions upon rewetting was observed for temperate and boreal peatlands (IPCC, 2014; Wilson et al., 2016a). Most

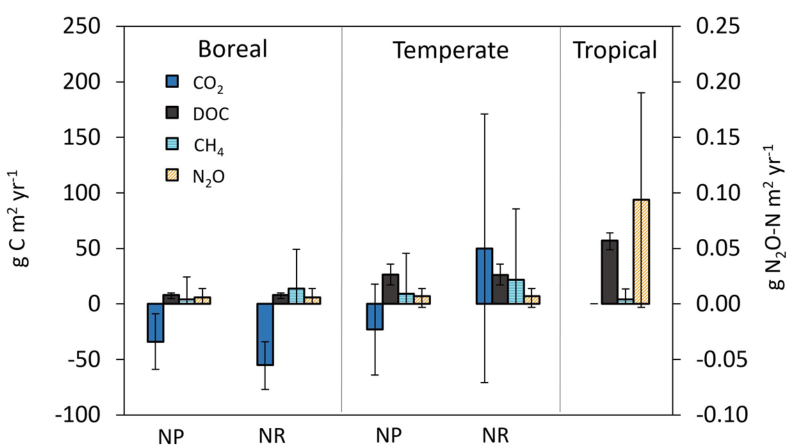

Figure 7 Carbon fluxes of rewetted and natural peatlands for boreal, temperate, and tropical climate zone. Data from Wilson et al. (2016a). NP, nutrient-poor; NR, nutrient-rich.

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rewetted peatland types are, on average, carbon sinks. The strongest annual CO_2 sink is found in nutrient-rich boreal peatlands (−53 g C m^{-2} yr^{-1}), followed by nutrient-poor peatlands (−41 g C m^{-2} yr^{-1} for boreal and −33 g C m^{-2} yr^{-1} for temperate peatlands, Wilson et al., 2016a, Fig. 7). As an exception, nutrient-rich temperate peatlands showed lower CO_2 emissions of 26 g C m^{-2} yr^{-1} (Wilson et al., 2016a). Nutrient-rich peatlands display a wider range of fluxes than nutrient-poor sites in temperate and boreal peatlands, which can be explained by their high hydrological, biogeochemical, and biological diversity (Wilson et al., 2016a). Different functional plant types, including mosses, grasses, and sedges, are growing in fens (Joosten and Clarke, 2002). Especially boreal fen sedges are known to be a highly productive system, explaining the greatest carbon uptake in this system (IPCC, 2014). The elevated diversity of previous land use of temperate fens might add to the high site-to-site variability, as these fens were more intensively used than nutrient-poor bogs (Joosten and Clarke, 2002). In addition, a higher disturbance degree or the incorporation of studies on incompletely rewetted sites, where the natural peat-forming vegetation has not re-established, or recently rewetted sites that are still undergoing transitional changes, may explain the observed CO_2 emissions from rewetted nutrient-rich temperate peatlands (Wilson et al., 2016a). Also severe droughts might shift peatlands into a temporary carbon source (Alm et al., 1999b; Lund et al., 2012; Rinne et al., 2020).

Like in drained peatlands, an influence of mean water table on the CO_2 balance was observed for rewetted peatlands, with lower emission or higher sequestration for elevated water tables. The decrease in CO_2 emissions per cm change in the water table is more pronounced in temperate climate (44 g C m^{-2} yr^{-1} per 10 cm water table increase) than in boreal climate (17 g C m^{-2} yr^{-1} per 10 cm water table increase, IPCC, 2014). A water table above −10 cm depth is necessary to achieve a CO_2 sequestration in temperate peatlands, while boreal peatlands showed a net CO_2 uptake down to a water table depth of −30 cm. Up to date, no data of rewetted tropical peatlands are available; therefore, Wilson et al. (2016a) suggested a default value of 0 based on the observation that soil subsidence stopped when rewetting tropical organic soils. Due to the low number of observation in tropical peatlands, these emission factors have a high uncertainty (Fig. 4).

3.1.2 CH_4 emissions after rewetting of peatlands

As methane is produced under anaerobic conditions, increased CH_4 emissions after restoration of anaerobic conditions is a logical consequence. Up to date, the database for CH_4 emissions of rewetted sites is limited. As Wilson et al. (2016a) found no significant differences in CH_4 emissions between natural (undrained) and rewetted sites, data of natural sites are useful to explain the driving factors of CH_4 emission. Several factors, like water table height, plant species composition, and nutrient status, control CH_4 emissions of intact peatlands. Similar to the CO_2 emissions, the water table is considered the key driver for CH_4 emissions. A water table depth of <−0.25 m is sufficient to markedly reduce CH_4 emissions as CH_4 produced in the anoxic zone is oxidized by methanotrophs in the oxic zone (Turetsky et al., 2015; Evans et al., 2021; Tiemeyer et al., 2020). Annual CH_4 fluxes exponentially increase with raising the water table but with very high site-to-site variability. Comparing 108 peatlands from the boreal and temperate zone, Abdalla et al. (2016) reported higher CH_4 emissions for nutrient-rich environments, such as fens (15.4 g C m^{-2} yr^{-1}), compared to naturally nutrient-poor bogs (7.1 g C m^{-2} yr^{-1}). This pattern was confirmed for temperate peatlands in Germany (Tiemeyer et al., 2016). However, CH_4 emissions were considerably higher for shallow-drained nutrient-rich peatlands (79 ± 104 g C m^{-2} yr^{-1}) and for nutrient-poor peatlands (10.3 ± 24 g C m^{-2} yr^{-1}), indicating overall higher CH_4 emission upon rewetting in temperate peatlands (Abdalla et al., 2016; Tiemeyer et al., 2016).

In addition to hydrology, peatland vegetation is a key factor to explain the magnitude of CH_4 fluxes. Some graminoid plant species have an aerenchyma tissue, facilitating the direct gas exchange between roots and the atmosphere, thereby allowing methane to bypass the upper oxic peat layers (Schimel, 1995). Accordingly, wet temperate peatlands without this specialized plant species showed considerably lower CH_4 emissions (mean: 5 (range: −0.02 to 25 g C m^{-2} yr^{-1})) than peatlands supporting a vegetation with an aerenchyma tissue (mean: 17 (range: 0–76 g C m^{-2} yr^{-1})) (Couwenberg and Fritz, 2012; Turetsky et al., 2014).

Temperature is another crucial driver for CH_4 emissions. Plant productivity and input of labile carbon from plant exudates and mineralization processes depend on temperature (Conant et al., 2011). CH_4 fluxes decreased with increasing latitude from temperate to subarctic in most fens (Turetsky et al., 2014). In contrast, tropical peatlands showed an opposite trend. Wet tropical peatlands released on average only 6.1 g C m^{-2} yr^{-1} (Wilson et al., 2016a). Recent CH_4 fluxes from EC indicate considerably higher CH_4 emissions of 7.5–22 g C m^{-2} yr^{-1} in tropical peat swamps (Deshmukh et al., 2020; Griffis et al., 2020; Wong et al., 2018). Higher CH_4 emissions from these recent EC measurement suggested that plant-mediated CH_4 transport may play a role in the overall CH_4 emissions to the atmosphere, and this process is usually not captured by chamber measurement on which earlier observations were predominantly based (IPCC, 2014; Pangala et al., 2017; Welch et al., 2019). However, compared to boreal peatlands, CH_4 emissions of tropical sites are low despite the higher temperatures. This may be explained by high mineralization rates under warm temperatures in combination with a high contribution of lignin from tropical forests, which result in recalcitrant peat (Section 2.3) with a lower availability of labile substrate for methanogenesis.

In most cases, rewetting drained peatlands reduces GHG emissions (Section 3.4), however, high CH_4 emissions after rewetting can counteract mitigation strategies of peatlands that aspire an improved GHG balance (Waddington and Day, 2007; Wilson et al., 2009). High annual emissions up to 370 g CH_4-C m^{-2} yr^{-1} were measured in a nutrient-rich fen in the first years after rewetting (Augustin and Chojnicki, 2008). Rapid flooding of the site led to a plant die-off, with the decomposing litter serving as an easily degradable carbon source fuelling microbial decomposition and subsequent development of anaerobic conditions leading to the high CH_4 emissions (Hahn-Schöfl et al., 2011). In addition, high nutrient availability and lateral carbon import from floating biomass supported high methanogenesis rates (Augustin and Chojnicki, 2008). However, 8 years later considerably lower annual CH_4 emissions of about 53 g CH_4-C m^{-2} yr^{-1} from open water surfaces and 13 g CH_4-C m^{-2} yr^{-1} from emerging littoral vegetation were measured (Franz et al., 2016). Similarly, CH_4 emission after rewetting of a highly degraded coastal fen decreased in the first 2 years after rewetting (Hahn et al., 2015; Koebsch et al., 2015). These studies suggest that rapid flooding of nutrient-rich fens, where the vegetation was not adapted to flooding conditions, may induce a CH_4 peak and seems to represent a transition state with decreasing CH_4 emissions over time. However, other studies indicated high CH_4 emission rates depend on the vegetation types even after 30 years of rewetting (Vanselow-Algan et al., 2015). These emissions were explained by a strongly fluctuating water table, including inundation periods with the anaerobic decomposition of easy decomposable plants and shunt species (Vanselow-Algan et al., 2015). Similarly, mulching a rewetted peatland where the grass cut was left on the side, with temporary inundation, resulted in relatively high CH_4 emissions (Kandel et al., 2019b). Thus, it appears that not the time since rewetting is important, but that the hydrological regime and plant communities should be controlled to prevent high CH_4 emissions.

CH_4 emissions may additionally be affected by nutrient availability: removal of the upper 30 cm of degraded topsoil prior to rewetting of a former intensively used bog reduced CH_4 emissions considerably in the first year (Huth et al., 2020). Alongside the CH_4 reduction following topsoil removal, reduced fluxes of DOC (60%) and cations (80–90%) from a rewetted former agricultural peatland were recorded (Harpenslager et al., 2015). Former land use and hydrologic condition should be considered before rewetting. In particular, additional input of easily decomposable carbon and nutrients and prolonged flooding conditions, especially when vegetation is not adapted to inundation, should be avoided.

3.1.3 DOC changes following rewetting

DOC fluxes contribute essentially to total carbon fluxes in some peatlands systems (Section 2.2.2). Drainage usually increases DOC fluxes (Section 2.2.4). It is assumed that rewetting of drained organic soils generally decreases the DOC fluxes from peatland (IPCC, 2014; Evans et al., 2016; Wilson et al., 2016a). Limited information is available on the influence of land use and nutrient status on DOC fluxes from rewetted peatlands. Accordingly, only aggregated values are given for the climate zones (51, 24, and 8 g C m^{-2} yr^{-1} for tropical, temperate, and boreal zones, respectively (IPCC, 2014, Fig. 7). Several studies report higher DOC fluxes in the initial phase after rewetting (Glatzel et al., 2003; Rowson et al., 2010; Worrall et al., 2007). Zak and Gelbrecht (2007) explained the increased mobilization of DOC with increased amounts of redox-sensitive substances and enhanced availability of decomposable OM in the upper highly decomposed peat of a nutrient-rich fen. In 24 Finnish peatlands drained for forestry, DOC concentration in pore water increased following the first years of rewetting, thereafter DOC decreased (Menberu et al., 2017). However, 6 years post-rewetting, DOC was still higher in rewetted sites than in pristine peatlands. The increase of DOC together with P-concentration in recently rewetted peatlands under forest was explained by the leaching from forest residues and soil disturbances following clear-felling (Howson et al., 2021).

Differences in the vegetation of rewetted and natural sites were attributed to different DOC concentrations in a cutover bog in Canada: Ten years post-restoration, DOC in a rewetted site remained intermediate between natural and unrestored peatlands (Strack et al., 2015). In addition, DOC was characterized by smaller and less-aromatic structures at the restoration site, which indicated an input of DOC from the fresh litter of growing vegetation at the restored peatland (Strack et al., 2015). In line with this, Herzsprung et al. (2017) found higher DOC concentrations with a higher polyphenol contribution in summer time on sites rewetted for 9 years compared to a natural bog in the Harz Mountain in Germany. The authors attributed this to the higher degree and biomass of vascular plants at the rewetted site.

Given the limited data set, the increased DOC fluxes following drainage generally seem reversible by rewetting. However, as DOC is derived from turnover processes of peat and the living vegetation, the temporal evolution of vegetation after rewetting may be accompanied by temporal changes in DOC as well.

3.2 Agricultural management of peatlands without drainage: paludiculture

The term 'Paludiculture' refers to farming and agroforestry systems, which grow a commercial plant under wetland conditions in a sustainable way. The idea is that the SOC stocks of peatlands are preserved under a high water table and that a cultivated plant species – adapted to wet conditions – generates an economic income to the farmers (Wichtmann et al., 2016). Co-benefits of paludiculture include biodiversity and water purification (Giannini et al., 2017; Vroom et al., 2018; Wichtmann et al., 2016). From a conceptual viewpoint, paludiculture would – under optimal conditions – restore a peat-forming system, although the aboveground part of plant biomass is removed and can consequently not contribute to carbon input. To date, much research on this practice is conducted on pilot studies, which have been recently emerging. The research in northern peatlands focuses on the climate mitigation potential, growth conditions, practical implementation barriers, and political barriers for the cultivation of paludiculture plants. Many of the plants suggested for paludiculture are wetland species that tolerate a wide range of environmental conditions (Abel et al., 2013; Joosten and Clarke, 2002; Melts et al., 2019). As paludiculture is a relatively recent technique, the number of studies providing empirical data concerning field measurement of GHG exchange is limited. Particularly, studies covering multiple years of the complete carbon balance of paludicultures with a whole or several production cycles are missing. As paludiculture plants are colonizing natural habitats as well, few studies exist of these unmanaged systems. It is speculated that these unmanaged systems can serve as a proxy for the carbon balance of managed system and that the effect of biomass removal will not significantly alter the complete carbon balance (Tanneberger et al., 2020).

Recently, the transfer of the paludiculture concept to the tropics has been attempted (Tan et al., 2021). After severe peatland fires in 2015 in Indonesia, there was a need to rewet degraded and burned peatland and simultaneously provide benefit for livelihood, which was supported by the Indonesian Government new peatland restoration strategy (Tata, 2019). In this context, paludiculture research focuses on short-term socio-economic and environmental aspects of existing agroforestry systems on peatlands (Budiman et al., 2020; Tata, 2019). Those socio-economic aspects receive most attention that attempt to find alternatives to land clearance by fire for agriculture (Budiman et al., 2020). Tropical paludiculture systems are even less studied than temperate or boreal ones.

3.2.1 Sphagnum farming on nutrient-poor temperate peatlands

Sphagnum is a typical paludiculture plant for northern bogs, which can substitute natural peat for horticulture and thus simultaneously reduce peat mining activities (Gaudig et al., 2017; Wichtmann et al., 2016). A 6-year-old sphagnum farm, established on a rewetted cutover bog without topsoil removal, was characterized at the 2 last years of the production cycle by CH_4 emissions of 2 ± 0.6 g C m^{-2} yr^{-1} and a net carbon uptake amounting to −98 ± 20.1 g C m^{-2} yr^{-1} (Beyer and Höper, 2015). Optimization of the sphagnum farm in terms of availability of labile C and water management might lead to even lower CH_4 emissions (Günther et al., 2017). Particular attention should be paid to CO_2 emissions from ditch water and from constructed dams, which have the potential for further improvements in sphagnum farming (Günther et al., 2017). One option would be to use subsoil irrigation to avoid high emissions from ditch waters (Brown et al., 2017). A constant water table was shown to be essential in providing optimal conditions for growth and carbon balances in a North American sphagnum farm (Brown et al., 2017). However, all experimental plots were CO_2 sources in the second year of establishment, which was partly caused by the decomposition of straw, which was introduced for establishing the sphagnum cultivation (Brown et al., 2017). In addition, a low water table (around 10–25 cm below peat surface) may have added to the higher CO_2 emissions from this farm.

3.2.2 Cultivation of bioenergy plants on nutrient-rich temperate peatlands

In contrast to sphagnum farming on nutrient-poor bogs, reed (Phragmites) and Thypa are common wetland plants in fens. Several studies measured GHG emissions from naturally occurring reed and Thypha sites without harvesting (Franz et al., 2016; Mander et al., 2012; Minke et al., 2016; Shurpali et al., 2009; van den Berg et al., 2016). However, no GHG measurement took place at the harvested sites (Mander et al., 2012; Minke et al., 2016; Shurpali et al., 2009). Like in other peatlands, a high water table lowered carbon emissions from peat decomposition, while on the other hand, CH_4 emissions increased up to around 100 g C m^{-2} yr^{-1} under flooding (Franz et al., 2016; Mander et al., 2012; Minke et al., 2016; Shurpali et al., 2009; van den Berg et al., 2016). To the best of our knowledge, the only study that explicitly compares the effect of biomass harvest on the carbon balance was conducted in temperate rewetted fen by comparing gas fluxes of harvested and non-harvested plots of Phragmites, Thypa, and Carex vegetation (Günther et al., 2015). Cutting above ground biomass changed the net CO_2 uptake depending on the year and vegetation type. When including the biomass export from the harvested plots, harvested paludiculture plots became in most cases a stronger carbon source compared to the non-harvest plots. However, carbon losses of these 15-year-old rewetted sites were lower compared to their drained counterpart and thus would present a mitigation option (Günther et al., 2015). Günther et al. (2015) concluded that multiple-year studies are needed to evaluate the long-term effect of harvest on the carbon balance of managed peatlands. In view of increasing the yield for bioenergy plants and intensifying cultivation in future, practices including more frequent biomass harvest and fertilization have to be carefully evaluated (Mander et al., 2012; Kandel et al., 2019a).

3.2.3 Rice as potential paludiculture crop in subtropics and tropics

Rice is a staple crop with high water tolerance. Whereas the GHG balance and CH_4 release of paddy rice on mineral soil is in general well studied, few field experiments evaluated these parameters for paddy rice cropping on organic soils. Studies in Indonesia showed that rice crops had favourable GHG balances compared to other plant species and that even if rice cultivation let to high CH_4 emissions, compared with other corps, these were overcompensated by reduced CO_2 release from peat degradation (Hadi et al., 2005; Furukawa et al., 2005). In a review of other data from SE Asia, Hergoualc'h and Verchot (2014) compared data sets from measurements of various land-use and management types on intact and drained tropical peat. Paddy rice emitted less CO_2 than croplands or acadia plantations but did not reveal an improved GHG balance over most of the land-use types when CH_4 was taken into account. Water levels were the highest with rice (−4.5 cm), whereas water levels of other land-use types were between −18 and −78 cm, resulting in partially high CO_2 emissions. Hatala et al. (2012) measured CO_2 and CH_4 fluxes on degraded peatlands managed as paddy rice or pasture using EC over 2 years in California. The water table of the pasture was −50 to −80 cm throughout the year, whereas the paddy rice was inundated most of the year but drained 45 days before the start of the growing season for cultivation and planting and for about 55 days at the end of the growing season for harvest. Their study showed that paddy rice was a net GHG sink in both years owing to a substantial CO_2 uptake, whereas the pasture released large amounts of CO_2, in addition to CH_4 from cattle, and was overall a strong GHG source. With harvest exports taken into account, paddy rice management turned into an overall GHG source which was, however, substantially smaller than the one from the pasture system.

Together these few available GHG measurements with paddy rice on organic soil reveal that CO_2 emissions from peat decomposition can be largely reduced whereas CH_4 increases. The latter has long been recognized as an issue of paddy rice cropping (Seiler et al., 1984), and CH_4 release can be reduced by measures such as intermittent flooding to keep periods with preferable conditions for methanogens short (Minamikawa et al., 2018). Paddy rice on organic soils might be a more suitable agricultural crop than many others, but optimization of its GHG balance is still an understudied topic.

3.2.4 Potential of paludiculture to reduce carbon loss

Based on the available data and mechanistic understanding, a high water table reduces aerobic peat decomposition and thus CO_2 emissions of paludiculture systems compared to management after drained peatland. However, the empirical database is too small to say how the full carbon balance changes when harvest is included and hence, if paludiculture systems can be carbon neutral or even carbon sinks. More field studies investigating the complete carbon balance of paludicultures are needed to give the scientific background towards reducing GHG emissions from drained and rewetted peatlands. In addition, site-specific management options, including technical requirements to stabilize water tables throughout the year, and the importance of topsoil removal have to be taken into account.

3.3 Engineering measures to reduce greenhouse gas emissions under sustained agricultural management

3.3.1 Water management

Due to the strong dependence of the GHG emissions on the water table, the idea of finding the optimal water table – as high as possible but at the same time maintaining crop productivity and reducing CH_4 emissions – is obvious. Evans et al. (2021) estimated that raising average water tables of organic soils from currently −90 for cropland and −60 cm for grassland by half could save 3.3 Mt CO_2 yr^{-1} (2.7–4.0 Mt CO_2 yr^{-1}) with a negligible increase in CH_4 emissions in UK peatlands. Globally, a reduction potential of 65% corresponding to 508 Mt CO_2 yr^{-1} was approximated by reducing drainage depth by half in all agriculturally used peatlands (Evans et al., 2021).

However, this technique has received little attention so far, resulting in few experimental field studies quantifying the potential of reducing carbon losses through active water management. Short-term mesocosm experiments revealed that raising the water table from the typical drainage depth of −70 cm to an optimum of −30 cm would reduce total net emissions by about 42% in grassland systems (Regina et al., 2015). Musarika et al. (2017) tried to find the best balance between peat preservation and yield of radish. Raising the water table from −50 cm to −30 cm in mesocosms not only reduced the peat mineralization by half but also improved the productivity of radish. In contrast, mineralization was reduced by 31% but also decreased the yield of celery by 19% (Matysek et al., 2019). To avoid high economic loss, the authors suggested to raise the water table only in the non-growing season. Wen et al. (2020) compared precisely this setting. While raising the water table from −50 cm to −30 cm for both, fallow period and growing season, CO_2 emissions and yield of lettuce decreased by 36% and by 37%, respectively. In contrast, a higher water table restricted to the fallow period reduced annual CO_2 emissions by 30% without affecting lettuce yield. To balance the reduction of soil subsidence and yield in sugarcane fields in the Everglades, Florida, Glaz and Morris (2010) investigated the influence of water table on yield. A constant water table of −20 cm resulted in a lower sucrose yield compared to a water table of −45 cm. However, periodic flooding of up to 14 days did not reduce the yield but showed potential to reduce soil subsidence (Glaz and Morris, 2010). In contrast, short flooding cycles could also induce higher CO_2 emissions as compared to a constant drainage depth of −30 cm (Rodriguez et al., 2021). As the major part of emission with constant drainage occurs in summertime, constant flooding restricted to the summer period reduced CO_2 emissions. The authors, therefore, suggested to implement crop rotations that allow flooding during summer to reduce peat loss. Although most studies agree that raising the water table reduces carbon emissions, Berglund and Berglund (2011) observed that raising the water table from −80 to −40 cm resulted in slightly higher CO_2 emissions. These results demonstrate that the direction and size of effects following water table manipulations depend on crop and peatland type.

One technical approach to reach the targeted water table depth is subsurface drainage, also called subsoil irrigation. The basic principle is that irrigation pipes are placed at about −70 cm in agriculturally used peatlands to drain the sites at high water tables in autumn, winter, and spring and to raise the water table in summer to reduce peat mineralization. A relative dense network of the sub-irrigation pipes as narrow as 10 m was suggested to maintain a stable water table in the whole field for a lowland UK peat soil (Kechavarzi et al., 2007). Subsoil irrigation was suggested to reduce soil subsidence from drained pastures in the Netherlands (Querner et al., 2012). The effect of this technique on carbon loss was tested at four dairy farms by Weideveld et al. (2021). However, neither substantial higher yield nor reduced peat mineralization was found in 2 consecutive years (Weideveld et al., 2021). This could be related to a relatively small increase in the water table by about 6–18 cm in the summer. Indeed, only at periods with a water table difference of >20 cm, a reduction in soil respiration was measured.

In the tropics, a reduction of drainage depth in palm oil plantations to 40–60 cm below surface was proposed as a best management practice in order to reduce soil subsidence and CO_2 emissions and thus to extend the economic life span of a drained peat area (Lim et al., 2012). Moreover, the best plant growth of oil palms was found at a drainage depth of −55 cm compared to a drainage depth of −85 and −25 cm (Hashim et al., 2019). However, the average drainage depth is often considerably lower than −55 cm. Uda et al. (2020) evaluated the potential of low-drainage food crops to not only reduce CO_2 emissions but also include profitability, scalability of market, and acceptability to farmers. While sago palm (Metroxylon sagu) and illipe nut/tengkawang (Shorea spp.)

provided the most sustainable crops, as they grew at near-surface water levels, bananas (Musa paradisiaca), pineapples (Ananas comosus), and sweet melons (Cucumis melo) yielded the highest scores in terms of scalability of market and acceptability to farmers but needed a drainage depth of at least −30 cm (Uda et al., 2020). Yet, experimental data on the carbon balance of these systems are missing.

3.3.2 Soil management and cropping

To improve trafficability and increase the yield on drained peatlands under agriculture, different strategies are applied (Blankenburg, 2015; Sognnes et al., 2006). In principle, peat is covered with mineral material – mainly sand – with a thickness of about 20–40 cm. With shallow ploughing without mixing a deeper peat layer with mineral soil, the sand remains as a covering layer on top of the peat. Another strategy is to mix the mineral material with peat in a ratio of 1:1 (Blankenburg, 2015). These strategies were often applied at sites with underlying sand, which could be directly taken as cover material. Another way of mixing the peat is deep ploughing, where soils with underlying sand layers are deeply ploughed at a maximum of 2 m to shift and turn the soil layer at 130–150°, thereby creating alternating tilted bars of sand and peat (Schindler and Müller, 1999). However, sometimes other material than sand is used as covering material, including moraine material, material from construction sites, or clayey material in coastal regions (Blankenburg, 2015; Sognnes et al., 2006; Ferré et al., 2019). Addition of mineral material consistently alters the physical properties of drained peat. Hydraulic properties change and result in increased drainage and aeration of the upper soil layers, attenuation of spring frost, increased warming in spring, reduced fire risk, increased trafficability with heavy machinery, and ultimately increased yield (Bambalov, 1999; Blankenburg, 2015; Schindler and Müller, 1999; Sognnes et al., 2006). Yet the effects of these soil treatments on soil subsidence and finally carbon loss are poorly studied. Long-term soil subsidence measurements in the Netherlands revealed that peat soils with a naturally occurring clay cover subsided less as compared to peat soils without mineral covering (van den Akker et al., 2008). Moreover, Schindler and Müller (1999) investigated the soil subsidence of a sand-mixed site during the first 10 years after establishment in Germany. After initial subsidence of 1.5 cm yr^{-1}, rates decreased to 0.3 cm yr^{-1}, which was interpreted as a reduction in SOC loss. In peatlands of north-east Germany, a mineral top layer reduced soil subsidence from 0.5 cm yr^{-1} to 0.27 cm yr^{-1} when the initial underlying peat had a thickness of 30–70 cm but increased subsidence to 1.33 cm yr^{-1} at sites with an underlying peat layer of more than 100 cm (Fell et al., 2016). In contrast, in a peat soil with sand mixing and sand coverage, both treatments showed higher carbon mineralization rates, which was explained by higher soil temperatures

(Zaidelman and Shvarov, 2000). Some recent studies suggest that the addition of sand does not principally alter the CO_2 fluxes. Beyer (2014) measured an average SOC loss of 370–650 g C m^{-2} yr^{-1} on three peatland sites covered with sand. Furthermore, organic soils mixed with sand, resulting in a carbon content of about 10%, lost SOC in a similar range as 'true' peat soils with high carbon content (Leiber-Sauheitl et al., 2014). However, Wang et al. (2022) showed significantly reduced N_2O emissions after addition of mineral soil material on a drained organic soil.

As for sand additions, field experiments in the Netherlands showed that GHG emissions from peat soils with a naturally occurring clay layer were not systematically different as compared to peat soils without clay (Weideveld et al., 2021). Based on the current knowledge, addition of mineral material does not seem to reduce carbon mineralization. Interaction of the organic material with the mineral phase does not seem to play a role, in contrast to mineral soils. However, addition of mineral soils might lead to SOC protection if it allows to keep higher water levels which are needed to protect the underlying peat.

Based on the current research, the choice of crop or soil tillage practice seems to have a minor effect on SOC loss. Norberg et al. (2016a,b) concluded from 11 field experiments on different organic soil types that differences between years and sites were in general much larger than differences between crops (cereals and row crops) grown on the same field. Comparisons of barley with grassland revealed no clear effect of crop type on CO_2 emissions (Elsgaard et al., 2012; Maljanen et al., 2001; Lohila, 2004; Lohila et al., 2003) and also Tiemeyer et al. (2020) found no clear difference between the response of CO_2 emissions to water table differences between crop- and grasslands. In addition, Taft et al. (2018) concluded from the incubation of intact soil columns of horticultural peatland that zero or minimum tillage could not reduce SOC loss. Even after cessation of cultivation practices, soil carbon loss remains high in northern peatlands when cultivation was abandoned (Kløve et al., 2010; Maljanen et al., 2010). Similarly, bare peatlands, where a vegetation cover is missing, showed slightly increased carbon loss in three northern peatlands (Maljanen et al., 2010).

3.3.3 Can afforestation reduce soil carbon loss from drained peatlands?

Under specific conditions, afforestation of shallow-drained Fennoscandia peatlands may lead to an increase in SOC stocks. Measuring the components of NECB with the EC methods showed that a naturally forested pine bog with drainage and afforestation lead to an increase on average of about 60 g C m^{-2} yr^{-1} SOC in 5 consecutive years (Lohila et al., 2011; Minkkinen et al., 2018).

Even in a severe drought year, the drained peatland remained a carbon sink (Minkkinen et al., 2018). This is in line with earlier research, based on SOC stock measurement (Minkkinen and Laine, 1998; Minkkinen et al., 1999). These results were explained with the relatively shallow drainage depth of about −30 to −40 cm in Finnish peatlands, a similar plant structure compared to undrained sites and a significantly increased tree stand growth and litter production (Minkkinen et al., 2018). In addition, in drained afforested peatland, a new secondary humus layer formed on the top of peat (Minkkinen et al., 2008). Drainage induced a decrease in pH due to oxidation processes, enhanced nutrient uptake by trees, and decreased enzyme activity. In addition, soil temperature decreased in the long-term, caused by the decrease of thermal conductivity in the drier surface peat and increasing shading by growing trees (Minkkinen et al., 2008). It was shown that drier surface conditions at the drained peatlands lead to retarded decomposition of pine needles and fine roots compared to undisturbed peatlands (Laiho et al., 2004). However, this SOC increase is restricted to nutrient-poor sites, whereas nutrient-rich forests were net carbon sources (Ojanen et al., 2013). Also Meyer et al. (2013) observed that a spruce-dominated fertile organic soil on a former agricultural land in southwestern Sweden represents a small carbon sink during peak forest production but would become a carbon source in the long run. Laine et al. (1996) evaluated the climatic impact of drainage in nutrient-poor and -rich southern Finnish peatland by modelling the change in radiative forcing. Drainage in nutrient-rich sedge fen lead to a cooling effect due to the low CH_4 emission, although the peat layer turned from a C sink to a C source, but increased CO_2 emissions would have generated a warming effect over longer time scales. A similar, but smaller, effect was predicted for bogs (Laine et al., 1996).

The assessment of temporal dynamics through the full forest production cycle is important, as GHG fluxes change with stand age. Forest clearance comes with soil preparation and drastic change in site conditions. Directly after clear-cutting, high CO_2 emissions of around 840 ± 40 g C m^{-2} yr^{-1} were measured for the first and 565 ± 34 g C m^{-2} yr^{-1} for the second year in a nutrient-rich boreal peatland in southern Finland (Korkiakoski et al., 2019). These high CO_2 emissions were explained with a high rate of decomposition of peat and logging residues, which were left on site. It was estimated that about 49% of total ecosystem respiration was attributed to the decomposition of logging residues. In addition, GPP was sharply reduced because of the removal of assimilating trees and a declining understory. Reduction of evaporation led to a rise in the water table of about 20 cm, which reduced the volume of aerated peat and likely reduced decomposition rates of peat. However, increased soil temperatures may have an inverse effect. Recovery of ground vegetation in the second year and decreased decomposition of logging residues caused a reduction in CO_2 emissions by about 41%.

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Unlike the Fennoscandia-afforested peatlands, considerably fewer data exist for other global regions, and it was questioned if results from northern peatlands can be transferred to afforested peatlands under different climate and management conditions in UK and Ireland (Sloan et al., 2018). In contrast to relative shallow-drained Fennoscandia afforestation, closely spaced plough furrows between deeper drainage is a common soil preparation in the UK, which may also influence carbon mineralization (Sloan et al., 2018). Mature forests in eight sites in Ireland represented soil carbon sources of about 60–300 g C (Jovani-Sancho et al., 2021). Using a snap-shot measurement and modelling, Hargreaves et al. (2003) estimated that the ecosystem of a Scottish peatland conifers acts as a net source of about 200–400 g C m^{-2} yr^{-1} in the first years after ploughing. After 4–8 years when trees started to grow it became a sink of about −300 g C m^{-2} yr^{-1} that was further increased up to −500 g C m^{-2} yr^{-1} in the following years when tree growth dominated. After accounting for the carbon stored in the biomass, the soil appeared to lose carbon at a rate of about 100 g C m^{-2} yr^{-1}. Similarly, a temperate 44-years-old spruce afforestation on a German bog was a carbon sink of about −157 ± 36 g C m^{-2} yr^{-1} (Hommeltenberg et al., 2014). However, when accounting for SOC loss since plantation and average rotation length of 60 years, the site was found to be a strong SOC source. Quantifying the carbon balance over a whole production cycle for forests is challenging but unavoidable for the robust estimate of the effect of peatland afforestation on the carbon budget. Up to date, available data suggest that – apart from nutrient-poor bogs in Fennoscandia – many afforested peatlands are still considerable carbon sources, especially in the tropics.

3.4 Metrics to evaluate reduced CO_2 emissions versus increased CH_4 emissions after rewetting

The balance between CO_2 sinks and CH_4 emissions strongly determines the net climatic impact of rewetted peatlands, and these two GHGs show a strong and inverse dependence on water table depth. To assess the climatic impact of rewetting activities through ecosystem GHG exchange, the differing radiative properties and atmospheric lifetimes of GHGs need to be accounted for. Compared to CO_2, CH_4 has a much larger radiative efficiency but a much shorter lifetime of 9–12 years (Myhre et al., 2013). The huge difference in lifetime leads to strongly time-dependent climatic effects. A steady emission of CH_4 at constant rates lead to rapid increase of atmospheric CH_4 that stabilizes after a few decades when natural atmospheric removal of CH_4 balances out ongoing emissions (Lynch et al., 2020). Correspondingly, the radiative forcing by CH_4 emissions rapidly increases due to the high radiative efficiency of CH_4 but then largely stabilizes. In contrast, continuous emissions of CO_2 lead to steady increases of atmospheric CO_2 concentrations and a steady increase of

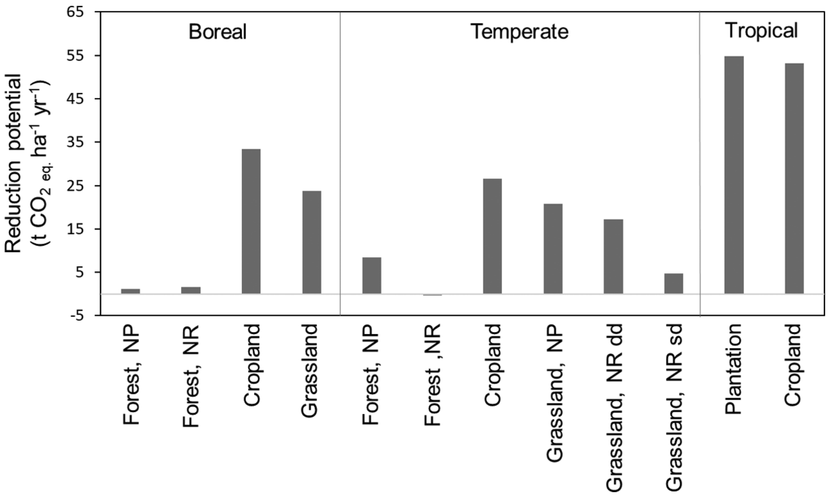

Figure 8 Emission reduction through rewetting for different land use and climatic zones (data from Wilson et al., 2016a). NR, nutrient-rich; NP, nutrient-poor; dd, deep-drained (>30 cm); sd, shallow-drained (<30 cm).

radiative forcing as long as the emissions sustain (Lynch et al., 2020) because of the long atmospheric lifetime of CO_2. In consequence, CO_2-induced radiative forcing is strongly depending on the cumulative emissions, while CH_4 emissions contribute predominately as a function their emission rate (Allen et al., 2016).

The most commonly used metric global warming potential (GWP) integrates the radiative forcing of a GHG emission pulse over a prescribed period, often 20 or 100 years. Using the GWP, non-CO_2 GHG fluxes are converted to a common unit called CO_2 equivalent (i.e. the cumulative radiative forcing of that gas relative to that of CO_2). Owing to the different lifetimes of the GHGs', the GWP is time-dependent. While this measure is convenient and widely adopted, it does not properly reflect the radiative forcing of quasi-continuous emissions and uptakes of GHGs' from peatlands. To overcome this shortcoming, two approaches were suggested. First, the GWP of sustained emissions and removals, also called sustained flux GWP, captures more precisely the time-dependent radiative forcing of short-lived GHGs (Allen et al., 2018; Lynch et al., 2020; Neubauer and Megonigal, 2015). It is able to relate the change in the emission rate of a short-lived GHG such as CH_4 rather than the emission rate itself to the radiative effect of CO_2. Yet, Lynch et al. (2020) encouraged to report the radiative forcing effect of each GHG separately as the most transparent way for GHG reporting and communication. Despite these recent advantages in metrics, the most straightforward approach for describing the temperature effect of GHG emissions and removals is to report the radiative forcing over time either as instantaneous or cumulative forcing.

This is particularly meaningful because emission and removal rates may change during, for example, rewetting, which limits the suitability of static approaches. Calculating the change in radiative forcing of peatlands over time, based on GHG fluxes, has been first done by Frolking et al. (2006) who showed that 'the current radiative forcing of intact peatlands is determined primarily by a trade-off between the total C sequestered since the peatland's formation and the recent methane fluxes' and that the net CO_2 sink of an intact peatland outweighs its CH_4 emissions after some centuries or millennia. The time after which a net cooling (cumulative radiative forcing becomes negative) is achieved can be up to 14 000 years for intact wetlands as shown by Neubauer (2014). This point in time, termed radiative forcing switchover time, is strongly and non-linearly related to the ratio of CO_2 uptake to CH_4 release (Frolking et al., 2006; Neubauer, 2014). Another way of approaching the warming versus cooling effect of peatlands is to look at the instantaneous, not the cumulative, forcing. Ojanen and Minkkinen (2020) showed that, by calculating instantaneous radiative forcing, rewetting offers climate benefits for tropical and agricultural peatlands but not for peatlands previously drained for forestry because of the different balance of CO_2 emission savings versus rewetting-induced CH_4. Despite long switchover times, rewetting peatlands reduces their instantaneous radiative forcing already after a few decades (Günther et al., 2020).

In this chapter, we have followed the conventional GWP notion as it is still the most used metric to compare the climatic impact of different GHGs. Based on the difference of IPCC default values of drained peatlands and an updated default value for rewetted peatlands, Wilson et al. (2016a) calculated the emission reduction through rewetting for different climate regions and land uses, following the IPCC classification (IPCC, 2014; Fig. 8). The magnitude of the reduction potentials is displayed by climate zone, land-use intensity, and drainage depth. The highest reduction potential is located in tropical peatlands, followed by boreal and temperate peatlands used as croplands, and boreal and temperate peatlands under grass. In almost all cases, rewetted peatlands turn into CO_2 sinks whereas CH_4 emissions increase and N_2O becomes negligible. Based on the 100-years GWP applied by Wilson et al. (2016a), the increased CH_4 is the dominant factor for the overall GWP after rewetting for all climate and land-use combinations. Yet, in all combinations except under nutrient-rich temperate forests, rewetting provides a climate benefit. Because CH_4 plays a relatively important role when GWP100 is used, the climate benefit will likely increase over longer time scales.

3.5 Outlook

In the course of the discussion about carbon sequestration potentials in soils, scientific attempts of how to restore the C sink or to reduce the climatic impact of organic soils and finally to implement them in climatic mitigation strategies are rapidly emerging (e.g. Günther et al., 2020; Humpenöder et al., 2020; Leifeld and Menichetti, 2018; Paustian et al., 2016; Tanneberger et al., 2020). Rewetting and restoration of natural peatlands is the main option to avoid SOC losses, while at the same time increasing CH_4 losses and economic losses as main trade offs. The first major rewetting activities occurred in the temperate and boreal zone at sites, which were not intensively used before. Therefore, research up to now has focused on little or comparably moderately degraded peatlands, which were rewetted with the aim of re-establishing the origin habitat conditions or of restoring peat extraction sites (Wilson et al., 2016b). Owing to their high emissions, the demand for mitigation and rewetting activities shifted to more intensively used peatlands as well (Harpenslager et al., 2015; Tiemeyer et al., 2020), emphasizing the need to investigate the GHG dynamics following rewetting activities in these formerly intensively used sites with a high potential of CH_4 and N_2O emissions. Rewetting such sites has a high conflict potential between economic revenue and GHG reduction potential (Ferré et al., 2019). Further investigations are needed to develop sustainable management strategies for the agricultural use of peatlands where restoration is not an option due to hydrological restrictions or the socio-economic constraints. Paludiculture was recently introduced as an agricultural practice on rewetted organic soils to achieve biomass production with water tolerant species such as sphagnum mosses, reed, and cattail.

Reducing high carbon loss from peat decomposition and fires in tropical drained peatlands by rewetting offers a substantial mitigation potential (Ojanen and Minkkinen, 2020). Yet, implementation is challenging because of the high pressure to clear land by fire for agriculture or plantations (Budiman et al., 2020), as well as owing to the very high costs of re-establishing ecosystem services (Hansson and Dargusch, 2017). To overcome this land-use conflict, more research about tropical paludiculture species and implementation strategies is urgently needed (Giesen, 2021), particularly in the light of the tremendous and tragic failure of converting peat swamp forests to rice production in Indonesia in the 1990es (Goldstein, 2015).

The need for paludiculture research likewise applies to temperate agriculture, as rewetting offers also rapid climatic benefits (Ojanen and Minkkinen, 2020). In addition, technical measures such as combining soil coverage with active water management enhancing SOC protection and reducing GHG emissions are worth to be investigated further. Finally, for the mechanistic understanding and modelling of carbon dynamics in drained and rewetted peatlands, more long-term field studies are needed that integrate different phases of rewetting and production cycles for various peatland types and historical uses. These data, together with drainage depth and peat quality, are necessary to further develop existing peatland models (e.g. Bona et al., 2020; Frolking et al., 2010; Premrov et al., 2021) as tools for evaluating mitigation options.

4 Where to look for further information

There are several societies, non-profit organizations and networks aiming at the conservation and restoration of peatlands, their sustainable management, interdisciplinary research on the functioning of these ecosystems, and knowledge transfer to policy makers and the society. Below is a selection of the most relevant webpages (all webpages were accessed on 25/4/2022).

- The International Peatland Society (IPS, https://peatlands.org) was founded in Canada in 1968. Its mission is to serve all those involved in peatlands and peat through the promotion, gathering, exchange and communication of knowledge and experience, by means of events and projects, which address key issues, including climate change, biodiversity, the need for responsible use and restoration.
- The International Mire Conservation Group (IMCG, http://www.imcg.net/) is an international network of specialists promoting conservation and exchange knowledge of mires and related ecosystems. Together with the IPS it publishes the peer-reviewed journal Mires & Peat (http://www.mires-and-peat.net/), covering all aspects of peatland science, technology and wise use.
- The aim of the Global Peatland Initiative (https://www.globalpeatlands.org/) is to save the carbon stored in peatlands by improving the conservation, restoration and sustainable management of peatlands.
- Ramsar Convention on Wetlands of International Importance especially as Waterfowl Habitat (https://www.ramsar.org/) is an international convention for the conservation and sustainable use of wetlands.
- Wetlands International https://www.wetlands.org/ is a global not-for-profit organization dedicated to the conservation and restoration of wetlands.
- The IUCN UK Peatlands Programme (https://www.iucn-uk-peatlandprogramme.org) is a national program to promote peatland restoration in the UK. The Program advocates the multiple benefits of peatlands through partnerships, science, policy and practice. The aspect of climate change mitigation is a central theme.
- The Greifswald Mire Center (https://www.greifswaldmoor.de/home.html) performs interdisciplinary research, provides policy makers and society with knowledge and imparts theoretical and practical knowledge on climate protection, biodiversity and sustainable use of peatlands. As one special research focus, they study paludiculture.

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