Effect of re-wetting on greenhouse gas emissions from different microtopes in a cut-over bog in Northern Germany

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List of Abbreviations and Symbols

µg	microgram
µm	micrometer
µmol	micromol
r	Pearson correlation coefficient
AIC	Akaike Information Criterion
C	carbon
C/N	carbon nitrogen ratio
CH_4	methane
CO_2	carbon dioxide
CO_2 eq	carbon dioxide equivalents
D	bare ditch refilled with peat
DV	ditch refilled with peat with Eriophorum vegetation
DN	dissolved nitrogen
DOC	dissolved organic carbon
E	extraction on-going site
GC	gas chromatograph
GHG	greenhouse gas
GTP	global temperature change potential
GWP	global warming potential
ha	hectare
L	liter
mg	milligram
N	nitrogen
Pg	petagram
PD	dry peat dam site
R_{eco}	ecosystem respiration
R	rewetted in 2009 site
RV	rewetted in 2004 site with vegetation
SD	standard deviation
T	temperature
WC	water content
WHC	water holding capacity
WRB	World Reference Base
WTL	water table level
yr	year

Definitions

Bog (ombrogenic peatland) – an acidic and nutrient-poor mire which obtains water from the atmosphere in form of rain and snow.

Fen (minerogenic peatland) – a mire whose hydrological inputs are groundwater as well as rainfall. This type of peatland is more nutrient-rich and more alkaline than a bog.

Flooding – an overflow of water onto a land area that is usually dry.

Global Temperature Change Potential – a relative value calculated for different greenhouse gases to compare their contribution to global temperature change to carbon dioxide, which was chosen as reference gas.

Global Warming Potential (GWP) – a relative value calculated for different greenhouse gases to compare their contribution to global warming with reference to carbon dioxide, which was chosen as reference gas and has a GWP value of 1.

Greenhouse gas – a gas that can absorb or emit radiation energy in the atmosphere in the thermal infrared range, and has a strong influence on radiative forcing.

Mire – a wetland area with mostly moss vegetation, peat-forming plant cover and peat accumulation on organic or mineral substrates. Includes swamps, bogs, fens and peatlands.

Peatland – any terrain ecosystem with a peat depth of at least 30 cm. Includes wetlands as well as disturbed organic soils, and can be natural, drained, afforested or rewetted.

Restoration – the technical challenges and management procedures for the conservation of damaged peatlands using hydrological- (rewetting), chemical- (nutrient availability monitoring), and structural techniques, as well as microclimatic controls and revegetation.

Rewetting – the technical procedure of blocking drainage ditches or constructing peat dams in order to prevent water losses and to restore the natural ecosystem functions of a drained peatland.

Wetland – a land area that is inundated or saturated by water, with typical aquatic vegetation and a water table close to/above the soil surface for a short-/long-term period. Includes ecosystems built on organic and mineral substrates, flowing and shallow waters, and most undisturbed peatlands (Charman, 2002).

Summary

Under natural conditions, peatland ecosystems play an important role as a long-term net sink of carbon and nitrogen, storing 445-550 Gt of global carbon. Human activities, such as forestry or peat extraction, are known to affect peatland ecology and hydrology. These practices often lead to drainage, resulting in peat oxidization and enhanced greenhouse gas emissions (GHG) as long as the peatland area remains drained. Currently, the rewetting of drained peatlands is a common practice with the aim of re-establishing near-natural conditions and reducing peatland GHG emissions, especially of CO_2 and N_2O, which are enhanced under peatland degradation and the aeration of peat.

At the cut-over drained bog, Himmelmoor (Quickborn, Germany), peat extraction started in 1780, and since the 1930s extraction has being occurring at a rate of about 38,000 m^3 yr^{-1}. Rewetting started stepwise here in 2004 by blocking drainage ditches with peat as well as creating polders surrounded by peat dams, and currently, more than 56 % of the former extracted central area has been rewetted. In this research project, five study sites in the Himmelmoor with different microtopes, land-use history, vegetation cover, and water table level were examined: a peat extraction area (E), a ditch refilled with peat (D), a peat dam (PD), and areas rewetted in 2004 (RV) and 2009 (R). In order to compare GHG emissions from rewetted and extraction areas, chamber measurements of CO_2, CH_4, and N_2O fluxes were performed in the field over a two year period (2014-2015) in weekly to biweekly intervals. To complement these measurements, peat samples from rewetted and extraction areas were incubated under oxic and anoxic conditions in order to investigate their CO_2 and CH_4 production potentials.

The results show that all study sites act as GHG sources, although large differences were identified between sites. For example, the total mean greenhouse gas emissions* over the two-year measurement period varied from 4.7 ± 1.8 t CO_2-eq. ha^{-1} yr^{-1} at the rewetted site (R) to 38.1 ± 6.8 t CO_2-eq. ha^{-1} yr^{-1} at the peat dam site (PD). However, in the longer term, it was found that the strong warming impact of GHG emissions from the extraction area had been reduced by more than 35 % only five years after rewetting. At all study sites, annual GHG emissions were dominated by CO_2, which contributed between 50% and 74% of the mean annual emissions across the various sites. For a 1-hectare area, the annual emissions for rewetted sites were estimated at 123 ± 43 kg CH_4, 3.01 ± 1.1 kg N_2O, and 2.4 ± 0.8 t CO_2; for ditches at 130 ± 16 kg CH_4, 7.8 ± 3.3 kg N_2O, and 6.6 ± 1.4 t CO_2; and for drained bare sites (including peat dams) at 17.7 ± 10.8 kg CH_4, 20.6 ± 4.3 kg N_2O and 16.0 ± 2.9 t CO_2. CO_2 and N_2O emissions from the dry bare peat dam areas were larger than previously observed for other harvested and drained bare peat sites in Europe. The estimated area-weighted mean annual GHG emission for the former extraction area and for the 74 ha investigation area as a whole

* Greenhouse gas emissions – emissions of gas with each considered GHG weighed by its Global Warming Potential (GWP) for a 100-year period.

amounted to 10.5 and 13.9 t CO_2-eq ha^{-1} yr^{-1}, respectively. Findings from the incubation experiments are in agreement with those from the chamber studies, with the mean CO_2 production potential (for the top 1 m soil) of peat from the vegetated rewetted area incubated under anoxic conditions being considerably smaller (2.3 t CO_2 ha^{-1} yr^{-1}) than that observed in incubated peat material from the drained extraction area incubated under oxic conditions (8 t CO_2 ha^{-1} yr^{-1}).

In terms of the success of rewetting practices, soil processes at sites that were rewetted 5- or even 25 years ago remain affected by previous management, and have not been fully restored to natural conditions. This practice was, however, found to have a significant effect on microbial activity and microbial biomass, as indicated by the microbial C and N concentrations in the upper horizons, which were 25-70 % higher in the rewetted bare and vegetated plots than in the drained plots. The findings of this study suggest that the full restoration of soil conditions as well as mire-typical microbial activity after ditch closing and rewetting will take at least decades to occur. The results of this study have been used to provide suggestions to improve restoration practices and enhance ecosystem services, such as reducing the height of dams.

Zusammenfassung

Unter natürlichen Bedingungen spielen Moor-Ökosysteme eine wichtige Rolle als langfristige Kohlenstoff- und Stickstoffsenken, die fast 25% des gesamten globalen Kohlenstoffs im Boden speichern. Die menschlichen Interventionen, wie z.B. die Forstwirtschaft oder die Torfgewinnung, beeinflussen die Moorhydrologie und die Ökologie und führen zur Drainage, was zu Emissionen der Treibhausgasen (THG) führt. Zurzeit werden entwässerte Moore renaturiert mit dem Ziel die natürlichen Bedingungen wiederherzustellen, indem permanent hohe Wasserstände im Torf zurückgebracht werden um die Treibhausgasemissionen, insbesondere CO_2 und N_2O, zu verringern.

Im Himmelmoor (Quickborn, Deutschland), eines der größten Hochmoore in Schleswig-Holstein mit einer Torfgewinnung von ca. 38.000 m^3 seit 1780, begann die Wiedervernässung im Jahr 2004 durch die Schließung von Drainagegräben mit Torf. Heutzutage sind bereits mehr als 56% der ehemaligen Abbaufläche wiedervernässt. Das untersuchte Forschungsgebiet besteht aus fünf Standorten mit unterschiedlicher Vegetation, Wasserständen und Geschichte der Landnutzung: das Abbaugebiet (E), mit dem Torf geschlossene "Pütten" (D), Torfdamm (PD), Wiedervernässungsfelder 2004 (RV) und 2009 (R). Um die Wiedervernässungs- sowie Abbauflächen zu vergleichen wurden von 2014 bis 2015 zwei Jahre lang im wöchentlichen bis zweiwöchentlichen Rhythmus im Feld Haubenmessungen zu CO_2, CH_4 und N_2O durchgeführt sowie die Torfproben aus dem Oberboden oxisch und anoxisch inkubiert mit dem Ziel die Produktionspotentiale von CO_2 und CH_4 zu bestimmen.

Die Ergebnisse zeigen, dass alle Standorte Treibhausgasquellen sind und die gesamten jährlichen THG Emissionen* für den zweijährigen Zeitraum variieren von 4,7 ± 1,8 t CO_2-Äquivalent ha^{-1} $Jahr^{-1}$ für wiedervernässte Flächen bis zu 38,1 ± 6,8 t CO_2-eq ha^{-1} $Jahr^{-1}$ für trockene Torfdämme. Es wurde auch festgestellt, dass nach 5-Jahren die starke Klimawirksamkeit des Abbaugebiets, bezogen auf die THG Emissionen gewichtete gemäß GWP, durch die Überstauung um mehr als 35% verringert wurde. Die durchschnittlichen jährlichen Emissionen wurden an allen Standorten von den CO_2-Flüssen (50% bis zu 74%) dominiert und für eine Fläche von 1 Hektar kalkuliert als: 123 ± 43 kg CH_4, 3,01 ± 1,1 kg N_2O und 2,4 ± 0,8 t CO_2 für Wiedervernässungsfelder; 130 ± 16 kg CH_4, 7,8 ± 3,3 kg N_2O und 6,6 ± 1,4 t CO_2 für Püttenfläche; 17,7 ± 10,8 kg CH_4, 20,6 ± 4,3 kg N_2O und 16,0 ± 2,9 t CO_2 für drainierte Standorte und Torfdämme. Die CO_2- und N_2O-Emissionen aus den trockenen Torfdämmen waren größer als Werte für andere drainierte und trockene Moore beschriebene früher. Die geschätzte mittlere jährliche THG-Bilanz für das ehemalige Abbaugebiet und für das gesamte Untersuchungsgebiet von 74 Hektar betrug somit 10,5 und 13,9 t CO_2-eq pro Hektar, oder 1029 t CO_2-eq pro Jahr. Ähnlich wie bei den Feldmessungen zeigen die Ergebnisse von den Inkubationsexperimenten, dass das mittlere CO_2-Produktionspotential der

* THG Emissionen – Emissionen des Gases bewertet und gewichtet gemäß seinem CO_2-Äquivalent oder Treibhauspotential (GWP) bei einem Zeithorizont von 100 Jahren

Wiedervernässungsfläche mit Vegetation unter anoxischen Bedingungen 2,3 t CO_2 ha^{-1} $Jahr^{-1}$ beträgt, während das Abbaugebiet ein Potenzial von durchschnittlich 8 t CO_2 ha^{-1} $Jahr^{-1}$ aufweist.

Mehr als 25 Jahre nach Schließung der Gräben und 5 Jahre nach Beginn der Wiedervernässung sind die Bodenverhältnisse im Himmelmoor noch stark von der vorherigen Nutzung betroffen und wurden bisher nicht wiederhergestellt. Es wurde festgestellt, dass die Wiedervernässung einen signifikanten Effekt auf mikrobielle Aktivität sowie C- und N-Konzentrationen hat, die um 25-70% höher waren als in den entwässerten Mikrotopen. Auf der Grundlage dieser Erkenntnisse ist zu erwarten, dass die vollständige Renaturierung der Bodeneigenschaften nach Grabenschluss und Überstauung Jahrzehnte dauern kann. Die Ergebnisse dieser Untersuchung bilden die Grundlagen für weitere, verbesserte Praktiken zur Renaturierung von Mooren, nicht nur im Himmelmoor, sondern auch in anderen europäischen Mooren und Moorökosystemen im gemäßigten Klima weltweit, die in Bezug auf Klimawandel impliziert werden können.

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1 Introduction and objectives

Global peatlands are receiving an increasing amount of attention in scientific and political discussions related to human activity and climate change. One of the most important aspects to assess here is the contribution of different natural and anthropogenic sources of greenhouse gases (GHG) to global radiative forcing, as well as their possible response to on-going climatic warming. In this area of research, peatland ecosystems are considered a unique natural reservoir of carbon and fresh water (Vompersky, 1994). In general, peatlands are wetlands which are characterized by peat, which is organic soil material consisting dead Sphagnum mosses or other plants forming and accumulating peat. They store around 455-550 Gt of carbon, which is equivalent to twice that stored in the world's forest biomass (Kivinen and Pakarinen, 1981; Gorham, 1991; Bridgham et al., 2006, Parish et al., 2008). Covering only 2.7 % of the planet's terrestrial land surface (Oertel et al., 2016), peatland ecosystems provide numerous ecosystem services: having regulative functions in many biogeochemical cycles, provisioning fresh water as well as regulating its quality, and influencing climate by storing a significant amount of carbon (Solantie, 1999; Joosten and Clarke, 2002; Vasander et al., 2003; Landry and Rochefort, 2012; Charman et al., 2013). Drained or anthropogenically disturbed peatland areas are becoming major and increasing sources of carbon dioxide (CO_2) and nitrous oxide (N_2O) as well as decreasing sources of methane (CH_4), which are all GHGs. Human activities such as peat extraction for fuel, peat drainage, peat mixing and drying act to reverse the process of peat accumulation, increasing the release of soil carbon and nitrogen to the atmosphere and the hydrosphere. Globally, peatland drainage leads to the emission of 2 - 3 Gt CO_2 per year (Couwenberg, 2009), and previous research has suggested that the rapid drainage of peatlands and consequent soil subsidence and decomposition (Cuenca and Hanssen, 2008) result in significant GHG emissions (Hooijer et al., 2010; Brouns et al., 2015; Wigena et al., 2016). However, Novikov (2003) showed that overall, the long-term anthropogenic use of peatland areas with gentle drainage leads to lower GHG emissions. In general, peatland rewetting and restoration can have a significant impact on the GHG balance at these sites, and may return these disturbed areas to acting as a carbon sink.

In the context of climate change mitigation research, natural and degraded peatlands have been investigated and monitored in order to determine the impact of management practices on GHG emissions (Christen et al., 2016; Jordan, 2016; Wilson et al., 2016). However, there are still many aspects that require further research. For example, Koebsch et al. (2013; 2015) reported that at German degraded fen, peat plant communities which were not adapted to flooded conditions suffered, causing a reduction in CO_2 fluxes and an increase in CH_4 emissions during the first year of flooding. Similar findings have also reported in previous studies, where rewetted peatlands were identified to have lower CO_2 emissions (by 1 to 28 %) and significantly larger CH_4 emissions than their drained counterparts (Best and Jacobs, 1997; Komulainen et al., 1998; Waddington and Day, 2007). Although only relatively few studies have investigated the impact of rewetting on N_2O emissions, research suggests that this practice causes an overall reduction in N_2O emissions (Jordan et al., 2016; Wilson et al., 2016), with Davidsson et al.

(2002) for example finding a summer emissions reduction of around 7.1 mg N_2O m^{-2} d^{-1}. Overall, although existing research provides evidence of a reduction in CO_2 and N_2O emissions after rewetting, studies examining the impact of rewetting at large peat extraction areas are lacking, particularly for those with extensive peat dams. Furthermore, to date there are no published studies that have simultaneously measured the emissions of all three GHGs from peat dams, and hence there is a lack of data available about the overall balance of the rewetted sites including dams. In general, peatland rewetting leads to changes in hydrology, oxygen availability and soil redox potential, and possibly to the re-establishment of mire-typical soil microbiological and vegetation communities (Weltzin et al., 2000); however, there are still open questions regarding the re-vitalization of microbial activity after rewetting and the extent of its overall impact on the three GHG fluxes. In the field, closed chambers are the most widely employed method for process-based studies examining peatland GHG fluxes and their controlling factors. Laboratory incubation experiments can also be used to study the effects of hydrology, soil temperature, oxygen availability and other soil parameters on GHG fluxes, and is one of the most effective methods for investigating GHG production potentials (Borken and Matzner, 2009; Eickenscheidt et al., 2014; Garcia-Marco et al., 2014). Long-term research using incubation methods has shown that the highest GHG production potential is often observed under oxic conditions in the upper organic horizon and usually decreases with depth (Knoblauch et al., 2013). Although incubation results cannot be directly compared to *in situ* soil conditions and GHG fluxes, they do demonstrate the importance of different land use treatments for organic matter decomposition as well as GHG production.

The rewetting process currently implemented in the cut-over drained bog Himmelmoor (Quickborn, Northern Germany) offers a unique case study to monitor the GHG emissions before, during, and after rewetting. Peat extraction began at this site in around 1780, and since the 1930s yearly extraction rates of around 30-38,000 m^3 of peat have been recorded. The rewetting process started in the Himmelmoor in 2004 and was implemented by blocking drainage ditches with peat and creating polders surrounded by peat dams, which retain rainwater in the peatland by reducing runoff. Due to the high spatial variability of land use, hydrology and soil properties in strongly degraded peatlands under ongoing restoration, a complete assessment of GHG emissions from such areas requires data from a wide range of these sites with different properties.

The goal of this study is to investigate the range, seasonal variability, and meteorological controls of CO_2, CH_4 and N_2O fluxes from different common microtopes in a partly rewetted cut-over bog area. The main objectives of this study are: 1) to determine the impact of water level changes and rewetting on annual GHG emissions from the Himmelmoor bog; 2) to assess the capacity for restoration of typical ecosystem functions and the warming effect of the implemented rewetting methods; 3) to improve understanding of the effects of rewetting on biogeochemical soil processes; and 4) to analyze the spatial variability of GHG emissions due to different land and plant cover changes connected to the rewetting activity.

Validation of the five hypothesis groups listed below will help to provide a better knowledge base for the future application of rewetting and conservation management techniques for the Himmelmoor and other degraded northern peatlands:

H1: Five years of peat flooding leads to significantly lower CO_2 (**H1a**) and N_2O (**H1b**) emissions from formerly extracted sites and bare peat soils, whereas CH_4 emissions are significantly increased (**H1c**). Re-wetting leads to a significantly reduced (although still positive) radiative forcing due to the GHG emissions (**H1d**).

H2: Peat dams are hotspots of CO_2 (**H2a**) and N_2O (**H2b**) emissions, and strongly reduce the overall climate mitigation potential of the rewetting activity (**H2c**) in the first ten years of the re-wetting project.

H3: Ditches re-filled with peat are hotspots of N_2O emissions at bare sites (**H3a**), whereas they are hotspots of CH_4 emissions at sites vegetated by vascular plants (**H3b**).

H4: The aerobic and anaerobic microbial production potentials of CO_2 (**H4a**) and CH_4 (**H4b**) of organic material at recently abandoned but still drained peat extraction sites are low compared to both pristine peatlands and re-vegetated peatlands under restoration.

H5: Low CO_2 production potentials of peat can be related to (i) a low supply of fresh and easily decomposable soil organic matter (**H5a**) and (ii) to low soil microbial activity (**H5b**).

2 Background

2.1 The importance of peatland ecosystems in the global C and N cycle

With an area of 4 million km^2 or about 2.7% of the global terrestrial surface (Oertel et al., 2016), peatland ecosystems, probably the most widespread group of organic wetlands, are unique natural environments which are recognized as major regulators of the global carbon and nitrogen cycles. Peatlands play a key role in many biogeochemical cycles, in the preservation of biodiversity of plants and animals, as well as in the freshwater cycle (Charman et al., 2013). It is estimated that peatland ecosystems contain between 450-550 Gt of carbon (Kivinen and Pakarinen, 1981), which is equivalent to approximately 30% of the global soil carbon accumulated in organic soils in the form of the peat (Gorham, 1991; Parish et al., 2008). Furthermore, research suggests that peatlands in the northern hemisphere alone contain between 242 and 486 Pg C (Roulet et al. 2007, Kleinen et al. 2012). Natural peatlands can mitigate climate extremes due to their ability to act as an atmospheric carbon sink (Aurela, 2002) as a result of organic matter decomposition under anoxic conditions, but, they can also act as a source of methane (CH_4). According to Laiho (2006), 2 to 16% of the net primary production of the plant biomass can be transformed into plant residues (peat) and accumulated for long-term periods (Vompersky, 1994).

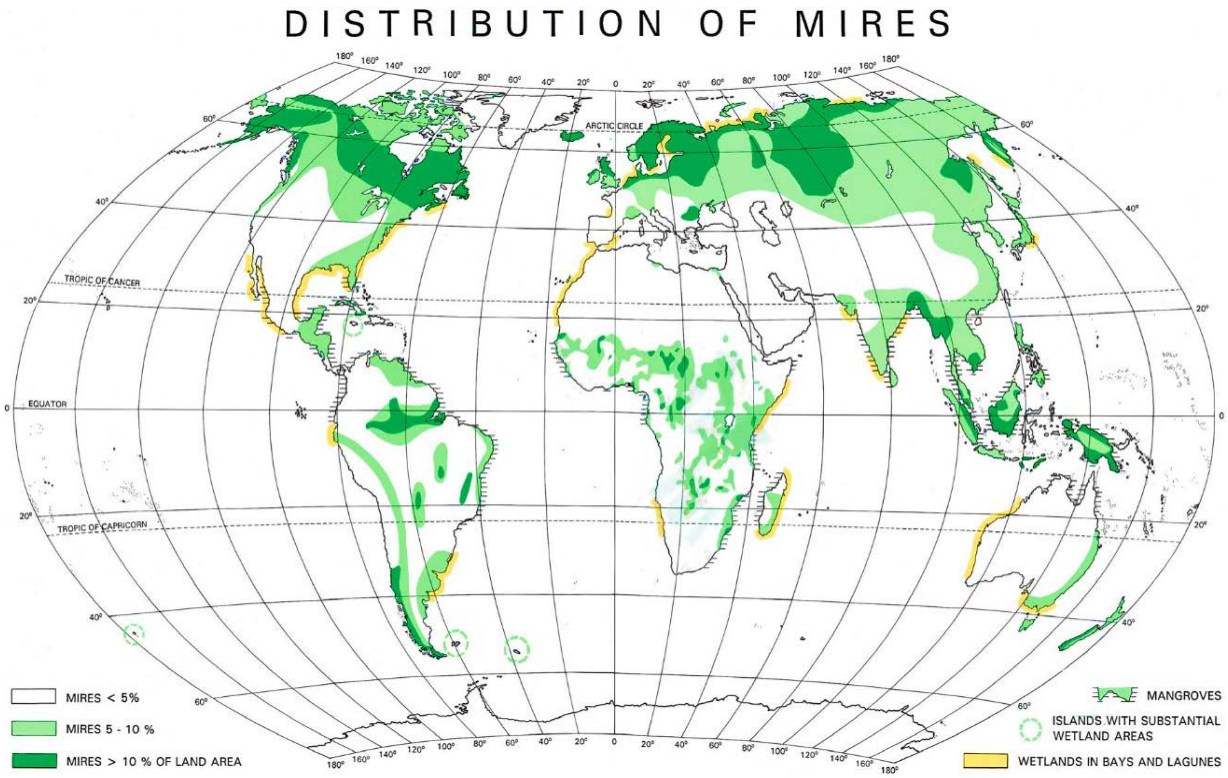


Figure 1. Distribution and location of global mires and peatlands (Lappalainen, 1996).

In general, most peatlands consist of two structural layers: (1) the oxygen-rich upper acrotelm with living moss, considered as the active layer where the rate of decay is high; and (2) the

deeper oxygen-poor catotelm. However, there are also exist peatlands where the upper oxygen-rich layer is less well developed or absent and only the catotelm layer remains (Ingram, 1978). The acrotelm is mainly composed of Sphagnum or brown mosses, which grow on the slightly decomposed substrate below (Romanov 1968, Schouwenaars 1993) and is characterized by periodically aerobic (oxic) conditions, and where high hydrological conductivity and high porosity in the sediment allow for little capillary rise effect. Below the waterlogged catotelm, peat is actively formed and accumulates due to constant anaerobic conditions and low hydraulic conductivity, leading to peat preservation (Moore 1989). But, the hydraulic conductivity changes in different peat types and increases with decreasing degree of decay, as well as less compacted plant fragments (Eggelsmann et al, 1993).

As far as the role of water in peat formation is concerned, two main processes are known, namely terrestrialisation and paludification. Both of these are determined by specific environmental factors (autogenic) such as topography, hydrology, substrate and vegetation type and external climatic (allogenic) conditions (Foster and Wright 1990, Anderson et al. 2003). The former begins with peat development in open water areas due to plant invasion, for example by forming peat Sphagnum mosses; and the latter is an accumulation of peat directly over a mineral soil (Parish et al. 2008). During the formation of a peatland, terrestrialisation can be followed by paludification (Anderson et al. 2003). On average, peat in natural peatland ecosystems tends to grow at a rate of 20-30 g m^{-2} annually, corresponding to a long-term C accumulation of 10 – 40 x 10^3 kg C km^{-2} a^{-1} (Parish et al, 2008).

Historically, mires were distinguished into two groups according to their situation, as well as their water and nutrient source. The first group, minerotrophic peatlands (or fens), are situated in depressions and fed by surface- or groundwater as well as rainfall (Parish et al, 2008). They have slightly acidic to alkaline soils with a pH between 5.8 and 8.4 (Sjörs 1950, Limpens et al. 2008). However, depending on the hydrological connectivity with the mineral soil, fens tend to be eutrophic or oligotrophic (rich or poor in nutrients). In oligotrophic fens, *Sphagnum* moss dominates, while eutrophic fens are normally vegetated by sedges, reeds and elder-swamp forests (Parish et al 2008). The second group, ombrotrophic peatlands (or bogs), are characterized as nutrient and mineral poor *Sphagnum* dominated bogs (Payette and Rochefort, 2001) and have their largest surface distribution in the temperate climatic zone. Ombrotrophic bogs are raised with respect to the surrounding surface and are exclusively rain-fed, hence they are consequently classified as oligotrophic (nutrient poor) and acidic peatlands with pH values ranging from 3.3 to 4.5 (Sjörs 1950, Clymo 1964, Parish 2008). Living Sphagnum mosses have no roots and grow over the thick accumulation layer of preserved plant detritus from previous years (Charman, 2002). Bogs have a low nitrogen concentration in the form of nitrates and a high C:N ratio resulting in a low rate of decomposition.

The water chemistry of a peatland is influenced mostly by the quality of water entering ecosystem via precipitation, surface runoff or groundwater flow, as well as the chemical and biochemical processes happening in the peat. These transformations can critically affect the availability of elements for plants and microorganisms, and thereby the oxidation-reduction

potential (redox) is a key factor in determining the electron availability, and consequently, bacterial activity (Ross, 1995). The reduced conditions not only reduce organic matter decomposition rates but, also favor the production of methane (CH_4) over that of carbon dioxide (CO_2). As mentioned before, undisturbed mires are generally sinks of CO_2 and N_2O and sources of CH_4, however, long-term studies show that GHG emissions are spatially and temporally variable, and that a given peatland may act as net CO_2 sink or source, depending on climatic conditions.

2.2 Greenhouse gas exchange in peatlands

2.2.1 Carbon dioxide (CO_2)

Carbon dioxide is considered to be the most important anthropogenic greenhouse gas (GHG) with a radiative forcing factor of 1.83 W m^{-2} (Etminan et al., 2016). CO_2 is inert in the atmosphere, however, it plays a key role in the carbon cycle of all ecosystems as well as soil-biosphere-atmosphere coupling. The global mean CO_2 concentration in the atmosphere was estimated at approximately 399 ppm in 2015, which is an increase of 20.66 ppm since 2005 (IPCC, 2007; Hartmann et al., 2013). Figure 2 gives an overview of atmospheric GHG concentrations determined from ice core data and from direct atmospheric measurements.

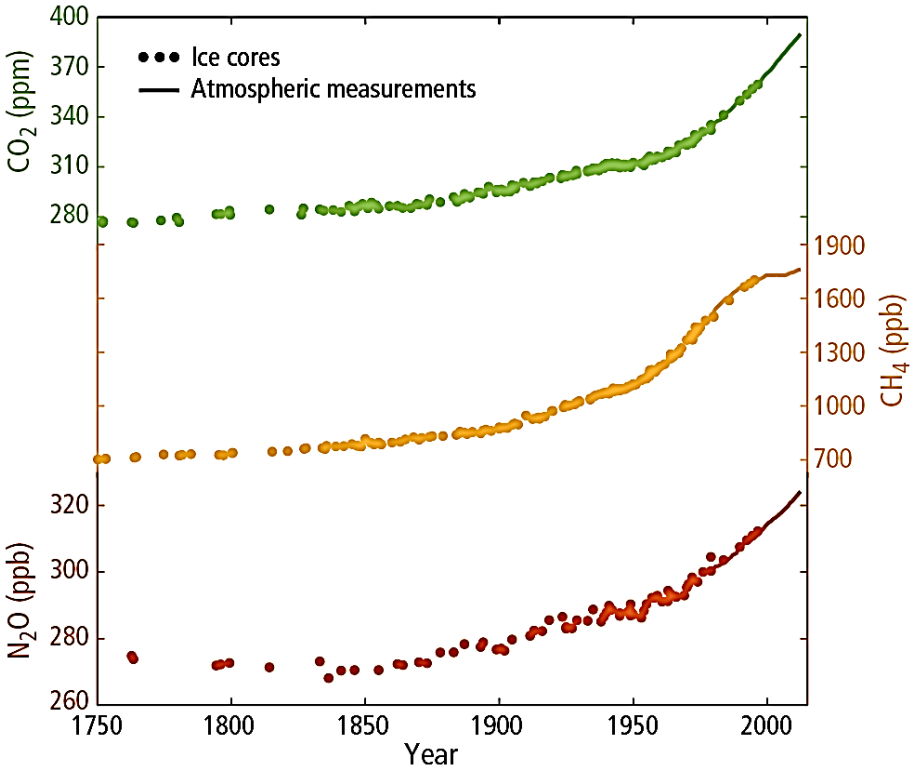


Figure 2. Atmospheric concentrations of CO_2, CH_4 and N_2O over the last 260 years (IPCC, 2014).

In terrestrial ecosystems, the processes of photosynthesis, respiration and decomposition are the main regulation pathways for atmospheric CO_2 concentration. When levels of carbon

dioxide uptake (as an input into the plant ecosystem) are higher than those of plant and soil respiration (ecosystem output), this results in a positive value of net ecosystem exchange (NEE) as displayed below in equation 1. The main input of carbon to the ecosystem is the assimilation of atmospheric carbon dioxide in living plant biomass via photosynthesis through gross primary production (GPP). Photosynthesis is mainly dependent on climatic (temperature, moisture availability) and soil conditions (Segers 1998; Lafleur et al. 2005), as well as vegetation type which is affected by the nutrient and hydrological status (Komulainen et al. 1999, Weltzin et al. 2003). Carbon dioxide is also produced and exported back to the atmosphere through ecosystem respiration (R_{eco}). R_{eco} is made up of above- and belowground autotrophic respiration by plants and roots (R_a), as well as heterotrophic respiration (R_h) by soil aerobic and anaerobic microorganisms which consume oxygen (O_2) and glucose (Schütz et al. 1991). Root respiration is the main process contributing to soil respiration, contributing between 10 % and 95 % depending on plant type and temperature (Hanson et al., 2000). The net ecosystem exchange and ecosystem respiration are defined as follows:

$$NEE = GPP - R_{eco} \tag{1}$$

$$R_{eco} = R_{vascular\ plants} + R_{sphagnum} + R_{soil} \tag{2}$$

The main controls on R_a and R_h are water availability, as well as water level and temperature respectively. Increasing temperatures have a strong positive effect on the rate of soil decomposition, and an even stronger impact on the photosynthesis and respiration. The aerobic decay and oxidation of methane transported from the catotelm are both alternative sources of CO_2. Additionally, carbon dioxide is also consumed by methanogens, transformed to methane or dissolved organic carbon (DOC) and released to the atmosphere or groundwater respectively.

The rate of CO_2 production and its emissions are dependent on a wide range of in-situ conditions such as temperature, soil type, vegetation cover, peat composition, nutrient- and pH status. Hooijer et al (2010) describe the relationship between CO_2 emissions from peat soils and groundwater depth as having a linear relation, calculating an increase of 9.1 t CO_2 ha^{-1} y^{-1} in CO_2 emissions for a 10 cm lowering of the water table.

2.2.2 Methane (CH_4)

Methane has a relatively low atmospheric concentration of 1,834 ppb (Etminan et al., 2016), however despite this fact it remains an important greenhouse gas with a radiative forcing factor of 0.61 W m^{-2}, and a global warming potential (GWP) 34 times higher than that of CO_2 for a 100 year period (Myhre et al., 2013). Historical atmospheric concentrations of methane have increased due to intensive human activity, and today more than 70% of all CH_4 emissions have anthropogenic sources (e.g. agriculture, rice paddies, livestock, biomass burning), with the remaining 30% coming from geological sources, oceans, forests, fires and peatlands (Denman et al. 2007). In peat soils, CH_4 is produced in the catotelm by obligate anaerobic bacteria (also called methanogenic bacteria) during anaerobic conditions often after prolonged water logging

by redox potential (Eh)<-300 mV (Smith et al., 2004). Acetotrophic methanogens are able to produce up to 80% of biogenic methane in the soil from acetate derived from soil organic matter (SOM), root exudates or DOC (Charman et al, 1994), whereas hydrogenotrophic bacteria can use hydrogen molecules to reduce carbon dioxide to methane (Garcia et al 2000, Lai 2009), a process which represents 10-30% of total biogenic CH_4 production (Norina, 2007):

$$CH_3COO^-H^+ => CH_4 + CO_2 \quad (3)$$

$$4HCOO^- + 4H^+ => CH_4 + 3CO_2 + 2H_2O \quad (4)$$

$$CO_2 + 4H_2 => CH_4 + 2H_2O \quad (5)$$

Once methane is produced, it can be transported from the soil to the atmosphere via the three mechanisms shown in Figure 3. Firstly via a slow diffusion process along the concentration gradient following Fick's first law of diffusion, secondly via plant-mediated mass flow through the aerenchyma tissue of vascular plants (Schäfer et al., 2012), and thirdly via ebullition in the form of gas bubbles (Kutzbach et al, 2004):

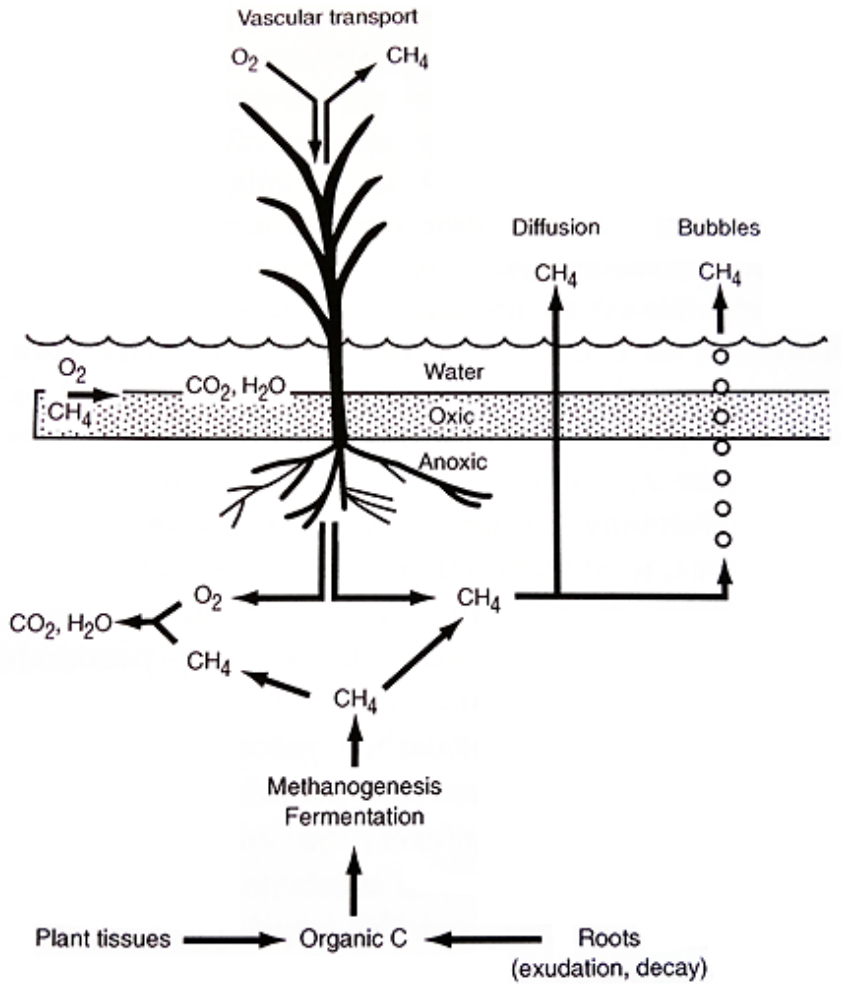


Figure 3. Schematic illustrating the methane production, transformation and transport mechanisms in mires (Charman, 2002).

In contrast to other two processes, the diffusional transport is generally the slowest transport mechanism, especially in water where before reaching the atmosphere 60- 90% of methane net flux can be reoxidized in the acrotelm immediately above the water table by methanotrophic bacteria (Hornibrook et al, 2009). Using the enzyme methane monooxygenase (MMO), methanotrophs utilize CH_4 as their main source of carbon and energy, and stepwise oxidise through methanol to formiate and then, finally to CO_2 (Whalen 2005; Dutaur and Verchot, 2007):

$$CH_4 + 2O_2 => CO_2 + 2H_2O \quad (6)$$

The other two mechanisms allow methane to move from the catotelm directly to the atmosphere. In this way, ebullition can release 50-60% of produced CH_4 to the atmosphere, and plant-mediated transport is responsible for 30-100% of passive and active methane transport through plants. The annual release of methane by undisturbed peat soils is about 2-5 g C m^{-2} y^{-1}, with increased rates of up to 205 g C m^{-2} y^{-1} measured in flooded rewetted peatlands (Drösler et al., 2008). The controls on CH_4 production and its release from the soil are very similar to those for CO_2, with the further impact of water table level.

2.2.3 Nitrous oxide (N_2O)

Nitrous oxide emissions have received little attention during the last decades, despite this gas being the third most important GHG with a radiative forcing factor of 0.17 W m^{-2}, and a GWP of 298 over a 100 year time scale (Myhre et al, 2013; Hartmann et al., 2013; Etminan et al., 2016). Current atmospheric concentrations of nitrous oxide are around 328 ppb, and this gas plays a central role in the soil nitrogen cycle. N_2O is produced in soils by two main processes (Fig.4): (1) as a by-product of autotrophic aerobic nitrification and (2) as an intermediate product of heterotrophic anaerobic denitrification (Davidson, 1991; Hayatsu et al, 2008), although in general the denitrification process can act as both a source and a sink of nitrous oxide.

The nitrification process, in other words the strictly aerobic oxidation of NH_4^+ to NO_3^-, has two steps: (1) nitritation, mediated by ammonia-oxidizing bacteria *Nitrosococcus sp*., *Nitrosomonas sp*. and *Nitrosospira sp*.; and (2) final nitratation mediated by nitrite-oxidizing bacteria *Nitrobacter sp*., *Nitrococcus sp*. and *Nitrospira sp*. (Moreira and Siqueira, 2006; Signor and Cerri, 2013). Under anaerobic conditions, during the denitrification process where the Water-Filled Pore Space (WFPS) is >50%, facultative anaerobic bacteria can produce N_2 from NO_3^- (Ussiri and Lal, 2013), while high amounts of nitrogen can be emitted into the atmosphere in form of N_2O (Oertel et al., 2016). According to Bremner (1997), small amounts of nitrous oxide can also be produced due to the chemical processes of nitrite decomposition, called chemidenitrification, and hydroxylamine (NH_2OH) oxidation, both occurring in neutral and acidic soils without the contribution of microorganisms.

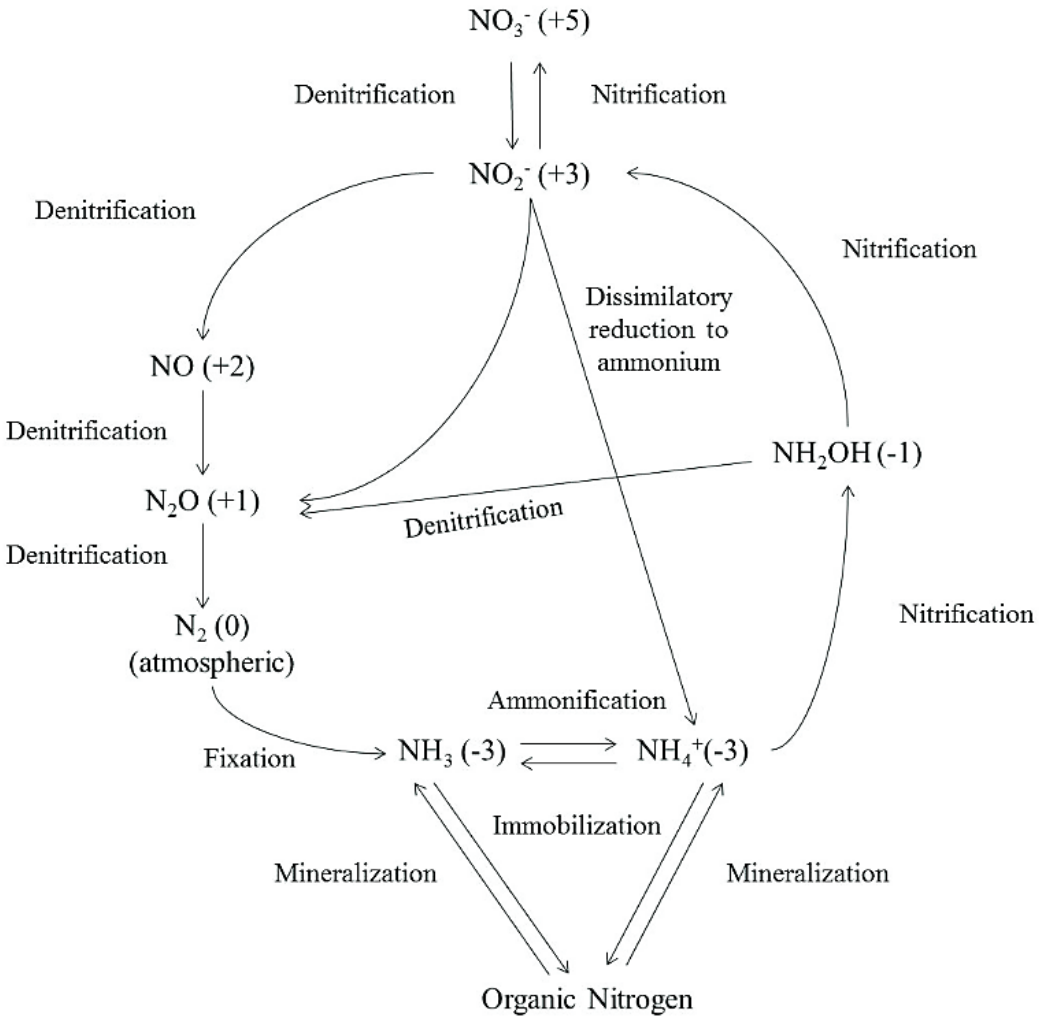


Figure 4. Biochemical processes in the soil nitrogen cycle (Signor and Cerri, 2013).

Pristine peatlands with a natural water table level are known to have low nitrous oxide emissions and low consumption rates of this gas by soil microorganisms due to O_2 poor conditions, and hence low rates of nitrification resulting in low nitrate (NO_3^-) content of the peat (Regina et al. 1996; Aerts 1997; Brumme et al. 1999). However, some mires can episodically emit high amounts of N_2O due to their high spatial landscape variability and dry climate conditions (Schiller and Hastie, 1994). In addition to nutrient status, high soil acidity also may restrict the denitrification processes in water and peat resulting in decreased N_2 formation. According to a study by Aerts (1997), in peat soils with low pH values the N_2O reductase may be inhibited, and hence the ratio between produced N_2O and N_2 is increased.

Some authors have identified a strong relationship between N_2O and CO_2 emissions depending on SOM decomposition and hence on the soil C/N ratio (Garcia-Montiel et al, 2002; Chatskikh and Olesen, 2007). Some possible N_2O gas uptake by peat soils is reported to be caused in part by denitrification, when nitrous oxide is converted to N_2 (Chapuis-Lardy et al., 2007). Observed annual nitrous oxide fluxes range from -0.1 kg N_2O-N ha^{-1} for undisturbed mires with a high water table level (Tauchnitz et al, 2008) to as much as 72 kg N_2O-N ha^{-1} for a drained forested

peatland in Germany (Brumme et al. 1999), however reports of N_2O emissions from rewetted degraded bogs and fens are still lacking.

2.3 Peatland drainage and climate change

Human activities such as peatland drainage, tillage operations and extraction create well aerated conditions, which induce deep peat oxidization and compaction (Figure 5) and hence, destroy all functions of natural peatlands (Landry and Rochefort, 2012). As a result of these processes, natural peatland morphology and original soil-physical parameters can be lost (Gebhardt et al., 2010).

Nieuwenhuis and Schokking (1997) described three components of peatland subsidence caused by drainage and peatland use: (1) primary consolidation as a result of a decrease in pore water pressure, (2) secondary shrinkage which leads to a peat volume reduction above the water table level, and (3) active peat mineralization (humification) under aerobic conditions. The first two stages can persist for a few years after draining due to the rapid compression of saturated peat layers below the WTL, whereas the last process may be lengthy and continue for several decades. In addition to drainage, the covering of areas by sand and loam, the creation of water filled ditches, and the use of heavy equipment for peat extraction can all cause further peat subsidence.

During the post-drainage phase, soil organic matter starts to decompose, soil carbon is released to the atmosphere as gaseous CO_2, and nitrogen is lost in run-off as nitrate (xNO_3) or is emitted as N_2O (Koppisch 2001). The largest emissions are expected on more nutrient-rich soils, although these emissions not significant in the context of global N_2O exchange (Martikainen et al, 1993). As a general rule, the emission of CO_2, CH_4 and N_2O, which account for 2-3 Gt of global CO_2-equivalents per year (Joosten and Couwenberg, 2009), are increased in drained compared to undisturbed peatlands (Couwenberg, 2011).

In Germany, peatlands cover ca. 15000 km^2, which is approximately 4.3% of the land area (Montanarella et al 2006), and are estimated to contain 422 Tg C (Joosten and Clark, 2002; Byrne et.al, 2004). Peat extraction for fuel and the demand for agricultural land caused large-scale peatland drainage here in the 19^{th} century. Of the natural peatlands which existed previously, more than 72% have been destroyed or moderately impacted by drainage for agriculture, which is the most important use of peatlands. According to Joosten and Clarke (2002), about 20-25% of all peatlands globally have already been altered and are currently used for agriculture and forestry. These affected peatlands release approximately 3 billion tonnes of carbon dioxide annually into the atmosphere, which represents 5-6% of the total anthropogenic GHG emissions globally.

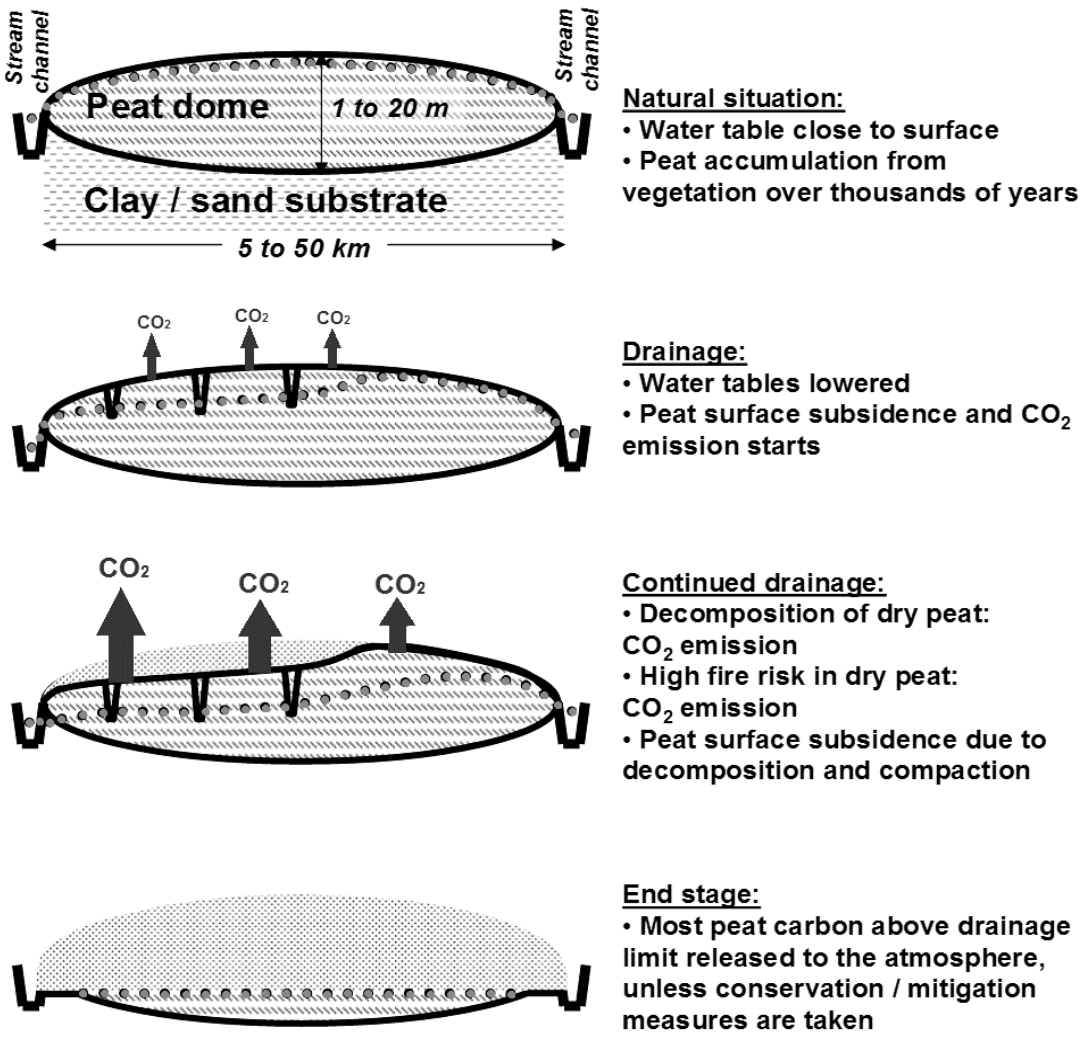


Figure 5. Schematic illustrating the progressive subsidence of the peat surface in drained peatlands due to drainage resulting in CO_2 emission and peat compaction (modified after Hooijer et al, 2010).

2.4 Peatland rewetting

A number of developed countries have begun the process of renaturalizing disturbed peatlands, with the goal to re-establish the natural wet peatland conditions as well as to reduce peatland GHG emissions. The rewetting projects in Germany have been the largest carried out in Europe during the last 40 years (Zerbe and Wiegleb, 2009). In practice, only 7 % of recent projects in northwestern Germany followed through with monitoring before and after restoration (MUNLV 2005).

Peatland restoration commonly takes place in two stages. The first is the re-establishment of a high water table level (rewetting) and the second is the recolonization of important peat-forming Sphagnum species (Holden et al, 2004; Landry and Rochefort, 2012). According to the drainage duration and the restoration objective, the water level can be raised and stabilized by different methods such as the blocking of drainage ditches, for example by covering them with wet

humified peat or sawdust; regulation devices; and by building dams from peat, wood or plastic around the rewetting site. These different techniques achieve rewetting by reducing peatland water losses, as well as decreasing the amount of surface runoff, groundwater flow, and evapotranspiration rates (Podschlud, 1988; Gunn and Walker, 2000; Holden et.al., 2004; Landry and Rochefort, 2012).

The rewetting of degraded peatlands can have a range of positive impacts, such as the reduction of fire risk and a decrease in CO_2 and N_2O emissions, but also less desirable consequences such as an increase in site CH_4 emissions (Glenn et al. 1993; Tuittila et al. 1999). Furthermore, natural yearly variations in water level can create oscillatory conditions (aerobic in summer and anaerobic in winter) and support the microbial mineralization of peat and thereby, CO_2 and CH_4 production (Strack, 2008). After the peat oxidation is stopped, peat may be accumulated again and the wet peatland can cool the regional climate through increased evapotranspiration (MLUV MV, 2009). Despite the fact that several previous studies have investigated rewetted peatlands (Beyer and Höper, 2015), there are a number of important aspects concerning rewetting that are poorly understood. As such, an assessment of the GHG emissions from a wide range of degraded and restored sites is still needed (Artz et al., 2013).

3 Materials and methods

3.1 Study site

3.1.1 Climatic and geological description

The Himmelmoor peatland, located in Schleswig-Holstein (53°44′20″N, 9°50′58″O), is one of the largest raised (rain-fed) heavily drained bogs in northern Germany with an area of 6 km^2. The region has a temperate hemiboreal climate, classified as Cfb (Kottek et al., 2006), with annual mean precipitation of 838 mm, and an average temperature of + 9 °C for the reference period 1981- 2010 (ID 4039, Deutscher Wetterdienst).

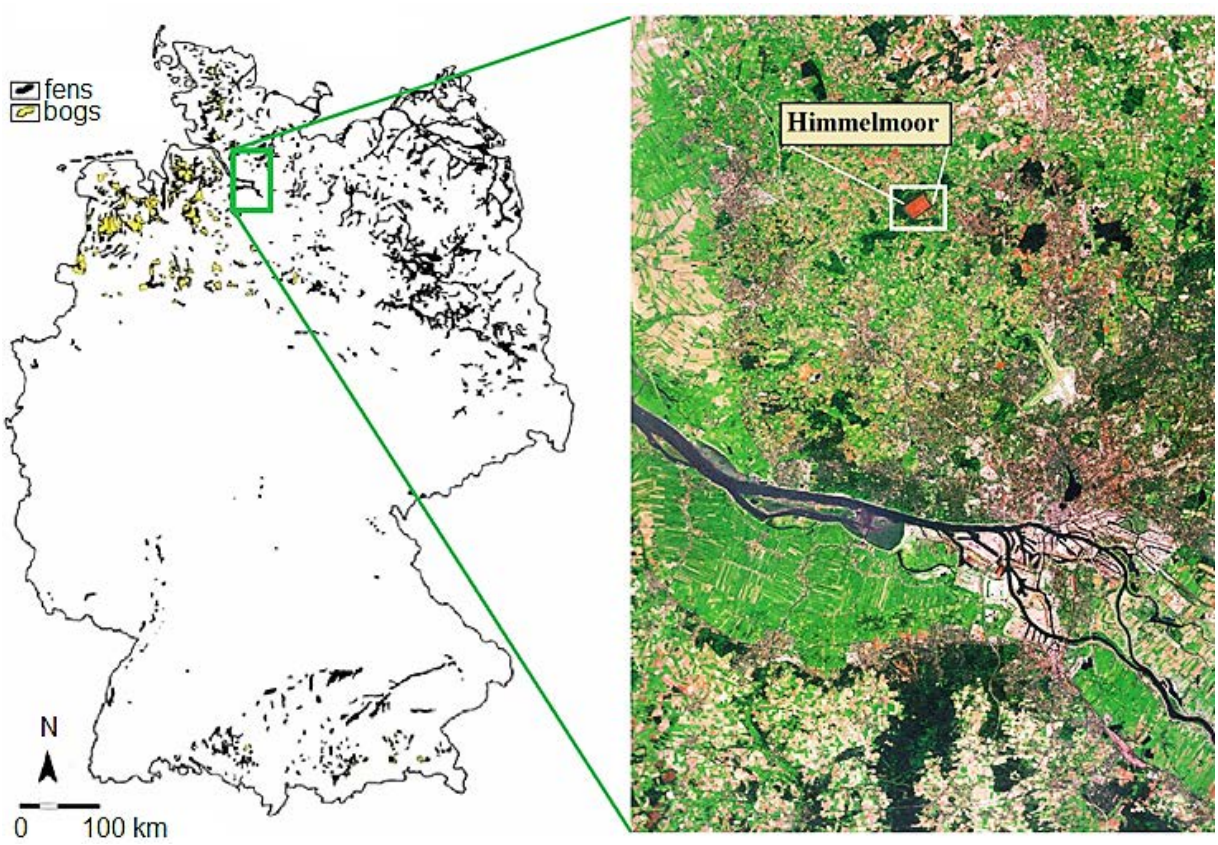


Figure 6. Location of the Himmelmoor, 25 km northwest of Hamburg (modified from Schopp-Guth, 1999 and Satellitenatlas, 2005).

Himmelmoor is bordered by the stream Bilsbek to the northwest and the river Pinnau to the southeast. Together they have as a biotope complex the certain significance for the regional hydrology, both collecting the water from the peatland and connecting to the river Elbe. This area was glaciated during the “Wolstonian Stage” in the mid-Pleistocene, and as the glaciers retreated they left behind moraine sediment sand the deposition of meltwater sands began to form the basis of the Himmelmoor (Meyer, 1983). The accumulation of peat is expected to have begun after the last ice age, around 10,020 ± 100 years before present (Pfeiffer and Becker-

Heidmann, 1996). This marked the start of the terrestrialisation process as peat clay began to form and a fen was formed. Afterwards, this formation shifted slowly to paludification and sedge peat accumulation began with *Phragmites* and *Carex* species, followed by reed and birch carr peat. During the "Atlantic period" (8,000-5,000 yr. BC), conditions were warmer, with average temperatures of 2-3 °C higher than today. In response to increased glacial melt, the sea level of the Baltic Sea rose from -22 to -5 m causing a shift in vegetation cover (Behre, 1988), and consequently the Himmelmoor became a raised bog dominated by *Sphagnum* mosses (Grube, et al., 2010). The peat covered area at the site accounts for approximately 605 ha and the total thickness of the peat accumulation was up to 10 m deep (Pfeiffer 1998, Vanselow-Algan 2014). In contrast, at present, the maximum depth which can be reached in some places is approximately 2-3 m. The vegetation cover of the rewetted areas in the central part of the peatland consists mostly of different types of Sphagnums mosses (*S. palustre, S. angustifolium, S. fimbriatum, S. rubellum, S. imbricatum*), birch and bog willow trees (*Salix, Betula pubescens*) as well as sedges (*Eriophorum, Molinia caerulea*). The areas under active peat extraction have no vegetation and have only bare peat strips.

3.1.2 Anthropogenic activity and current projects

The extraction of peat for fuel in Himmelmoor, started in the 1780s and continued until 1968. Since 1918, peat has been excavated for horticulture by Torfwerk Quickborn (Torfwerk Enfeld Carl Hornung Werk Quickborn), and at present around 10 cm of soil (in several 2 cm steps corresponding to the drained depth; K.-D. Czerwonka, personal communication, 2016) is excavated yearly, with an estimated total of 30,000 – 38,000 m³ peat extraction per year. The drainage of the central part of the bog began in the 1930s. At this point, a maximum peat extraction depth was established, limiting the cutting depth to 1-2 m above the Quaternary consolidated sediments. In some places drainage led to the soil compaction of up to 40% of the initial peat depth (Grube et al., 2010). Since the mid of 19th century, drainage practices have affected a total area of 142 ha (Zeltner, 2003), and in 1993, the decision was made to gradually restore the peatland to its natural state by rewetting the area (Fuest and Menzel, 2008).

As peatland rewetting and peatland flooding are two different restoration techniques with different consequences for the restoration effect, it should be mentioned that the "rewetting" method implemented in the Himmelmoor contained very special flooding events. This special "rewetting" process began in 2004 and was done both by blocking the drainage ditches with peat and by building peat dams. All excavation activities are to be ended in 2017, and already more than 60% of the former extracted area has been flooded since the start of the project (Czerwonka and Czerwonka 1985; Vanselow-Algan et al. 2015). The peatland is now an experimental study site managed by the Institute of Soil Science at the University of Hamburg, and continuous measurements of GHG fluxes have been made since 2011. Figure 7 shows an overview of the bog and the different stages of the rewetting process.

3.2 Sampling sites

The area examined in this research project consists of five study sites with different microtopes and land-use history, vegetation cover and water table level. It was assumed that sites with a different WTL and land-use have differing GHG budgets. At each site, one or two subsites (each with three to four replicates) were selected (Figure 7; Figure 8), which allow all analysis to be statistically estimated. Within two vegetated sites RV and D, both rewetted, areas with *Eriophorum/Sphagnum* species as the dominating type of the whole microtope were chosen.

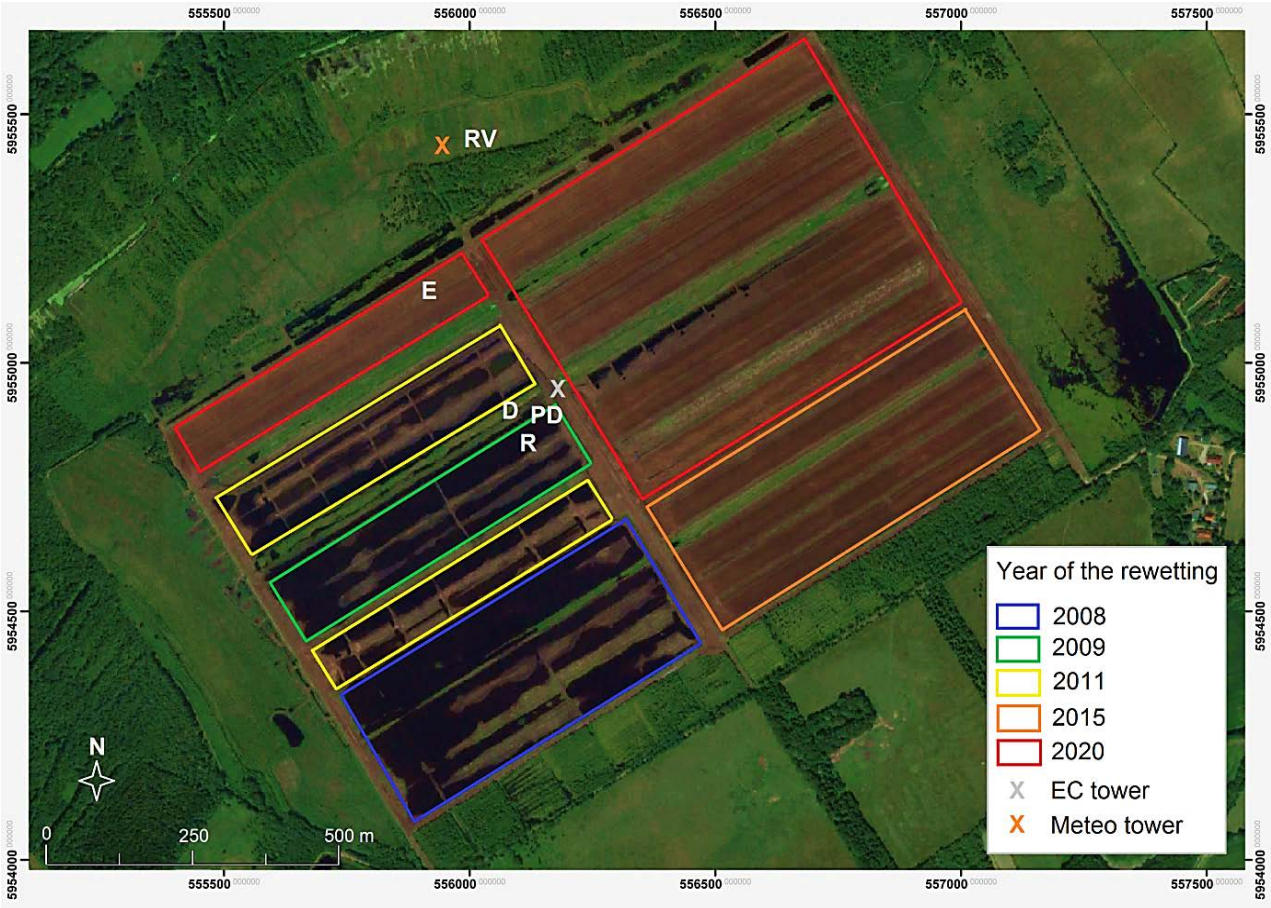


Figure 7. Aerial photograph of the Himmelmoor showing the rewetting status and location of the study sites. The different colors indicate different stages of the rewetting process, and. abbreviations in white show mark the location of the study sites: RV – an area with vegetation, rewetted in 2004; E – extraction area; D – ditch; PD – peat dam; R – rewetted in 2009 (modified from Google Earth, 2013).

Table 1. Description of study sites in the Himmelmoor.

Study site	Abbreviation	Subsite	Area (m²)	Dominating vegetation
Rewetted in 2004	RV	RV	84, 550	*Sphagnum sp., Eriophorum angustifolium*
Extraction site	E	E1, E2	307, 800	Bare peat
Ditch refilled with peat	D	D0, DV	221, 100	*Eriophorum angustifolium*
Peat dam	PD	PD	144, 900	Bare peat
Rewetted in 2009	R	R	81, 300	No vegetation

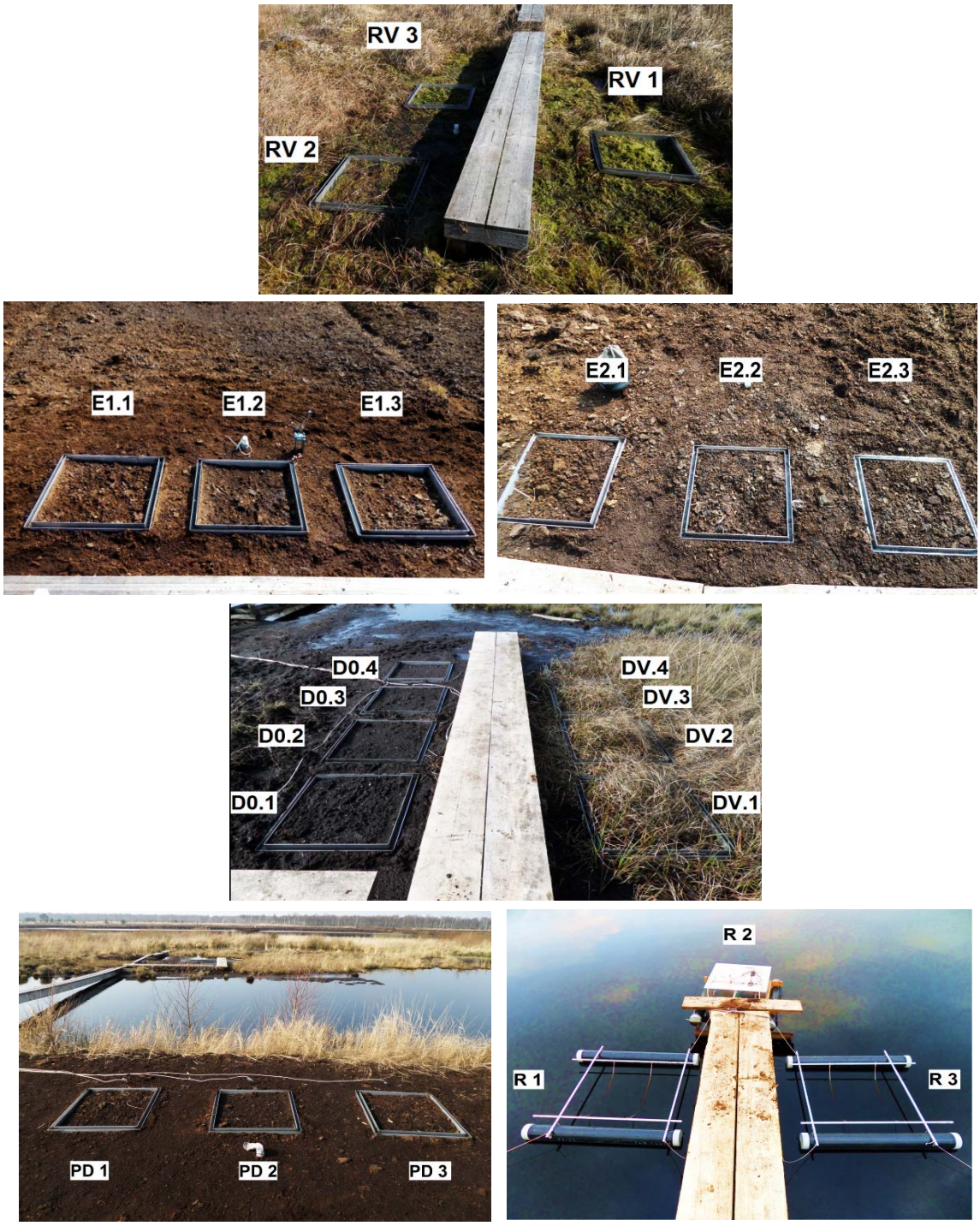


Figure 8. Overview of study sites with 3 to 4 collars/replicates at each subsite (O.Vybornova, 2014).

The **RV** site (Figure 8, upper picture), called the “Knustwiese” in German, is a former manual extraction site, which was active until the 1960s. This site was refilled with peat in the 1980s, and rewetted in 2004. Today, after 14 years of the rewetting, this area is overgrown by typical wetland vegetation dominated by *Sphagnum* mosses and *Eriophorum* grass.

In contrast, at site E, which has the largest area of 31 ha, extraction activity is still ongoing, and the upper soil layer is drained and regularly removed (approximately 2 cm five times per year, max. 10 cm annually). In January 2014 the area in the northwestern part of the Himmelmoor (Figure 7) was chosen as the test field of the E site; two representative subsites **E1** and **E2** (10 m apart) were chosen on two different strips divided by secondary drainage grooves because of possible spatial variability of the peat soil.

The measurement site **D**, called “Pütten” in German, is located in the southwest of former peat mining area. Until the 1970s it was an anthropogenic rearranged deep ditch under overburden black peat extraction from the lower layers, which was later refilled with mixed peat and flooded between 2009 and 2011. Since degradation was stopped in the area, it has been partially vegetated by *Eriophorum*, *Molinia* and *Calix* grass and can be clearly distinguished from the other study sites by its good vegetation cover and a WTL close to/above the soil surface depending on seasonal conditions. The site was divided into two subsites, one non-vegetated (**D0),** and one covered by *Eriophorum angustifolium* (**DV),** each with 4 replicas along the WTL gradient.

Comparing to the previous plots, the peat which formed the dry wall of the **PD** area (Figure 8, lower left picture) was strewed and compressed (50 cm high and 4 m wide) around the borders of the rewetted area (R) in 2009, and the drainage ditches were covered with peat material to reduce lateral water losses from the study site. This resulted in several bare dry peat dams with an area of approximately 2.1 ha (total area:14.5 ha), with a height difference between PD and R of 50 cm.

The last one, the study site **R** (Figure 8, lower right picture), or "Wiedervernässungsfeld", is located in the north-western part of the former extraction area near the PD site, and was flooded with rainwater in 2009. Under normal conditions, almost all of this area is water saturated, however during long dry periods which usually occur in August, some parts of the site may become dry at the surface.

3.3 Vegetation analysis

The vegetation coverage and abundance of the two re-vegetated study sites (RV and D) were analyzed twice inside each subsite during this research: once in February 2014, and then again in October 2016. The plant species were described using the vegetation classifications of Schubert et.al (2000), Laine et.al (2009) and Roger (1990) and collected in the herbarium. The vegetation coverage inside each collar was additionally determined in percentage terms as a number of pixels (px) from the colored pictures of chamber plots taken three times during this study. The high resolution figures were loaded into Paint 6.1 and GIMP 2.8, where the area of

framework collars was removed from the picture, leaving only the areas with different plant types marked. These selected objects were then used to calculate the total area covered by plants, which was subtracted from the total number of pixels, and finally calculated in % using proportion.

3.4 Instrumentation and environmental measurements

Two meteorological stations were installed at the site in 2011 in order to continuously monitor different environmental variables. These were located in the center and in the northwest of the peatland at 2 m height. The stations measured the following variables: air temperature and air pressure (Tair; model HMP45, Campbell Scientific Inc., USA), wind speed, wind direction and precipitation (Ws; model 05103-5, R. M. Young Company, USA).

In addition to the above meteorological data, measurements of different soil variables have been conducted since 2014. Soil temperature and soil redox potential (Ts, Eh) were measured at each collar with soil sensors located at four different depths (2, 5, 10, 20 cm) using Hypnos III (MVH Consult, Netherlands). The redox potential (Eh) was calculated by adding the potential from the reference electrode (E_{ref}) to the measured potential (E_m):

$$Eh = E_m + E_{ref} \quad (7)$$

A perforated PVC pipe (5 cm wide, 2 m long) was inserted vertically into the peat at each site to measure the groundwater table level and water temperature inside of pipe at 30 min intervals using Mini-Diver sensors (Schlumberger Water Services, Netherlands). WTL was calculated using the following equation:

$$WTL = (L_c - L_p - (P_{water} - P_{air}))* (-1) \quad (8)$$

where WTL is the water table level (cm), L_c is the length of the diver cord (cm), L_p is a supernatant of the pipe above the soil surface (cm), P_{water} is the water pressure and P_{air} is the air pressure (cm of water column).

3.5 Gas flux measurements

3.5.1 Gas sampling

During this research, gas flux measurements for the monitoring campaign were conducted at weekly or biweekly intervals using the manual static closed chamber technique. During chamber setting, the headspace of the chamber is completely isolated from the atmosphere, and the gas flux is calculated from the change in measured concentrations over time (Kutzbach et al, 2007). In contrast to the eddy-covariance method, which is the most direct and high-temporal technique of CO_2 and CH_4 measurements over an entire ecosystem, the application of manual

chambers requires extensive fieldwork and can disturb the ecosystem but, despite these drawbacks, it provides more representative information on GHG fluxes at the small scale (Drösler et al., 2008), which was the main goal of this research.

In this study, manual opaque closed chambers were used to measure fluxes of CO_2 (Total ecosystem respiration), CH_4 and N_2O in three to four replicates at seven subsites. The chambers were not connected to a gas analyzer and were used only for collecting gas samples, which were later analyzed using a gas chromatograph (GC). In order to minimize the compression of the underlying peat core and the ebullition of gas during flux measurements, a board walk was constructed at each study site one month before the measurements began. At site R three swimming chambers (60 x 60 x 33 cm, height varied according to the water level or wave height) were constructed near the wooden board walk.

A total of 20 stainless steel U-shaped collars (60 cm x 60 cm x 30 cm) with a groove were constructed with holes to allow for lateral water flow and minimize disturbance. These were permanently inserted into the peat at the RV, E, D and PD sites in January 2014, with approximately 50 cm distance between the collars, which extended no more than 5 cm above the soil surface.

The gas chambers were constructed of thin-walled aluminum to minimize disturbance to the headspace volume during setting it on a collar and had a volume of 0.11 m^3. The air inside the chamber headspace was continuously mixed using a battery-operated fan to provide well-mixed gas samples, and was thermostatically controlled (the chamber air temperature was monitored using an electronic thermometer) as shown in Figure 9. Additionally, to reduce the risk of pressure perturbations, and hence disturbance of the natural gas flux (diffusion or plant-mediated transport) from the soil to the atmosphere, a vent tube sized according to the recommended guidelines (Baker et al., 2003; Hutchinson and Mosier, 1981) and sampling port was installed.

Before the measurements began, the U-profile on the top of the collar was filled with water to provide an airtight seal. Two 4 cm wide vents at the front side of each chamber were opened during chamber setting, and closed directly after to avoid initial pressure shocks (Schneider et al., 2009; Vanselow-Algan, 2014). The non-swimming chambers were placed in the water filled on the top of the collar. After closing the two vents, six gas samples were taken from the chamber headspace (119 L) every 0, 6, 12, 18, 24 and 30 minutes using 50 mL plastic syringes equipped with a three-way stopcock, which allowed the sample to be stored and transported directly in the syringe. The deployment time of 30 minutes was chosen as a common closure time for measurements of methane and nitrous oxide (Alm et al., 2007; Parkin and Venterea, 2010), both of which do not vary as strongly as carbon dioxide fluxes over a short time period.

In most cases, all study sites were measured on the same days, and gas sampling was conducted between 10:00 and 12:00 a.m., as recommended by different authors in order to study possible diurnal flux variability and to provide a better approximation of the soil temperatures at the sampling time to the mean daily values (Parkin and Venterea, 2010; Vanselow-Algan, 2015). The gas fluxes were measured three to four times a month during the growing season and

biweekly during the winter season. In winter, as snow depth was less than 30 cm, the snow gas gradient method was not required (Alm et al., 2007) and GHG fluxes were measured in the headspace above the snow cover.

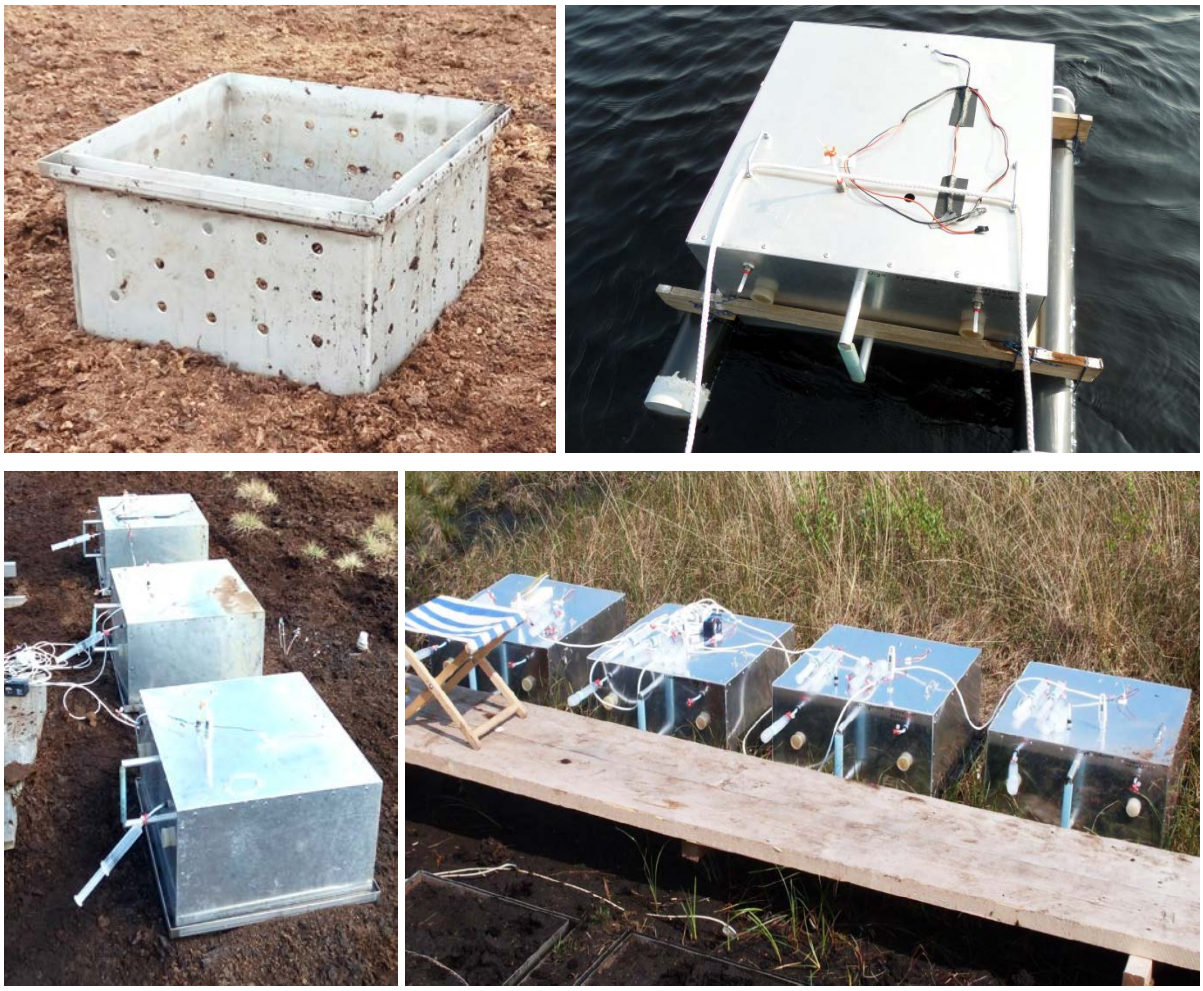


Figure 9. Pictures of the measurement set-up. Upper left: Steel collar before installation at the extraction site E. Upper right: one of three swimming chambers used in this study for gas sampling in the flooded area R. Lower left: Picture of gas measurements in area E with three metal chambers inserted on the grooves on the top of the collars. Lower right (Source: A.Schutt): Gas sampling in a vegetated part of Site D with four replicas.

3.5.2 Gas analysis

The gas samples were analyzed for CO_2 (total ecosystem respiration), N_2O and CH_4 as soon as possible (in most cases within two days after sampling) at the Institute of Soil Science (Hamburg University) using a GC Agilent 7890A (Agilent technologies, USA) equipped with a Flame Ionization Detector (FID) and an Electron Capture Detector (ECD). The samples were injected into the carrier gas stream via the septum module and transported to the analytical

column, where they were separated into three analyzed gas components depending on their retention times, and then measured by FID (CH_4, CO_2) and ECD (N_2O).

The GC was calibrated before every measurement in order to avoid the possible GC deviation, called "noise". Six standard gases (three mixed gases with CH_4 and CO_2: 1.79 ppm and 387.8 ppm, 209.7 ppm and 1005 ppm, 1004 ppm and 9710 ppm; and three N_2O gases: 0.29 ppm, 1.05 ppm and 1.55 ppm) were used for calibration by injecting them (i) three times before, (ii) periodically throughout and (iii) after the sample measurements from each measurement day. The output concentrations from the GC were averaged and compared with the known actual standard values and thus, a new calibration was conducted if required. A total of 9,738 gas samples were measured manually during this study.

Two additional measurement tests were conducted to estimate the losses of gas concentration from the sampling syringes. Hereby, five plastic syringes were filled with different standard gases (concentrations are described above) and measured daily for a one week period. The tests suggest that an average of between 2.13 to 9.1% of the initial gas concentration in the syringes was lost per day in storage. Information on this loss was accounted for in the calculation of gas concentrations, and it was decided that syringes stored for longer than three days would not be used.

3.5.3 Flux calculation

After calibration, the CO_2, CH_4 and N_2O fluxes (F_{gas}) were derived from the concentration increase in the chamber headspace during chamber closure time by calculating the individual measured values of the slope parameter (concentration gradient, ∂_c/∂_t) using the ideal gas law as follows:

$$F_{gas} = \frac{P_a \cdot V_{ch}}{R \cdot T \cdot A} \cdot \left(\frac{\partial_c}{\partial_t}\right) \quad (9)$$

where F_{gas} is the gas flux in mol m^{-2} s^{-1}, Pa is the air pressure during sampling (Pa), V_{ch} is the volume of the chamber headspace (m^3), R is the universal gas constant (8,314 Pa m^3 mol^{-1} K^{-1}), T is the headspace temperature at the beginning of sampling (K), A is the soil area covered (m^2), $\frac{\partial c}{\partial t}$ is the change in gas concentration over time (mol mol^{-1} s^{-1}). The chamber headspace was calculated as the sum of the chamber volume and the volume of each individual collar. The latter was calculated using the collar area and the mean distance between the soil surface, moss or snow cover and the water-filled frame on top of the collar.

GHG emission rates were determined by conducting a linear or non-linear regression on the time series using the MATLAB software (Matlab R2013a; Mathworks, USA) template of Forbrich et al. (2010). The linear model was used for estimating the mean increase in concentration over the closure period, whereas the slope of exponential regression was used to determine the rate of initial concentration increase at the start of the measurement.

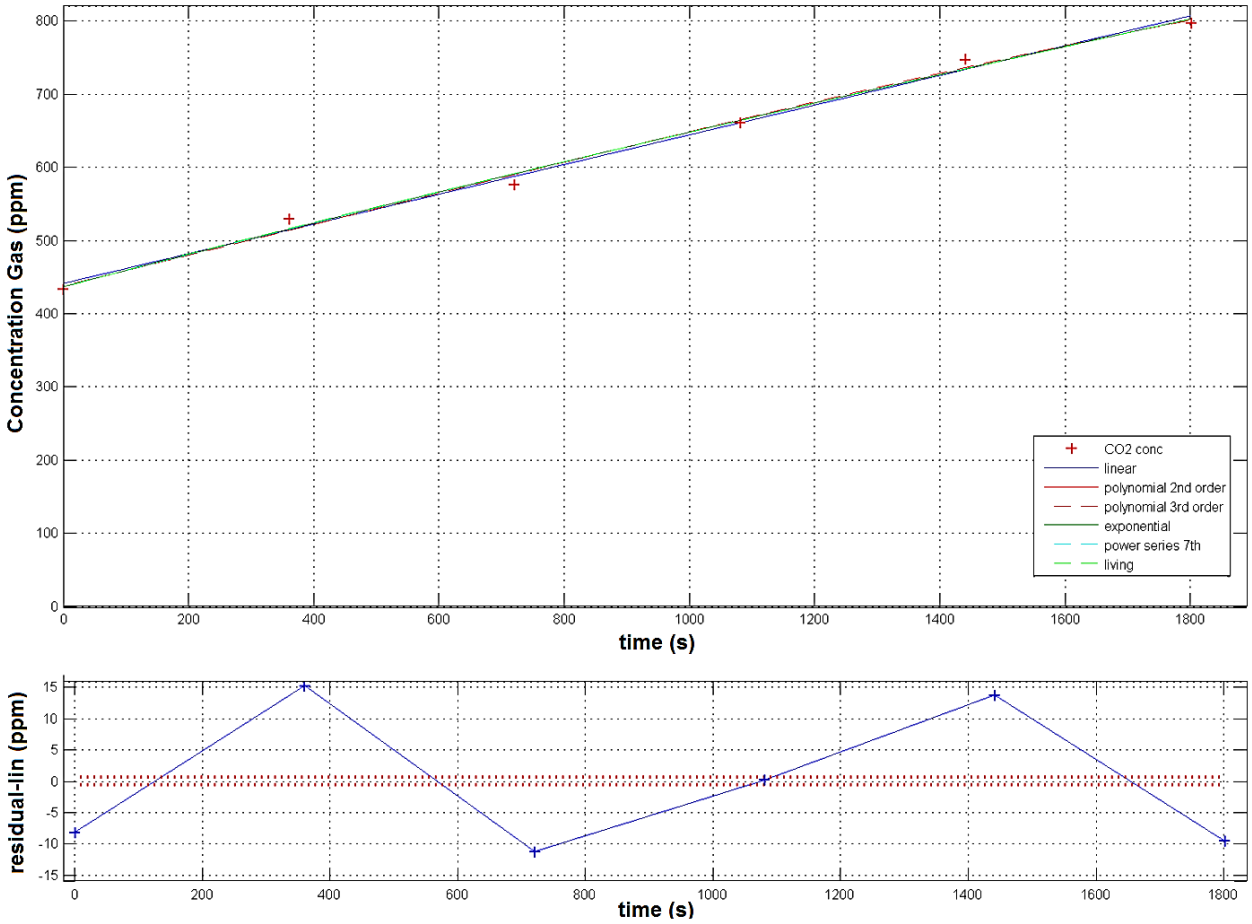


Figure 10. Sample evolution of CO_2 concentration over time. The sampling time is shown in seconds and time=0 describes the point of chamber setting. The six measured CO_2 concentrations in ppm are shown with red crosses on the upper panel. Several linear and non-linear regression functions were used for calculation of gas fluxes. The lower picture represents the linear residuals in ppm over time.

In order to find the best fitting relationship between the gas concentration change and the regression, all data sets were checked for errors and concentration anomalies, which were then removed from further calculations. Furthermore, the concentration detection limit was determined according to the width of the gas chromatograph's precision and only fluxes of chamber measurements with temperature changes of less than 5 °C were included in further analysis (Günther et al., 2014). From a total of 4,869 flux curves, more than 70% of data passed this criterion, and the most frequent reason for rejecting a measurement was the ebullition sign. As only six gas samples were analyzed in this research to estimate the flux curve, the *Akaike* Information Criterion AIC*c* with small sample second order bias correction (Akaike, 1974; Burnham and Anderson, 2004; Kutzbach et al, 2007) was used to select the linear/non-linear model which best represented the final slope for each measurement as described in Vanselow-Algan (2014).

3.5.4 Data processing and correction

The quality of each slope was checked by calculating its standard deviation, which were derived from the propagated standard deviations of the regression slope. Moreover, the obtained data sets were employed for the further calculation of statistical variables, and all calculations and processing steps were performed using SPSS and Microsoft Excel 2010 Software.

3.5.5 Calculation of annual GHG fluxes

Annual GHG fluxes were calculated from the mean fluxes measured at the three to four replicate plots. The annual CH_4 and N_2O fluxes were converted into CO_2-equivalents (CO_2 eq) according the Global Warming Potentials (GWP, over a 100 year time period) with included climate-carbon feedbacks of 34 and 298, respectively (Myhre et al., 2013) to estimate the total climatic impact of emissions from all studied areas.

Additionally, despite the widespread use and simplicity of GWP, some researchers (Smith and Wigley, 2000; Manne and Richels, 2001) have reported large differences in the indication of damage effect on air temperature caused by various GHGs and have hence criticized the GWP concept. Therefore, as an alternative to the GWP, the most well developed Global Temperature Change Potential (GTP, over a 100 year time period) measure was used for CH_4 and N_2O. GTP is based on the global mean surface temperature response to the emission of a specific gas relative to the emission of the same quantity of the reference gas CO_2, integrated over a specific time period (Shine et al., 2005; Wang et al., 2013). Thus, a 100-year-GTP is similar to a 100-year-GWP for long lived GHGs (297 for N_2O), however for short-lived climate forcing agents such as CH_4 the two measures show less agreement, with the GTP value for CH_4 (of 11 for biogenic CH_4) being 7 times lower than its GWP.

3.6 Soil surveying, sampling and storage

To study the pedogenic characteristics of the peat soils, two survey campaigns were conducted: five peat cores were drilled in May 2014 and additionally, three soil profiles as well as six peat cores were taken in May 2016. Three soil profiles were dug at non-rewetted areas (E, D, PD) approximately 1 m deep and were located close to sampling plots according to the vegetation and microtope of each study site. The depth of the peat cores was at least 2 m, and both the peat profiles and cores were described pedologically according to the World Reference Base for Soil Resources and US Soil Taxonomy (IUSS Working Group WRB, 2007; Soil Survey Staff, 2010). Soil samples were taken from each pedogenic horizon for all study sites (profile or core), transported in closed plastic bags to the Institute of Soil Science within 3 hours after collection, and stored cooled at + 4 °C. All living roots were removed from the mixed samples before analyses. Soil samples were crushed, sifted through a 2 mm sell sieve and mixed on the day of sampling. Samples from both sampling campaigns were then prepared for further analyses of

soil microbial biomass, C and N content, water content, NO_2/NO_3, NH_4, pH, bulk density, plant-available P and K. For carbon and nitrogen analyses, the soil samples were further dried at 105 °C for more than 12 hours.

3.7 Soil physical and chemical analyses

3.7.1 Total carbon and nitrogen content

The amount of total carbon and nitrogen in the samples was determined according to the guidelines of DIN ISO 10694. Hereby, 1.0 g of soil sample dried at 105 °C was measured automatically using a CN elemental analyzer (vario MAX cube CNS, Elemental analysis GmbH, Hanau, Germany) in stainless steel crucible. The tested sample was burnt in oxygen at temperatures of up to 900 °C and subsequently, the gases formed during this oxidation were separated and determined as CO_2 and N_2 in a thermal conductivity cell inside the instrument. Since peat soils showed low pH values and no reaction with HCl acid, it was assumed that the amount of total C is equal to the amount of organic C.

3.7.2 Soil pH and electrical conductivity

The soil pH value was determined using the electrometric method in an H_2O and $CaCl_2$ suspension: 25 g of air-dried sample were mixed with 75 ml of distilled water or 75 ml 0.01 M $CaCl_2$ solution, so that the ratio of soil to solution was 1:3. The resultant slurry was left for 1 hour whilst being regularly stirred. The pH of the supernatant suspension was measured using a glass single-rod calomel-electrode, connected to a pH-Meter (CG 820, Schott AG, Germany), and then the electrical conductivity of the aqueous suspension was determined using a conductivity sensor (Multi pH/Cond 350i, WTW GmbH, Weilheim, Germany). This was measured in two replicates for each horizon.

3.7.3 Contents of plant-available potassium and phosphorus

Plant-available potassium (K) and phosphorus (P) were measured according to VDLUFA guidelines (1991). Elements were extracted from a 2.5 g air-dried sample with a 125 ml double lactate (calcium silicate of the lactic acid) extract with pH 3.6 as described by Riehm (1942). This solution was shaken for 90 minutes and filtered with folded paper filters whereby 10 ml of the initial filtrate was not used. The potassium concentration was determined from this solution by Atomic Absorption Spectroscopy (AAS, type 1100B, Perkin-Elmer, USA), and plant-available phosphorus was measured photometric using a spectral photometer (DR 5000, Hach-Lange GmbH, Düsseldorf, Germany) after producing a molybdenum complex.

3.7.4 Water and organic matter content and water holding capacity

The soil water content (WC) was measured in 2 replicates of a 5 g fresh sample from each horizon by drying them for ≥24 hours by a temperature of 105 °C and massing them again to 0.0001 g scale. The water content was calculated as follows:

$$\mathrm{WC} = \frac{m_{water}}{m_{solid}} \cong \frac{m_{wet} - m_{dry}}{m_{dry}} \tag{10}$$

where WC is the water content in g, m_{wet} is the wet mass of the sample in g and m_{dry} is the mass of the dry soil sample in g.

After last weighing for WC, the samples were accurately weighed to 0.0001 g and placed in a porcelain crucible. To determine the loss on ignition (according to DIN 19684-3) and hence infer the organic matter content, the samples were annealed for >4 hours in a muffle furnace at 550 °C. These were subsequently cooled in a desiccator and then weighed again to the nearest 0.0001 g. The ignition process destroys the organic fraction and the organic matter (OM) content can then be calculated from the difference in mass before and after combustion.

The soil maximum water holding capacity (maximum WHC) was determined in 2 replicates of a 20 g fresh soil sample by saturating them with 100 ml of distilled runoff water. Samples were covered with aluminum foil to avoid water losses, then left for 12 hours (overnight), before being weighed and dried in a + 105 °C oven for at least 24 hours the next day. The water content of the samples was calculated as described before using the equation 11.

3.7.5 Bulk density

The bulk density was measured in fresh and undisturbed soil samples taken in the steel rings with a defined volume of 100 cm^3. Three replicates from each horizon were then weighed before and after being stored for 24 hours in the oven at 105 °C. The substance density was detected by a helium pycnometer (AccuPy II 1340 Gas Pycnometer, micromeritics), and the bulk density was calculated by dividing the dry soil mass (in g) by its volume (in cm^3):

$$\mathrm{BD} = \frac{m_{dry}}{V_{sample}} \tag{11}$$

Using the obtained results on C content (C) and bulk density, the carbon density (ρ_c in g/cm^3 or the C mass per unit volume) was calculated as follows:

$$\rho_c = \mathrm{BD} \times C_{org} \tag{12}$$

In addition, to estimatesoil organic carbon stocks (C_{stocks}) as well as total N stocks in tonnes of per ha for the surface and subsurface of all soil profiles, the value of soil carbon/nitrogen density

(ρ_c/ρ_n) was multiplied by peat depth increment (d) and the area of the plot (A) using the following equations:

$$C_{stocks} = \rho_c \times d \times A \quad (13)$$

$$N_{stocks} = \rho_n \times d \times A \quad (14)$$

3.7.6 Content of ammonium, nitrate and nitrite

The content of nitrate (NO_3^-), nitrite (NO_2^-) and ammonium (NH_4^+) was analyzed in a $CaCl_2$ extract of the fresh soil the day after soil sampling. The 5 g of wet soil sample from each layer was diluted with 40 ml of 0.0125 M $CaCl_2$ solution and then shaken for 1 hour and centrifuged at 8500 rpm. The extraction was filtered using a paper filter MN 615 ¼ (Macherey-Nagel GmbH & Co. KG, Germany), and the nitrate and nitrite contents were measured using liquid chromatography in a HPLC (Agilent technologies 1200, USA).

For the ammonium measurements, the solution was diluted with ultra-pure water (UPW) by a factor of 1:50, and NH_4^+ was determined photometrically at λ = 655 nm using a Thermo Spectro-photometer (Thermo Fisher Scientific Inc., USA).

3.7.7 Microbial biomass

The content of microbial carbon and nitrogen was measured using a $CHCl_3$-fumigation-extraction method as described in Vance et al. (1987). Microbial biomass was determined by the difference in measured values between fumigated and control (non-fumigated) replicates of each sample. For the control analysis, 2 replicates with 6.25 and 12.5 g of each soil sample were placed in plastic bottles and extracted with 100 ml of 0.5 M K_2SO_4 solution according to different soil-to-solution ratios of 1:16 and 1:8 respectively, in order to exclude possible heterogeneity of samples and mistakes in detectable microbial C and N amounts of the measuring instrument. Both extractions were shaken for 30 minutes and filtrated with a paper filter MN 615 ¼ (Macherey-Nagel GmbH & Co. KG, Germany). For the fumigation analysis, 20 g of dried sample with up to 40% of its WHC were weighed in glass beakers and placed in the desiccator in addition to 25 ml of solvent-free chloroform and Zeolite boiling stones. The desiccator then was closed and vacuum conditions were created with a pump for short period of 30 seconds in order to boil the chloroform. Afterwards, the samples were left for 24 hours at 25 °C to kill any microorganisms, and the next day all replicates obtained were extracted with K_2SO_4 by shaking, as done with the control samples. The control and fumigated replicates were all measured with a TOC-L total organic carbon (TOC) and TNM-L total nitrogen (TN) measuring units of ASI-L automatic analyzer (Shimadzu Corporation, Japan). The biomass was

calculated separately for microbial C and N as the difference between measured control and fumigated values using following equations:

$$C_{mic} = \frac{\left(TC_f - TC_{nf}\right)/K_{EC}}{m_{dry}} \quad (15)$$

$$N_{mic} = \frac{\left(TN_f - TN_{nf}\right)/K_{EN}}{m_{dry}} \quad (16)$$

where C_{mic} or N_{mic} is the biomass (mg C or N per g dry substrate), TC_f or TN_f is the concentration (mg/l) of total C or total N from the fumigated sample, TC_{nf} or TN_{nf} is the concentration (mg/l) of total C or total N from the non-fumigated sample, K_{ec} or K_{en} values are the extractable part of the microbial biomass after fumigation equal to 0.45 factor for C and 0.54 for N (Brooks et al., 1985; Wu et al., 1990; Joergensen, 1996), m_{dry} is the dry mass of the tested sample (g).

3.8 Soil incubation experiments

Two soil incubation experiments were conducted under aerobic and anaerobic conditions on peat soils from the extraction and rewetted in 2004 study areas using a common methodology in order to determine the CO_2 and CH_4 production potentials of incubated peat. For each peat sample four treatments were investigated (aerobe and anaerobe) during two batches that differ in their duration and soil solution. The first batch began earlier and was treated with first soil solution, whereas the second batch was developed on the results of first experiment and was treated with second soil solution differed from the first solution in the preparation method of soil pore water extracted from rewetted site RV. In total, both aerobic experiments were incubated for 133 days, and both anaerobic experiments for 427 days. In this work, only results of the second batch are shown.

3.8.1 Soil sampling

The peat samples for incubation were taken on the 23th of March 2015 from two study sites at the Himmelmoor: at the on-going extraction area (E) in three different depths, and on the rewetted in 2004 area with *Eriophorum* and *Sphagnum* vegetation (RV) at one depth. Due to abundant rainfall in the days prior to sampling, the soil was saturated with water and during sampling procedure the water table levels were measured at -40 cm in area E and +10.3 cm in area RV. The peat samples were taken directly inside each microtope at a distance of 2 m from the metal collars used for the GHG flux measurements.

A soil profile was dug at Site E and 2 kg of total sample weight were collected from the first three pedogenic horizons (0-5 cm, 5-19 cm and 19-33 cm depth) and stored in closed plastic bags. At the RV sampling site, an approximately 40 cm long peat core was dug with the help

of a spade, the upper part of which (around 15 cm) consisting of living mosses and plants was removed and hence the first peat horizon was taken from the 15-30 cm depth of water-saturated peat. In addition, two plastic bags were filled with the same sample from 15-30 cm depth to be used for the soil solution extraction. This is described detail in § 3.8.2. All six plastic bags were transported within 2 hours of collection to the Institute of Soil Science and stored in the fridge at +4°C until the start of standard soil analyses, and the first incubation batch started on the 8th of April 2015.

3.8.2 Soil solution sampling and extraction

As described above, during two incubation batches with two solutions of soil pore water were used. These soil solutions were extracted from fresh wet peat samples collected in the RV area from a depth of 0-15 cm.

Two mixed samples used for first soil solution were taken on the 23th of March 2015 and extracted three days later. The first solution was extracted using a glass filter with a pore size of 100 µm (ROBU Glasfilter Geräte GmbH, Germany), and then via a feeding bottle with rubber gasket connected to a pump. The extracted solution was centrifuged for 10 minutes at 4350 rpm and separated in to two parts. The first half was filtered with a 0.7µm pore size pre-combusted glass microfiber filter (GF/F Whatmann, Healthcare Europe GmbH, Germany) and considered to be filtrate with DOC and microorganisms from RV site according to the pore size; and the second half which was filtered with a 0.2 µm pore size polycarbonate filters Nucleopore Track-Etch-Membrane of Whatmann (GE Healthcare Europe GmbH, Germany) and considered to be filtrate containing only DOC without microorganisms due to the small pore size of the filter used (Freeman et al., 2004). Unfortunately, the extraction method used for this solution did not comply with the research questions or experiment conditions due to the lacking initial content on DOC and living microorganisms in the filtrate after using a microfiber filter, and hence the first incubation batch did not answer the working hypotheses.

A second experiment developed on these results was therefore conducted, and the second two mixed samples used for second solution extraction were taken on the 2nd of June 2015 and extracted two days after sampling. The principle used for the extraction was the same as used for first soil solution: pore water was extracted with a pure borosilicate glass filter funnel with a pore size of 100 µm, centrifuged and separated in to two parts in order to get obtain one solution with DOC and microorganisms (DOC+M) and another solution with DOC only (DOC). One half of the solution was kept fresh and intact and was considered to contain DOC and living microorganisms (filtrate 100 µm), whereas the other half was autoclaved at + 120°C and was considered to be a sterile solution containing only fresh DOC (sterile filtrate 100 µm). The autoclaving procedure was conducted over a 20-minute interval with the help of Systec 5050 ELV autoclave (Tuttnauer Europe B.V., Netherlands). Both soil solutions (intact and autoclaved) were preserved in closed dark-glass bottles covered with aluminum foil and stored at +4 °C in the fridge for three days until the start of the experiment. In this work, only results

of the second experiment are shown. The complete description and results of the first experiment under aerobic conditions can be found in the Master's thesis of Höppli (2015).

3.8.3 Aerobic and anaerobic samples preparation and experimental setup

Incubation experiments were established to expose peat material to a constant temperature of +8 °C (the mean annual soil temperature at the site) under either aerobic- or anaerobic conditions for a period of 72 and 365 days respectively. The aerobic incubation formed part of Höppli (2015), and the results of the second aerobic incubation shown in this study were used for comparison with the results of anaerobic measurements obtained during this research. Table 2 gives an overview of all sample treatments used during the first and second experiments.

Peat samples from 3 different depths at Site E and from 1 depth at Site RV were incubated in 4 treatments and in 5 replicates in order to investigate the effect of DOC and microorganisms on the GHG production potential in the tested samples as follows: 1) three control samples from E site (from first, second and third horizon); 2) three samples from Site E with added intact (DOC+M) solution extracted from RV peat pore water; 3) three samples from Site E with added sterile (DOC) solution extracted from RV peat pore water; and 4) a control sample from the topsoil of Site RV. It was suggested that the samples from RV site have a higher GHG potential and that the addition of intact solution containing microorganisms from the RV would have positive effect on the GHG production in control samples from Site E. To compare the possible impact of fresh DOC addition with the impact of living microorganism addition, a third treatment with sterile solution was applied. The additional two treatments were served with ultra-pure water (UPW) and incubated as control treatments.

Before incubations started, all samples were prepared (homogenized and large roots were removed) and analyzed for soil physical and chemical parameters firstly to determine the water holding capacity and water content. As the water content, or in other words the liquid phase, of peat material varied between both study sites and between soil horizons, it was important to adjust all peat samples further used for incubation to the same moisture conditions as the reference. To do this, the amount of liquid phase of each sample was calculated, as well as the dry mass of peat material which was calculated by weighting of exact wet mass according to water content. The use of an equal quantity of dry mass during the incubation makes it sure that the amount of peat material available for decomposition is equal across all samples. However, in order to avoid levels of > 3% CO_2 being reached in the headspace of the aerobe bottles and to prevent possible errors, it was necessary to control GHG production inside the bottle by limiting of the quantity of reactive material. The water content of all samples used for aerobic incubation was adjusted to a water holding capacity of 40-60%: the value of 2 g was chosen after the determination of the moisture content of each layer showed (for a random quantity of 15 g fresh field peat material) the lowest dry soil mass to be 1.4 g. This value was rounded up to 2 g and considered to be sufficient to react significantly according to previous incubations of organic soils (Ivaschenko et al., 2015).

For the aerobic experiment, the ratio between the solid and liquid phase in samples was taken as 1:1, meaning that for a 2 g of dry mass, the quantity of the soil solution or ultra-pure water was fixed at 2 ml. This quantity was the same for each treatment and allowed to keep peat material inside the bottle by 60% of maximum water holding capacity that is the main condition for optimal aerobic activity (Linn and Doran, 1984; Creamer et. al., 2014). Depending on the initial water content of the maximum water holding capacity, the samples were dried down or rewetted. Finally, 2 g weight was subtracted which represents the additional treatment that will be added, and it resulted the weight the sample had to be before adding the treatment. The difference between the weight with and the weight without the 2 g indicated if the sample had to be dried or rewetted, and how much. In order to determine the effect of the drainage at Site E, a sub-sample from the upper 0–5 cm depth, was dried by 25% moisture of the maximal WHC before being rewetted. The 20-40 g of the prepared samples corresponded to 2 g of the dry mass were collected in 250 ml glass bottles (DURAN Group GmbH, Germany) and diluted with different treatments (2 ml of UPW, DOC+M or DOC solution) in 5 replicates for each treatment. After the treatments were added, the bottles were closed with butyl rubber stopper and flushed by pumping the air out and re-filling them 5 times until 1,040 mbar of synthetic air in order to provoke confident air composition and available oxygen for aerobic microorganisms and hence ensure that every change of air composition will be considered as a result of the incubation. The first batch started on 08.04.15 and around 100 bottles were incubated in a dark environment for 56 days at a temperature of +8 °C.

For the anaerobic experiment, the same amount of 2 g dry mass as used for the aerobic samples was selected in order to keep the same quantity of reactive material. The ratio between solid and liquid phase was taken as 1:20 in order to maintain anaerobic water-saturated conditions during the experiment. The anaerobic CO_2-free water used for this experiment was produced by boiling fresh UPW water (bi-distilled water) for 20 minutes and subsequently cooling it under pure N_2 flushing. Depending on the initial water content of each sample, the 8-22 g fresh peat samples (corresponded to 2 g dry mass) were weighed and 5 replicates were placed in 250 ml closed glass bottles inside an oxygen-free glove-box filled with pure N_2. The oxygen-free atmosphere was maintained and the bottles were rewetted with up to 34 ml of anaerobic UPW water, depending on the amount of liquid phase inside of the samples. A further 2 ml treatment (UPW, DOC+M or DOC solution) was added, meaning that there was a total of 40 ml water in each bottle. The prepared bottles were closed with butyl robber stoppers and flushed 5 times with N_2 gas using a vacuum pump (as done with the aerobic bottles). After flushing, the air pressure inside the bottle reached 1,700 mbar and air headspace was considered to contain 0% GHG, and a total of 60 bottles were placed in dark, dry fridge again at + 8°C for the next 56 days. This temperature was chosen due to the mean annual soil temperature in the Himmelmoor of + 9°C. During the first incubation, the obtained results did not show the expected impact of added soil solution enhancing GHG production. It was suggested that since bacteria live on the soil aggregates and those were filtered out, the microorganisms were no-longer present in the first soil solution. Based on these results, the second experiment was developed and a new second solution was extracted without filtration of <100 μm as described in § 3.8.2.

Table 2. Overview of the first and second incubation experiments under aerobic and anaerobic conditions. First and second batches differed in preparation of the soil solution and the second experiment started after 2 months of first incubation on the 08.06.2015. Soil solution №1 was filtered through a filter with pore size of 0.7 µm (DOC+M) and 0.2 µm (DOC), while soil solution №2 was filtered through glass filter funnel with pore size of 100µm (DOC+M) and one half of the filtrate was autoclaved (DOC).

Horizon	Designation	Conditions	Treatment	№ soil solution	Start and end of incubation		Duration in days
Extraction site (53°44.488' N, 009°50.713' E)							
Hhv	E-Hhv1	Oxic	Control	-	08.04.15	20.08.15	133
0-5 cm			DOC+M	1	08.04.15	04.06.15	56
				2	08.06.15	20.08.15	133
			DOC	1	08.04.15	04.06.15	56
				2	08.06.15	20.08.15	72
		Anoxic	Control	-	08.04.15	08.06.16	427
			DOC+M	1	08.04.15	04.06.15	56
				2	08.06.15	08.06.16	366
			DOC	1	08.04.15	04.06.15	56
				2	08.06.15	08.06.16	366
	E-Hhv 1 10% WHC	Oxic	Control	-	25.05.15	20.08.15	86
	E-Hhv 1 25% WHC	Oxic	Control	-	08.04.15	20.08.15	133
			DOC+M	1	08.04.15	20.08.15	133
			DOC	1	08.04.15	20.08.15	133
Hhv2	E-Hhv2	Oxic	Control	-	08.04.15	20.08.15	133
5-19 cm			DOC+M	1	08.04.15	20.08.15	133
			DOC	1	08.04.15	20.08.15	133
		Anoxic	Control	-	08.04.15	08.06.16	427
			DOC+M	1	08.04.15	04.06.15	56
				2	08.06.15	08.06.16	366
			DOC	1	08.04.15	04.06.15	56
				2	08.06.15	08.06.16	366
Hhr	E-Hhr	Oxic	Control	-	08.04.15	20.08.15	133
19-33 cm			DOC+M	1	08.04.15	04.06.15	56
				2	08.06.15	20.08.15	72
			DOC	1	08.04.15	04.06.15	56
				2	08.06.15	20.08.15	72
		Anoxic	Control	-	08.04.15	08.06.16	427
			DOC+M	1	08.04.15	04.06.15	56
				2	08.06.15	08.06.16	366
			DOC	1	08.04.15	04.06.15	56
				2	08.06.15	08.06.16	366
Rewetted site (53°44.652' N, 009°50.817' E)							
jhH1-2	RV-jhH	Oxic	Control	-	08.04.15	20.08.15	133
15-30		Anoxic	Control	-	08.04.15	08.06.16	427

The aerobe and anaerobe bottles with control samples diluted with UPW were incubated and left to rest, whereas all anaerobe and some of aerobe bottles with first soil solution were reused for the second experiment due to their relevance inside the first batch. The aerobe EX-Hi 2 samples showed very similar results to the EX-Hi 1 samples and were not treated with new solutions. The bottles selected for the new experiment were opened and stored either under warm oxygen-free conditions for anaerobe samples or using the normal room atmosphere for aerobe samples in order to evaporate at least 2 ml of water. After 4 days, 2 ml of new solutions were added to the samples (as conducted in the first experiment), and the bottles were closed and flushed as described previously. Additionally, one new sample, EX-Hi 1, was dried up to 10% water content of WHC, rewetted and incubated aerobically in 5 replicates. Furthermore, 2 ml of new soil solutions (DOC+M and DOC) in 3 replicates were incubated in the same way as the other bottles, to allow for an assessment of the blank effect of treatments on GHG production rates.

3.8.4 Physical and chemical analyses of the incubation material

The sub-samples of incubated peat material were taken on the 23th of March 2015 in the Himmelmoor as described previously, and analyses of pH, electrical conductivity, water content, water holding capacity, content of microbial carbon and nitrogen (biomass) and content of nitrites, nitrates and ammonium were carried out together with Master's student Laure Höppli in April and May 2015 using the same methods as described in §3.7. The analyses of C, N content and microbial biomass were repeated after 133 days of aerobic- and 427 days of anaerobic incubation. In addition, the content of ammonium, nitrates and nitrites in both soil solutions was measured as described in §3.7.6. Furthermore, the dissolved C and N were determined in the first solutions using TOC-L and TNM-L analyzer and in the second solutions by drying the water out of 10 ml of each filtrate and measuring the dry resting material using the combustion method with the Vario max analyzer (Elementar Analysensysteme GmbH, Germany).

3.8.5 Measurement of gas production

Gas production was measured regularly: every second day at the beginning of incubation during the first four weeks –due to higher GHG production; once per week after first month; and finally, every two weeks. For the anaerobic experiment, after 6 months of incubation, production was measured at monthly intervals. During all incubation experiments, the incubated bottles were taken from the fridge for maximum of 6 minutes to measure the initial pressure and to take gas samples from the headspace.

Air pressure was measured with a manometer (Leo 1, Keller, Switzerland) using hollow single-use needles (Sterican®, size 12, B. Braun Melsungen AG, Germany) inserted through the butyl rubber stopper. If the measured pressure was lower than normal air pressure of 1000 mbar, 20 ml of N_2 gas were injected in the bottle headspace. After this, the gas inside the bottle was

mixed with a new needle connected to 1 ml plastic syringe (Becton Dickinson S. A., Spain) by pumping gas in and out. Before mixing, the syringe was pre-cleaned with 0.2 ml of the same gas sample. An amount of 1.0 ml of mixed gas sample was taken from the headspace of the bottle with the same needle and syringe and injected via a loop in a GC Agilent 7890 A with ECD detector (see §3.5.2) where gas concentrations of CO_2, CH_4 and N_2O were analyzed. Six standard gases were measured before and after the incubation of samples in order to check the GC calibration. After injection, the measurement of air pressure inside the bottle was repeated with a manometer and used in further calculations.

3.8.6 Calculation of gas production

The quantity of gas produced during incubation was calculated according to Henry's gas law, using the gas concentration measured in the headspace, peat mass, bulk density, incubation temperature, initial pressure and headspace volume of the bottle (Sander, 1999, Knoblauch et al., 2013). The value of bulk density was calculated as a mean measured for each horizon, and was used the same one for as all, aerobic and anaerobic, experiments.

The total sum of gas produced in the bottle was determined by adding equations 17, 18 and 19. As a first step, gas in the headspace was calculated from the concentration in the headspace, the pressure and temperature, and calculated by the ideal gas law. According to following equation the amount of gas produced in the bottle (in µmol per g of dry weight) was calculated as:

$$n_{gas} = \frac{V_g \times P}{V_m \times C_{gas}} / m_{dry} \quad (17)$$

where n_{gas} is the amount of gas produced in µmol of gas per g of dry weight, V_g is the volume of gas space in the bottle given in ml, P is the pressure in the headspace before sampling in mbar, C_{gas} is the measured gas concentration in ppm, V_m is the molar volume in ml at +8 °C and 1 bar (23378 ml), m_{dry} is the dry weight of the sample in g.

Depending on the solubility coefficients for different gases, the amounts of dissolved gases were calculated from the respective solubility of carbon dioxide and methane (according to Carroll and Yamamoto), considering temperature, pressure and water content using the following equation:

$$n_{gas_{(d)}} = \frac{K_{gas} \times WC_{soil} \times C_{gas}}{P} / m_{dry} \quad (18)$$

where $n_{gas_{(d)}}$ is the quantity of dissolved gas in µmol per g of dry weight, K_{gas} is the gas solubility in water in mol L^{-1} bar^{-1} (after Yamamoto et. al., 1976; Carroll, 1991), WC_{soil} is the soil sample moisture in g.

Considering the chemical properties of carbon dioxide, the concentrations of DIC (dissolved inorganic carbon in form of H_2CO_3, HCO_3^-) were determined using dissociation constants from Millero et al. (2007) according to equation 19 using the pH value of the incubated peat samples:

$$n_{DIC} = K_{gas} \times \rho_{CO_2} \times \left(1 + \frac{K_{H_2CO_3}}{[H^+]} + \frac{K_{H_2CO_3} \times K_{HCO_3^-}}{[H^+]^2}\right) / m_{dry} \quad (19)$$

where n_{DIC} is in µmol l^{-1}, $K_{H_2CO_3}$, $K_{H_2CO_3}$ and $K_{HCO_3^-}$ are Henry's Law constants in mol l^{-1} atm^{-1}, ρ_{CO_2} is the partial pressure of CO_2 in atm, $[H^+]$ is the activity of H ions.

The gas production potentials were additionally calculated in µmol of gas per g of SOC content and in µmol of gas per g C biomass.

3.8.7 Data processing and correction

For aerobic experiments, if gas concentration in the bottle headspace reached 3%, the bottle was flushed with synthetic air using the same method described previously in order to ensure enough oxygen was present for living aerobic microorganisms. After the flushing procedure, the concentration of GHG produced was assumed again to be zero.

The amount of gas after the flushing procedure was calculated from the amount before flushing, by adding the last measured concentration (assumed to be the quantity of gas missing in the next measurement) to the new value minus the amount dissolved in the water at the time of flushing, assuming that the CO_2 dissolved in the water was not removed by flushing.

3.9 Statistical analysis

For the gas flux measurements, the flux data calculation was performed from three or four replicates (three collars or three swimming chambers) as a mean ± standard deviation for each study site before statistical analysis. Data were tested for a normal distribution using the Shapiro-Wilk test and for homogeneity using Levene's test. The distribution of GHG fluxes for both locations and differences among the environmental data were tested using the parametric and non-parametric one-way analysis of variances (ANOVA and Kruskal-Wallis test). In addition, the post-hoc comparison was conducted using Tukey's Honestly Significant Differences test and correlation analyses (Spearman's rank-order and Pearson's correlation coefficients) were carried out to determine the effect of abiotic parameters on GHG fluxes. As mentioned before, all $\frac{\partial_c}{\partial_t}$ slopes with flux values close to zero were checked for significance using a T-test. The significance of all tests was accepted at $P < 0.05$. The data quality of incubation experiments was checked using a linear regression as well as the R^2 value for 5 replicates, and the differences between sets and treatments were confirmed by a one-way ANOVA. Significance of these tests was again accepted at $P < 0.05$. All calculations and statistics were computed using the SPSS software (IBM SPSS Statistics 22, IBM, USA), and the Origin software was used for graphical analysis (Origin 9.1, OriginLab Corp, USA).

4 Results

4.1 Environmental conditions

4.1.1 Meteorological conditions

Air and soil temperature

Annual mean air temperatures in Himmelmoor over the measurement periods were 10.2 °C (2014), 9.5 °C (2015) and 9.8 °C (2016), respectively. These temperatures are consistently higher than the long-term average of 8.9 °C calculated for the 40-year reference period 1975-2015 using meteorological data collected in Quickborn (DWD). The highest air temperatures (above 30 °C) were observed in June, whereas the lowest temperatures (below 7 °C) were seen in the months of December and January, when temperatures reached a minimum of -12.6 °C in 2014.

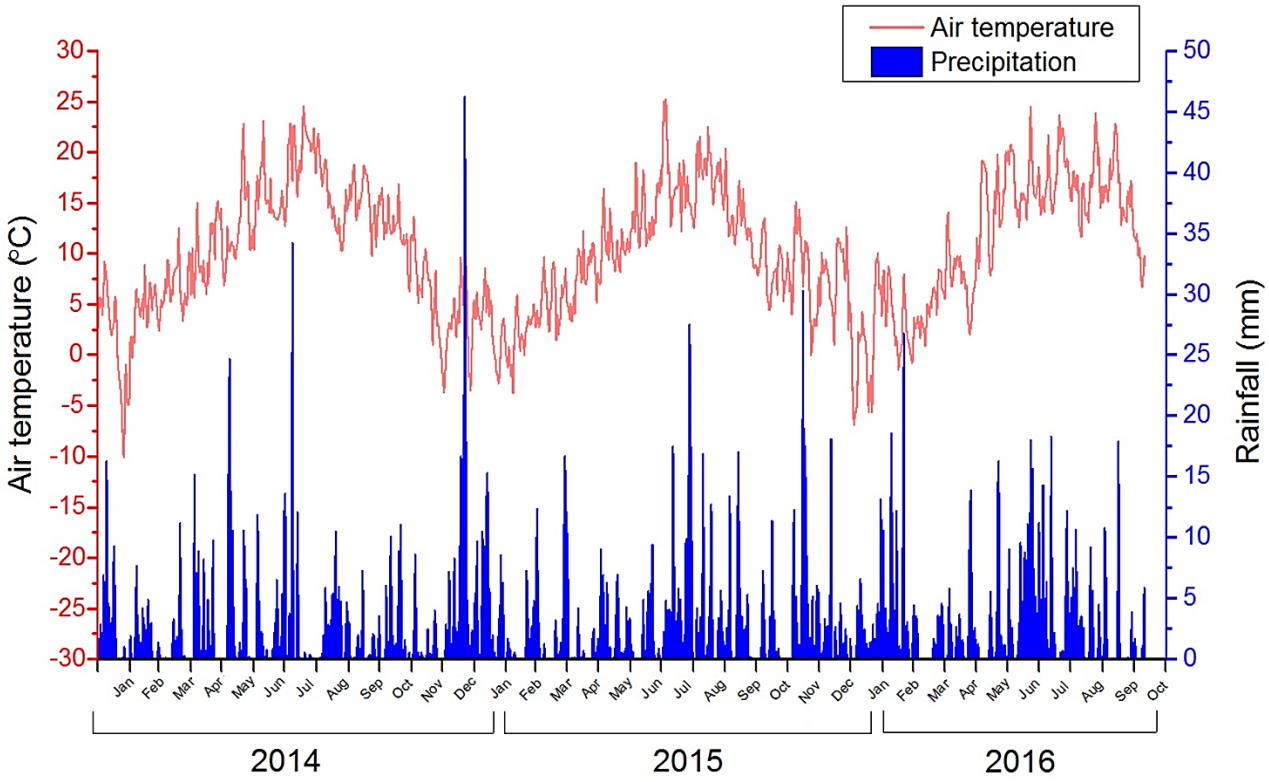


Figure 11. Mean daily air temperature (°C) and total daily precipitation (mm) recorded at the Himmelmoor study site for the period from January 2014 to November 2016.

Mean annual soil temperatures at Himmelmoor measured at 10 cm depth were 10.7 °C in 2014, 10.1 °C in 2015 and 10.4 °C in 2016. The highest seasonal amplitude in soil temperature was detected at the PD site, where soil temperatures ranged from a maximum of 24.3 °C in June 2015 to a minimum of -0.6 °C in January 2014. In contrast, the rewetted areas (RV and R) experienced the smallest seasonal amplitude in soil temperature, with 16.4 °C in summer (RV

site, July 2016) and 3.2 °C in winter (R site, January 2016). Soil temperatures measured at 2, 5, 10 and 20 cm showed a seasonal trend across depth levels at each study site; however as described above for temperature at 10 cm depth, the same differences for dry and wet areas and same amplitudes during warm and cold season were recorded. As seen for air temperature, mean daily soil temperatures were higher in 2014 at all study sites than in the subsequent measurement years. In terms of the seasonal cycle, soil temperatures showed stable soil warming in February-March, when daytime air temperatures did not rise above 0 °C, and freezing in January depending on the snow cover.

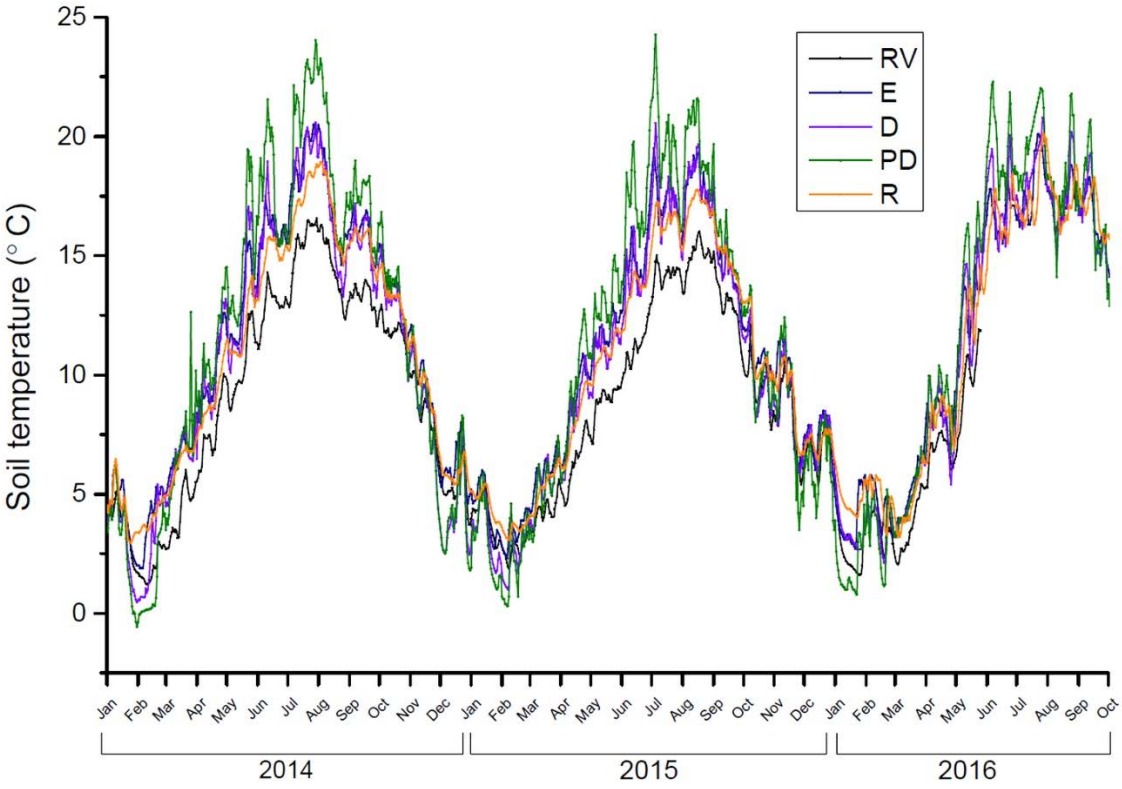


Figure 12. Mean daily soil temperature (°C) recorded at 10 cm depth for all study sites for the period from January 2014 to November 2016 (site abbreviations see in Table 1 and Figure 8).

Precipitation and wind speed

Total annual precipitation measured in Himmelmoor during this study varied between 739 mm recorded in 2016 and a maximum of 847.5 mm recorded in 2015. Mean daily precipitation totals are shown in Fig. 11 and Fig.13. Compared to the long-term rainfall average of 781.4 mm (1975-2015), 2014 experienced conditions which were slightly drier than average, whereas 2015 saw conditions that were considerably wetter than average. In 2014, May and December were the wettest months with 101 mm and 170 mm precipitation respectively, while for 2015 July and November saw the highest rainfall sums (144 mm and 140 mm respectively). The driest months in 2014 were March and September with 25 mm and 23 mm respectively, and in 2015 February and April with 30 mm and 19 mm respectively. During February 2016, 111 mm of rain was recorded.

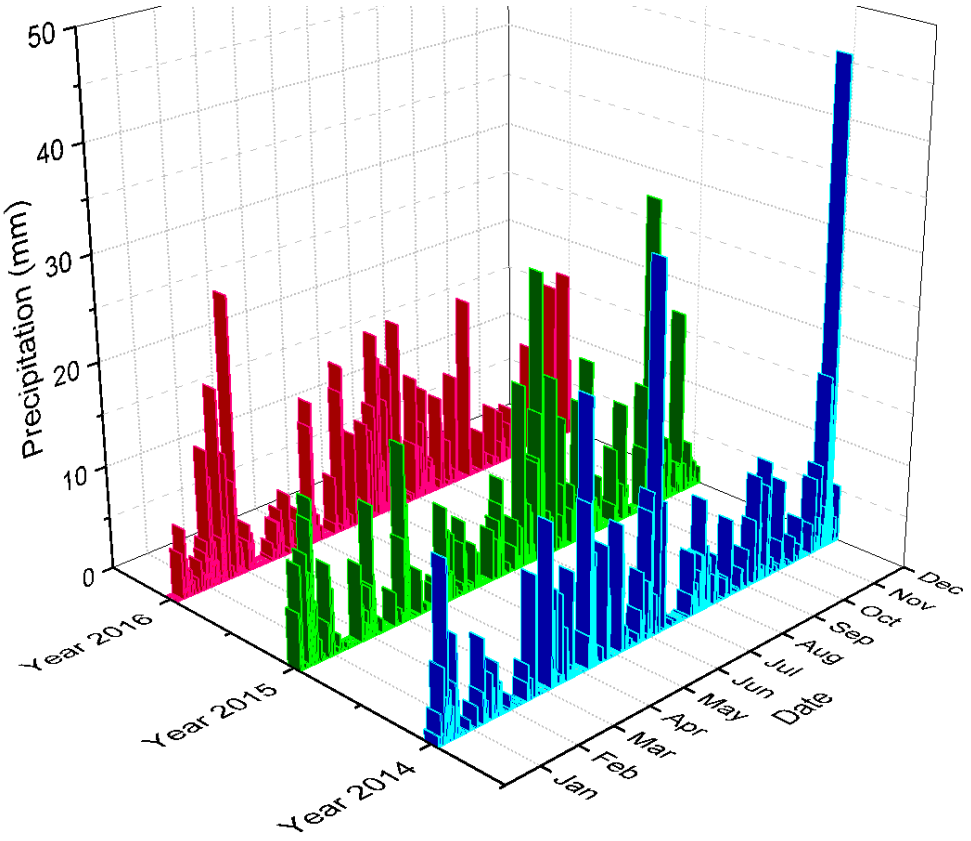


Figure 13. Total daily rainfall in mm measured in Quickborn over the study period (DWD Quickborn, 4039).

The wind speed and wind direction differed at the RV site in comparison to the other study sites, as shown in Figure 14, mainly because of its situation: RV area is bordered in south part by an elevation with birch vegetation and original peat height, called “Knust”, and in north part by a dense birch forest.

The mean wind speed measured by the eddy-covariance tower in the center of the peatland (at Site D) in 2014 was 3.5 m s^{-1} ranging between 0.1 - 11.0 m s^{-1}, and the dominant wind direction

was south westerly. In contrast, winds at site RV came predominantly from westerly and northwesterly as well as easterly directions (Figure 14a), and were generally lower in speeds ranging between 0 - 8.1 m s^{-1}, with a mean of 1.3 m s^{-1} for 2014, and 1.4 m s^{-1} for 2015.

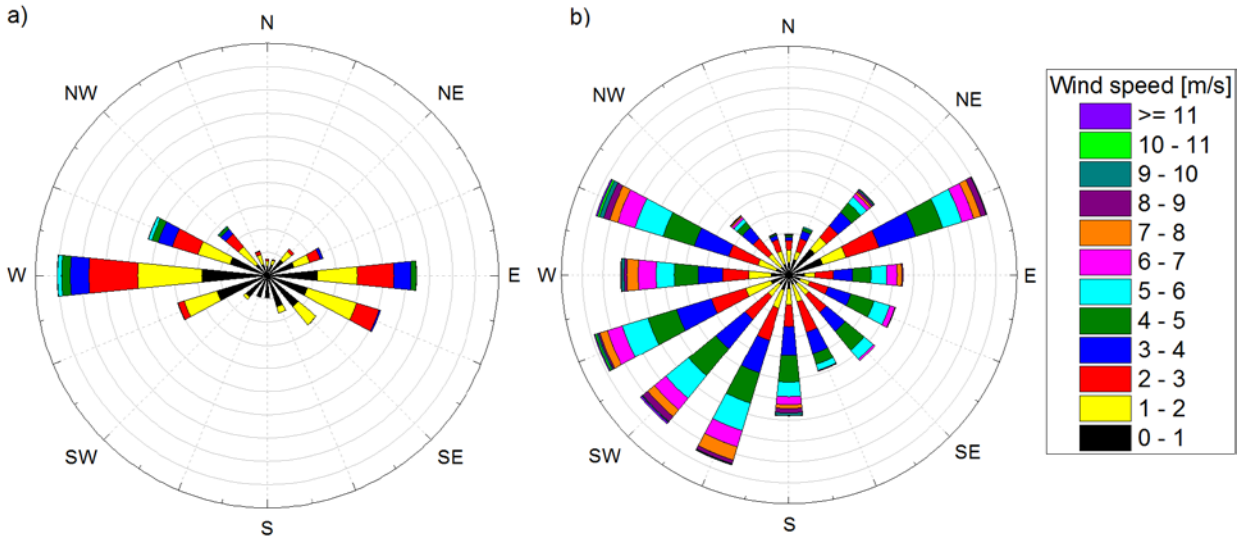


Figure 14. Mean daily wind speed (m s^{-1}) and wind direction for 2014 from two measurement sites: a) the meteorological station at site RV, and b) the eddy-covariance tower in the center of Himmelmoor at site D. Wind speed is indicated with color. The bars length indicates the frequency of occurrence of the wind direction: N = 0◦, NE = 45◦, E = 90◦, SE = 135◦, S = 180◦, SW = 225◦, W = 270◦ and NW = 315◦.

4.1.2 Soil redox potential

The soil redox potentials measured at the five study sites pronounced temporal variability depending on the position of the local water table as well as on precipitation rates for the whole measurement period. Mean daily redox potentials for all study sites at 10 cm depth are shown in Figure 15.

The most pronounced fluctuations of redox potential at 10 cm depth were detected at sites D, R, and RV, where values ranged between -614 and +136 mV. Generally, smaller fluctuations in redox potential as well as minimal precipitation-induced alterations were characteristic for the PD site with values ranging between +164 and +416 mV. A clear reduction of the redox potential was observed over the winter period and after soil thawing at sites E and D. Finally, the redox potentials at sites R and RV were generally lower than at the other sites.

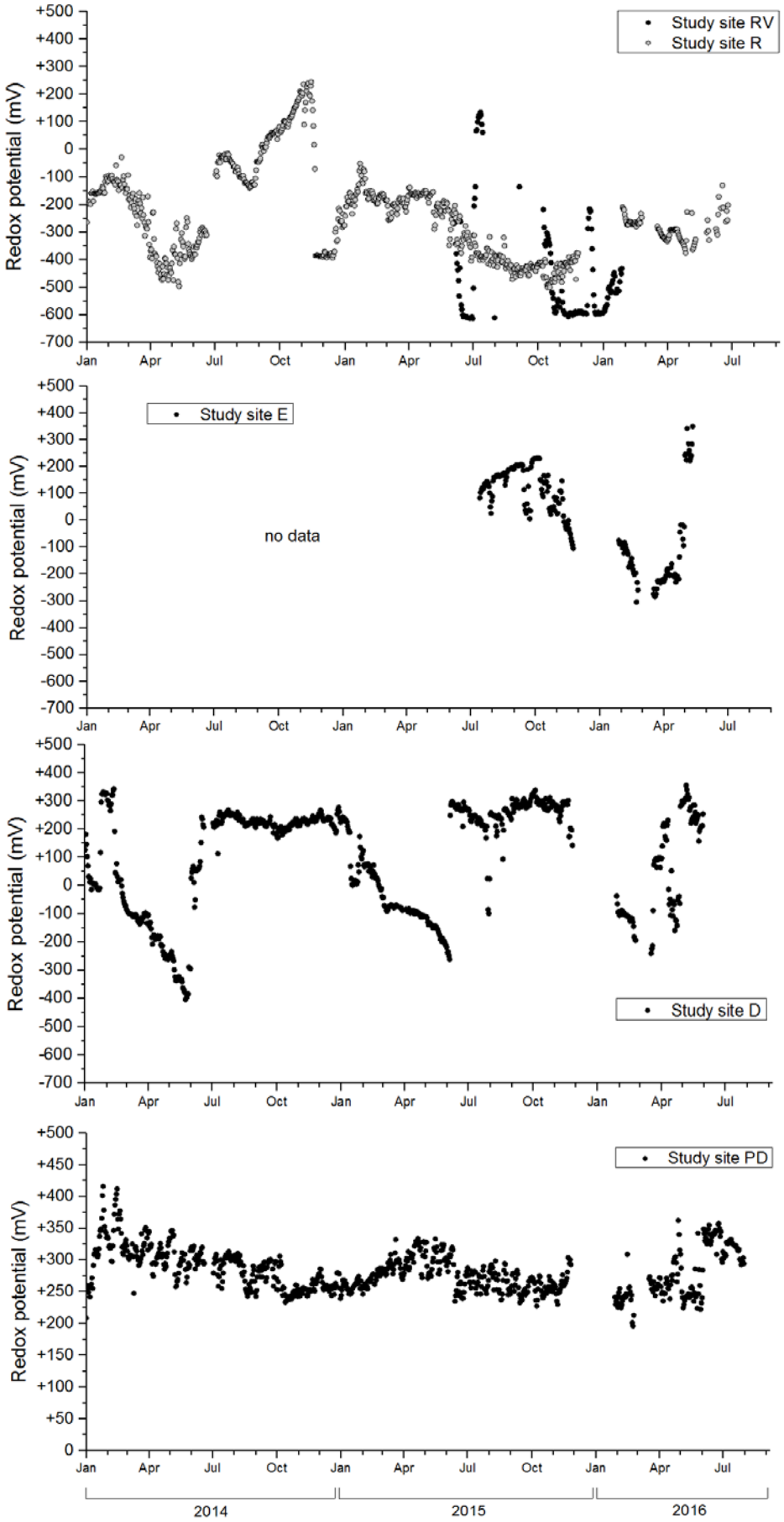


Figure 15. Redox potential in mV for all study sites over period 2014-2016. No data were available for Site E and Site RV site before June 2015.

4.1.3 Water table level

In general, flooding of the soil surface was observed in winter, most often in January, and after strong, prolonged rain events in the autumn. Mean daily WTLs at all study sites were lower in 2014 than in subsequent measurement years.

Water table position was found to vary between study sites, and in general, the rewetted and vegetated areas were characterized by higher WTLs. Site R for example had the highest water table position, followed by sites RV and D. The mean annual WTL at site R was +20.8 cm in 2014, and +28.1 cm in 2015 ranging between -4.2 (September 2014) to +53 cm (February 2014) showing drier conditions in 2014. The water table position at site RV fluctuated between -8.1 cm (August 2014) to +16.7 cm (February 2014) with mean annual values of +4.1 cm in 2014 and +5.1 cm in 2015. Both rewetted sites showed a large range in the amplitude of water table depth variations mostly as a result of vegetation cover. Since the site RV is revegetated by living Sphagnum mosses, which can store large quantities of water inside their cells, the site is consistently wet all year long, while at site R several patchy wet areas were identified during the measurement period.

All four subsites at site D (D1, D2, D3 and D4) were located along a water table level gradient. The field measurements confirmed slight differences in water table depth across these sites, with D4 having the highest WTL, and D1 having the lowest WTL, as described previously in §3.1.2. During the winter period, all four subsites were inundated from January-May, with the maximum WTL being up to +14 cm above the soil surface. Soil drying commenced in June and continued until the first autumn rain events for all study sites.

Variations in WTL at the two peat extraction subsites, E1 and E2, were very similar to those observed at site D, with a minimum WTL of -69 cm and -71.5 cm in October 2014 for the two sites respectively, and a maximum WTL of +1 and +2.1 cm above the surface in December 2014. In contrast, the water table position at site PD was consistently below the surface, with a minimum of -103 cm under the soil surface and a maximum of -65.8 cm.

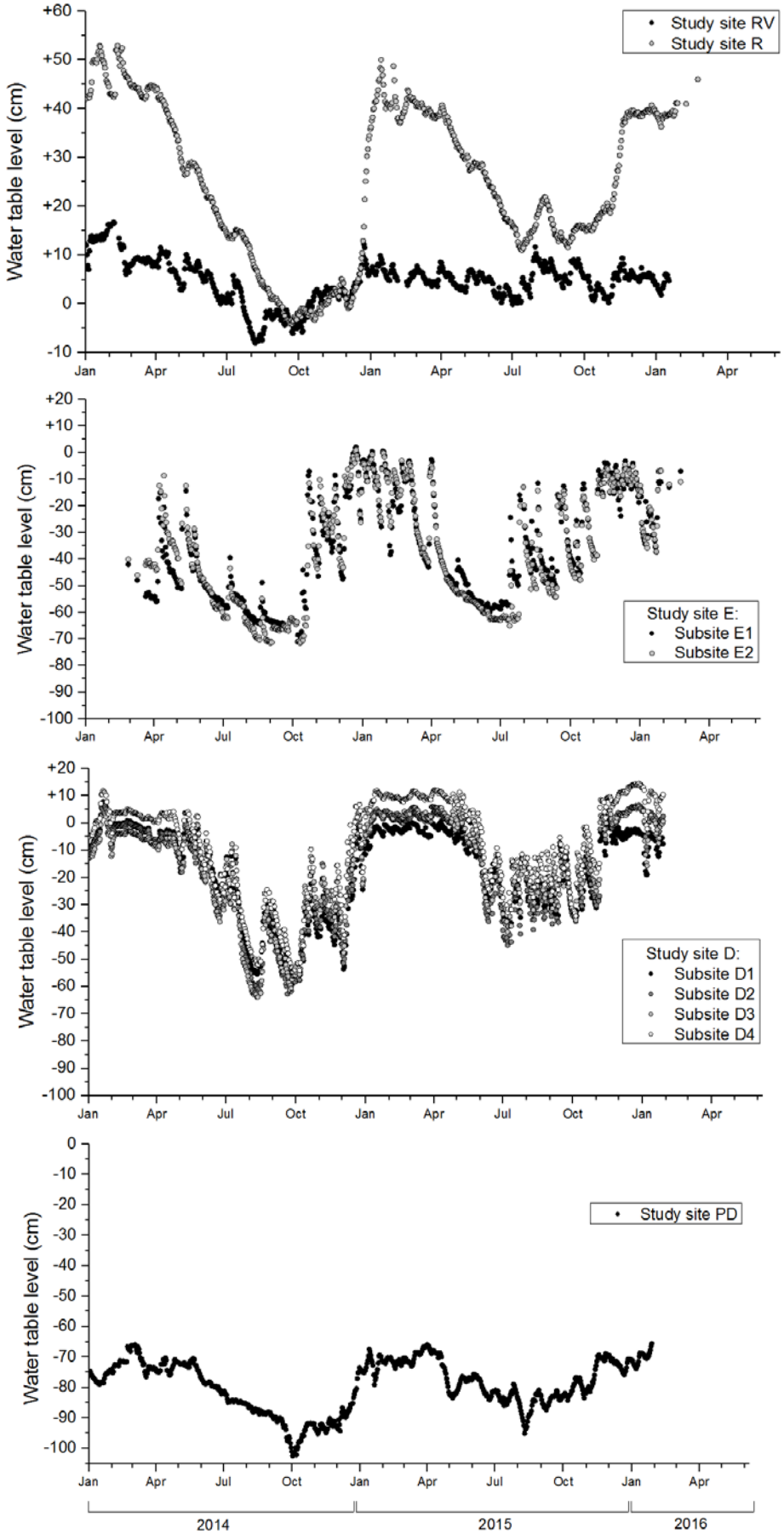


Figure 16. Water table level (cm) for all sites over the period 2014-2016. Negative values indicate WTL below soil surface, whereas positive values indicate inundation.

4.1.4 Vegetation characteristics

Vegetation cover at the two revegetated sites (RV and D) was monitored inside each subsite plot at the start and at the end of this research, and compared with data from previous studies. In general, in January 2014 by inserting the metal collars into the peat soil on both study sites, some plants inside the plots were damaged, but the majority remained undamaged.

Over the study period, the **rewetted site RV** was characterized by typical wetland vegetation, with a dominance of Sphagnum species (*Sph. magellanicum, Sph. fimbriatum, Sph. cuspidatum*), the narrow-leaved cotton grass *Eriophorum angustifolium,* as well as *Erica tetralix, Molinia caerulea* and *Vaccinium oxycoccus*. The average coverage analysis according to Londo (1976) implemented in 2011 also showed that this microtope had in average 98.9% coverage of Sphagnum species represented mainly by *Sph. cuspidatum* and *Sph. fimbriatum* (Vanselow-Algan, 2014). Photo analysis shows that the vegetation coverage of chosen replicates varied within all three subplots (RV1, RV2 and RV3), as reported in Table 3. The cover of *Eriophorum* grass was highest at plot RV1, where coverage increased from 4.7 % in 2014 up to 9.3 % in 2016. The lowest cotton grass coverage was observed at plot RV3, where a maximum coverage of 2.8 % was estimated for 2016. Living cotton grass was deeply rooted in the soil to depths of around 30 cm, and hence reached the anoxic zone in the subsoil even at very low water levels. The highest moss density was found at the beginning of the experiment at plot RV1 and at the end of the experiment at plot RV2.

The vegetation in the refilled with peat **ditch area D** was composed mostly of the narrow-leaved cotton grass *Eriophorum angustifolium*, *Molinia caerulea*, *Juncus effusus*, *Calla palustris*, and to a lesser extent of *Agrostis canina*, *Polytrichum commune*, *Iris pseudacorus*, and *Betula pubescens*. Most of this plant species are not typical bog plants and require very close to the fen nutrient-rich conditions. The vegetation density varied within the four collars and decreased from plot DV1 to DV4. This was not caused by differences in plant abundance or shoot number but rather in the size, coverage and age of the sprouts. The collars extend from the center of *Eriophorum angustifolium* clone patch with older shoots (collars DV1/DV2) to the patch edge region with younger shoots (collars DV3/DV4). The plot DV2 had sparser vegetation coverage than DV1 and it should be noted that at the beginning of the measurement period in 2014 some shoots died, which influenced estimates at the site. This effect could have influence on the GHG fluxes, and thus gas emissions from this area should be compared with other plots. Over the three-year study period, a significant change in vegetation cover was observed for all four subplots at Site D, as shown in Table 4.

Table 3. Percentage of total collar vegetation cover across site RV over the three study years.

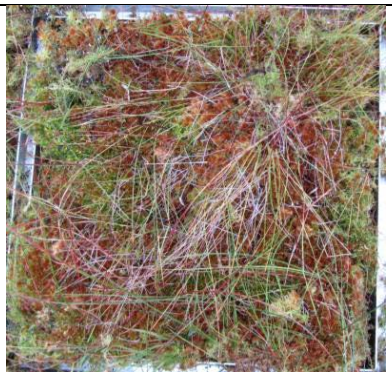
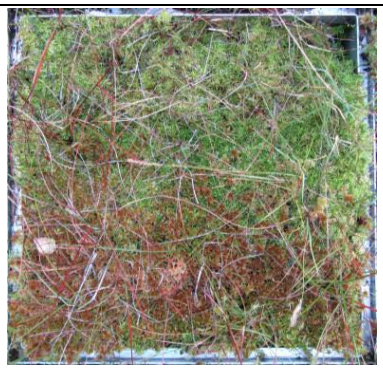
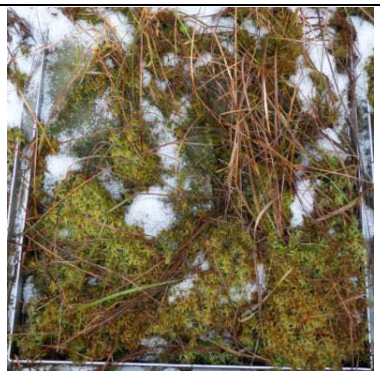
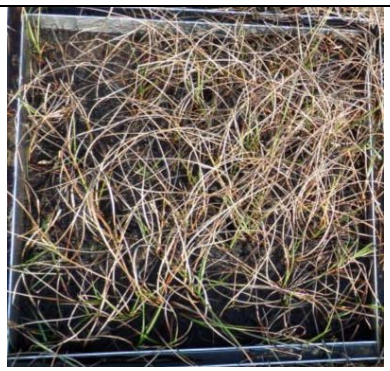
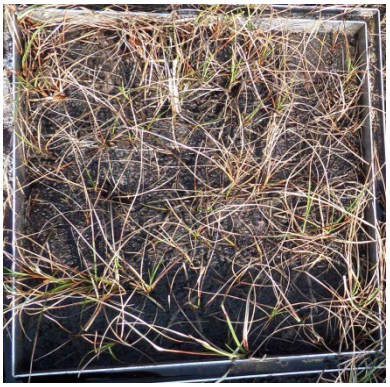
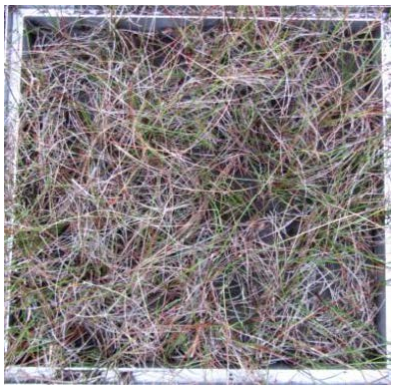
		
RV1, 01.02.2014 Sphagnum: 95.3 % Eriophorum: 4.7 %	RV1, 28.12.2015 Sphagnum: 94.9 % Eriophorum: 5.1 %	RV1, 06.10.2016 Sphagnum: 90.7 % Eriophorum: 9.3 %
		
RV2, 01.02.2014 Sphagnum: 31 % Eriophorum: 2.9 %	RV2, 28.12.2015 Sphagnum: 51.3 % Eriophorum: 3.1 %	RV2, 06.10.2016 Sphagnum: 96.7 % Eriophorum: 3.3 %
		
RV3, 01.02.2014 Sphagnum: 77.6 % Eriophorum: 1.2 %	RV3, 28.12.2015 Sphagnum: 81.1 % Eriophorum: 1.9 %	RV3, 06.10.2016 Sphagnum: 97.2 % Eriophorum: 2.8 %

Table 4. Percentage of total collar vegetation cover across site DV over the three study years.

		
DV1, 01.02.2014 Eriophorum: 38.5 %	DV1, 15.08.2014 Eriophorum: 37.7 %	DV1, 06.10.2016 Eriophorum: 41.2 %
		
DV2, 01.02.2014 Eriophorum: 30.6 %	DV2, 15.08.2014 Eriophorum: 27.2 %	DV2, 06.10.2016 Eriophorum: 31.4 %
		
DV3, 01.02.2014 Eriophorum: 21.8 %	DV3, 15.08.2014 Eriophorum: 23.9 %	DV3, 06.10.2016 Eriophorum: 30.3 %
		
DV4, 01.02.2014 Eriophorum: 10.9 %	DV4, 15.08.2014 Eriophorum: 12.7 %	DV4, 06.10.2016 Eriophorum: 29.8 %

4.2 Soil characteristics and classification

4.2.1 Soil morphology and soil profiles

Soil profiles were found to differ across the five study sites, indicating their contrasting historical and present land-use, stratigraphy, color, moisture and grade of humification. Soil descriptions and profiles of the study sites are shown in tables 5-9 and are described below.

The original peat stratigraphy at sites RV, D and PD was lost as a result of the former anthropogenic activities including mixing, covering/refilling of white peat material, and peat extraction. In contrast, peat layers at sites R and E, were slightly disturbed only in the upper horizon, with the other layers showing natural stratigraphy, and the upper peat was naturally allocated mainly due to water transport and wind erosion. This is confirmed by a clear shift in peat conditions of all pedogenic horizons indicated by color, type of plant residues and degree of humification, which are described in tables 5-9.

Pedogenic horizons of all soil profiles were described using the German classification system (Ad-hoc-AG Boden, 2005). The studied soils were accordingly described as “Hochmoor” (HH) and “Erdhochmoor” (KH), being saturated by rainwater, having peat extending to over 30 cm depth, and being dominated by organic material formed by accumulations of undecomposed or partially decomposed organic material. The main difference between the HH and KH soil types is that typical KH soils are characterized by aerobic process of enhanced humification and soilification under dry conditions in the upper 10-20 cm of the peat, also called “Vererdung” in German. This process is characterized by changes in peat color and structure that were detected in the upper horizons of soils at sites D and PD (dark brown to black color and crumbly structure).

According to the World Reference Base (IUSS Working Group WRB, 2014) and US Soil Taxonomy (Soil Survey Staff, 2014), the soil profiles were classified as “Fibric Ombric Histosols” (Fibric: two thirds recognizable plant tissue; Ombric: saturated predominantly by rainwater; Histic: soils with a thick organic layer), and as “Typic Sphagnofibrist” (Order: Histosols; Suborder: Fibrist; Subgroup: Sphagnofibrist), respectively. A Histosol has a surface layer which consists of partially decomposed plant remains with or without admixed sand, silt or clay (FAO, 2006) and a specific Hi horizon, where the “i” stands for slightly decomposed organic material.

Table 5. Description of the soil profile at Site RV.

RV: Rewetted in 2004, with vegetation

Location: Himmelmoor (53°44.652’ N, 009°50.817’ E)
Date of profile acquisition: 03.05.2016
Water level: 0 cm, all horizons were situated below the water level
Vegetation: *Sphagnum, Eriophorum angustifolium, Erica tetralix, Molinia caerulea*

Figure 17. Soil profile at RV study area (modified after Vanselow-Algan, 2014).

Soil type:

- German classification: Normhochmoor aus umgelagertem, durchmischten Sphagnum Torf (HHn: om-Hhs-qhi)
- WRB classification: Ombric Fibric Histosol
- US Soil Taxonomy: Typic Sphagnofibrist

Horizon number	Sampling horizon*	Depth	Color **	Grade of humification ***	WC (%)	Peat decomposition	Remarks
1	jhH1	0-8	2.5 YR 6/4	H1	91.2	Poorly decomposed	Sphagnum peat
2	jhH2	8-18	7.5 YR 2.5/1	H2	94.5	Poorly decomposed	Sphagnum peat
3	jhH3	18-55	2.5 YR 2.5/3	H3	91.4	Poorly decomposed	Sphagnum and cotton grass peat
4	jhH4	55-100	2.5 YR 2.5/2	H4-H5	93.8	Moderately decomposed	Sphagnum and cotton grass peat

* denotation from German classification system
** Munsell Soil Color Chart (2014)
*** denotation from Von Post and Granlund (1926)

Table 6. Description of the soil profile at Site E.

E: Extraction site

Location: Himmelmoor (53°44.488’ N, 009°50.713’ E)
Date of profile acquisition: 03.05.2016
Water level: -50 cm
Vegetation: absent

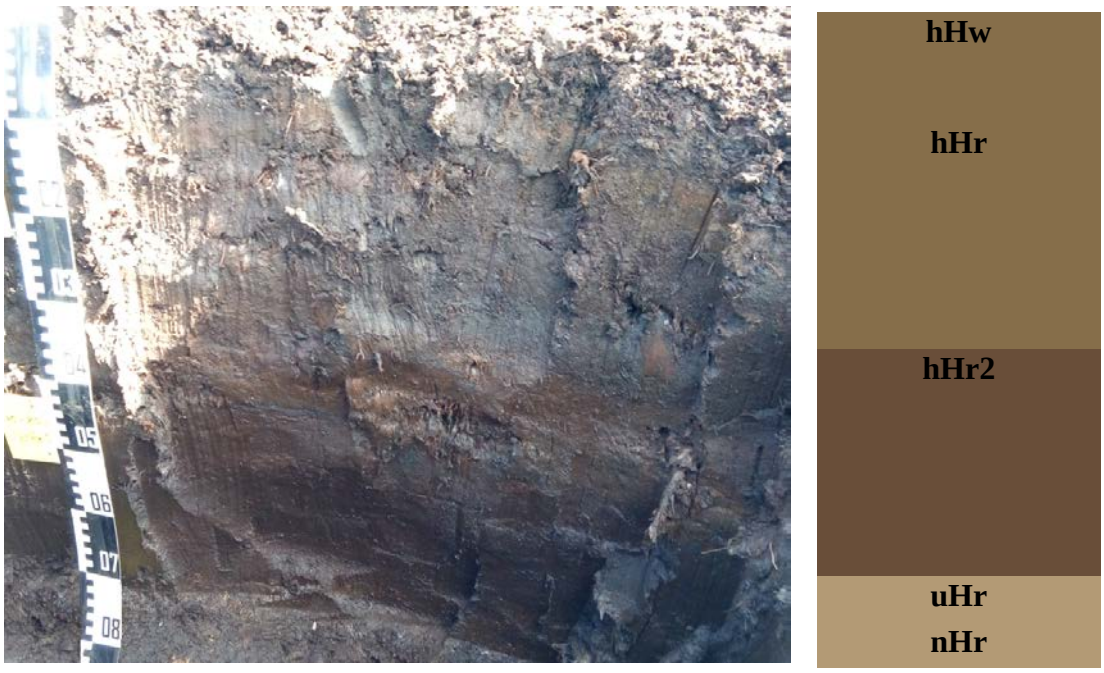


Figure 18. Soil profile at E study area (Vybornova, 2016)

Soil type:

- German classification: erodiertes Normhochmoor aus organogenem Sphagnum Torf (eHH: og-Hhs-qh)
- WRB classification: Ombric Fibric Drainic Histosol
- US Soil Taxonomy: Typic Sphagnofibrist

Horizon number	Sampling horizon*	Depth	Color **	Grade of humification ***	WC (%)	Peat decomposition	Remarks*
1	hHw	0-10	5 YR 2.5/1	H7	83.2	Moderately decomposed	Sphagnum and Eriophorum peat
2	hHr	10-42	5 YR 4/4	H7	87.9	Moderately decomposed	Sphagnum and cotton grass
3	hHr2	42-82	7.5 YR 2.5/2	H8	88.8	Strongly decomposed	Amorphous, no definable fibres
4	uHr	82-120	7.5 YR 3/3	H6	89.7	Moderately decomposed	Birch tissue, twigs
5	nHr	120-190	7.5 YR 3/3	H5	-	Moderately decomposed	Sedge fen peat

* denotation from German classification system
** Munsell Soil Color Chart (2014)
*** denotation from Von Post and Granlund (1926)

Table 7. Description of the soil profile at Site D.

D: Ditch refilled with peat

Location: Himmelmoor (53°44.488’ N, 009°50.713’ E)
Date of profile acquisition: 03.05.2016
Water level: -20 cm
Vegetation: *Eriophorum angustifolium*

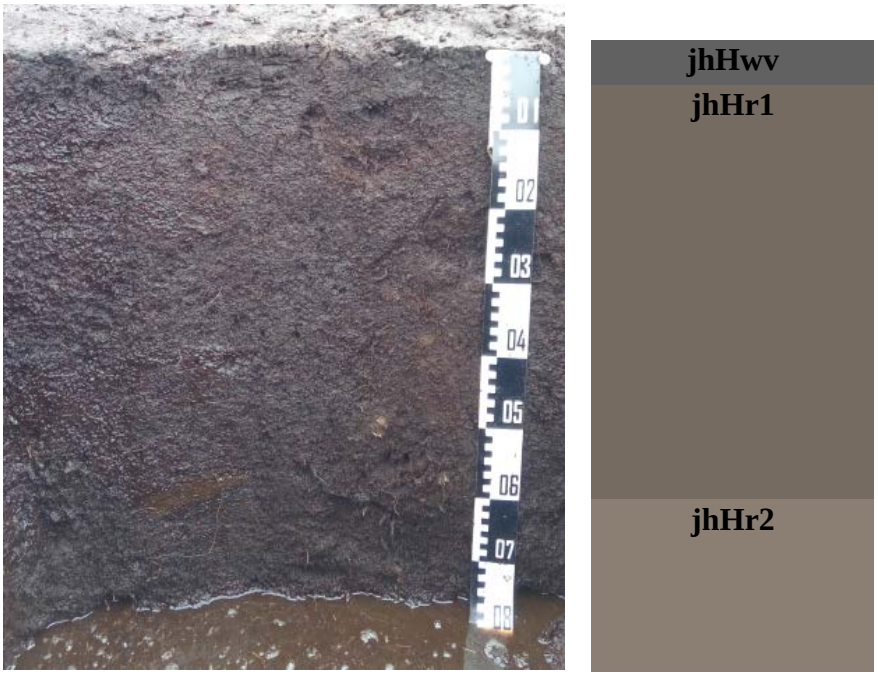


Figure 19. Soil profile at D study area (Vybornova, 2016)

Soil type:

- German classification: Erdhochmoor aus umgelagertem, stark durchmischtem, vererdetem Sphagnum Torf (eKH: og-oj-Hhs-qhi)
- WRB classification: Ombric Fibric Histosol
- US Soil Taxonomy: Typic Sphagnofibrist

Horizon number	Sampling horizon*	Depth	Color **	Grade of humification ***	WC (%)	Peat decomposition	Remarks*
1	jhHwv	0-5	2.5 YR 2.5/1	H6	83.3	Moderately decomposed	Few plant structure tissues, some fibres of Sphagnum, birch and cotton grass
2	jhHr1	5-60	5 YR 3/2 + 5 YR 3/1	H5	87.9	Moderately decomposed	More definable plant tissues, Sphagnum and cotton grass peat
3	jhHr2	60-100	5 YR 3/3	H4	86.2	Moderately decomposed	More dense peat, roots of living plants of cotton grass

* denotation from German classification system
** Munsell Soil Color Chart (2014)
*** denotation from Von Post and Granlund (1926)

Table 8. Description of the soil profile at Site PD.

PD: Peat dam

Location: Himmelmoor (53°44.488’ N, 009°50.713’ E)
Date of profile acquisition: 03.05.2016
Water level: -60 cm
Vegetation: absent

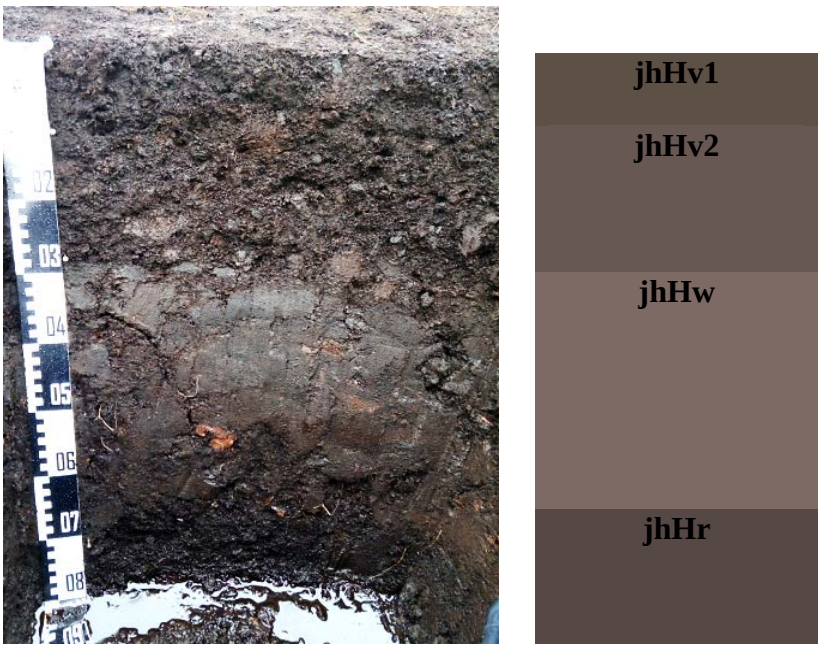


Figure 20. Soil profile at PD study area (Vybornova, 2016)

Soil type:

- German classification: Erdhochmoor aus umgelagertem, stark durchmischtem, vererdetem Sphagnum Torf (eKH: og-oj-Hh-Yi-qhi)
- WRB classification: Ombric Fibric Histosol
- US Soil Taxonomy: Typic Sphagnofibrist

Horizon number	Sampling horizon*	Depth	Color **	Grade of humification ***	WC (%)	Peat decomposition	Remarks*
1	jhHv1	0-8	7.5 YR 2.5/1	H8	68.9	Strongly decomposed	Transported, mixed black peat with Sphagnum, low fen sedge peat
2	jhHv2	8-28	5YR 2.5/2	H8	70.9	Strongly decomposed	Sphagnum, cotton grass and heath grass peat
3	jhHw	28-70	7.5 YR 2.5/1	H7	85.3	Moderately decomposed	Sphagnum, pine and heath grass peat
4	jhHr	70-100	7.5 YR 3/2	H8	86.1	Strongly decomposed	Sphagnum, pine and cotton grass peat

* denotation from German classification system
** Munsell Soil Color Chart (2014)
*** denotation from Von Post and Granlund (1926)

Table 9. Description of the soil profile at Site R.

R: Rewetted in 2009 area

Location: Himmelmoor (53°44.488’ N, 009°50.713’ E)
Date of soil core acquisition: 03.05.2016
Water level: +40 cm, all pedogenic horizons below the water level
Vegetation: no vegetation cover

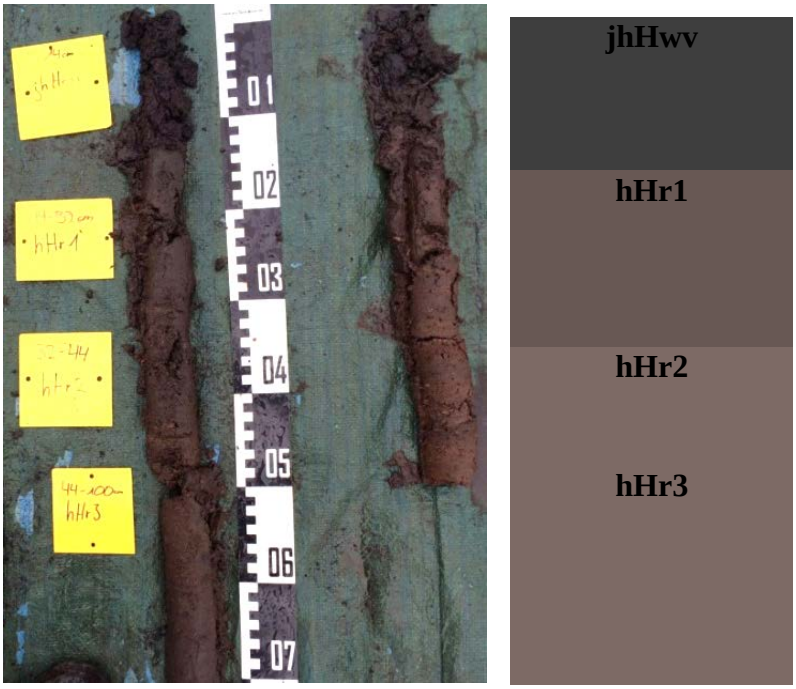


Figure 21. Soil core at R study area (Vybornova, 2016)

Soil type:

- German classification: Normhochmoor aus organogenem, natürlich umgelagertem, vererdetem Sphagnum Torf (eHH: og-oj-Hh-Yi-qhi)
- WRB classification: Ombric Fibric Histosol
- US Soil Taxonomy: Typic Sphagnofibrist

Horizon number	Sampling horizon*	Depth	Color **	Grade of humification ***	WC (%)	Peat decomposition	Remarks*
1	jhHwv	0-14	2.5 YR 2.5/1	H5	80.0	Moderately decomposed	Naturally transported, Sphagnum and cotton grass peat
2	hHr1	14-32	2.5 YR 3/2	H8	89.9	Strongly decomposed	Cotton grass peat
3	hHr2	32-44	5 YR 3/2	H7	90.0	Moderately decomposed	Sphagnum and cotton grass peat, Birch twigs and tissue
4	hHr3	44-100	5 YR 4/2	H7	90.3	Moderately decomposed	Sphagnum peat

* denotation from German classification system
** Munsell Soil Color Chart (2014)
*** denotation from Von Post and Granlund (1926)

4.2.2 Soil-physical and -chemical properties

Figures 22-24 show the mean values and associated standard deviations for all sites across the soil profile. Soil samples from all study sites were found to have low pH which ranged from 3.6 to 4.9, indicating very strongly and strongly acid soil reaction (Scheffer and Schachtsschabel, 2010). Soil electrical conductivity varied between 44 and 163 µS/cm, with the highest values being measured at sites PD and D. A significant correlation was found between pH and electrical conductivity on the vegetated plots (r= -0.98) and site R (r=0.87). Measurements of C and N content reveal a significant difference between rewetted and drained areas. The measured C_{org} ranged between 47.7 % at the rewetted RV site and 59.5 % at site E, whereas the drier drained sites PD and E had higher N content with up to 2.23 ± 0.06% of total N and consequently the lowest C/N ratio. The C/N ratios of all soil profiles ranged between 24 and 58, differed visibly between drained and rewetted plots, and varied with depth (Figure 22). The deeper horizons of non-vegetated areas were found to differ the most from the topsoil in that they have the highest N content and hence the lowest C/N ratio (24 for PD; 27 for E in the deep layers). In the vegetated areas D and RV, the C/N value increased with depth and amounted 48 and 58 for the two sites respectively.

Soil bulk density was relatively low due to a high organic matter content, and was in agreement with other values for peat soils given in the literature. The highest mean bulk density of 0.22 ± 0.01 g cm^{-3} was found in the upper soil layer of site PD (Figure 22), and is significantly higher than found at the rewetted sites (One-way ANOVA: p=0.014), mostly as a result of soil compression by drainage and the use of heavy peat-extraction machinery. Within the soil profiles there was a clear reduction of the mean soil bulk density with depth. Additionally, the lowest amounts of ash and water content as well as maximal WHC for each soil profile were found in the upper 10-20 cm of the examined soils, which are most affected by atmospheric and biospheric influences.

The percentage of soil pore space calculated using bulk density and particle density estimates as described in Tan (1990) showed small differences between samples. Peat samples had a very low particle density between 1.37 and 1.43 g cm^{-3} and high pore space rates ranging from 84.2% in the topsoil of Site PD to 93% in the lower horizon of Site E, with a mean value of 90.14%.

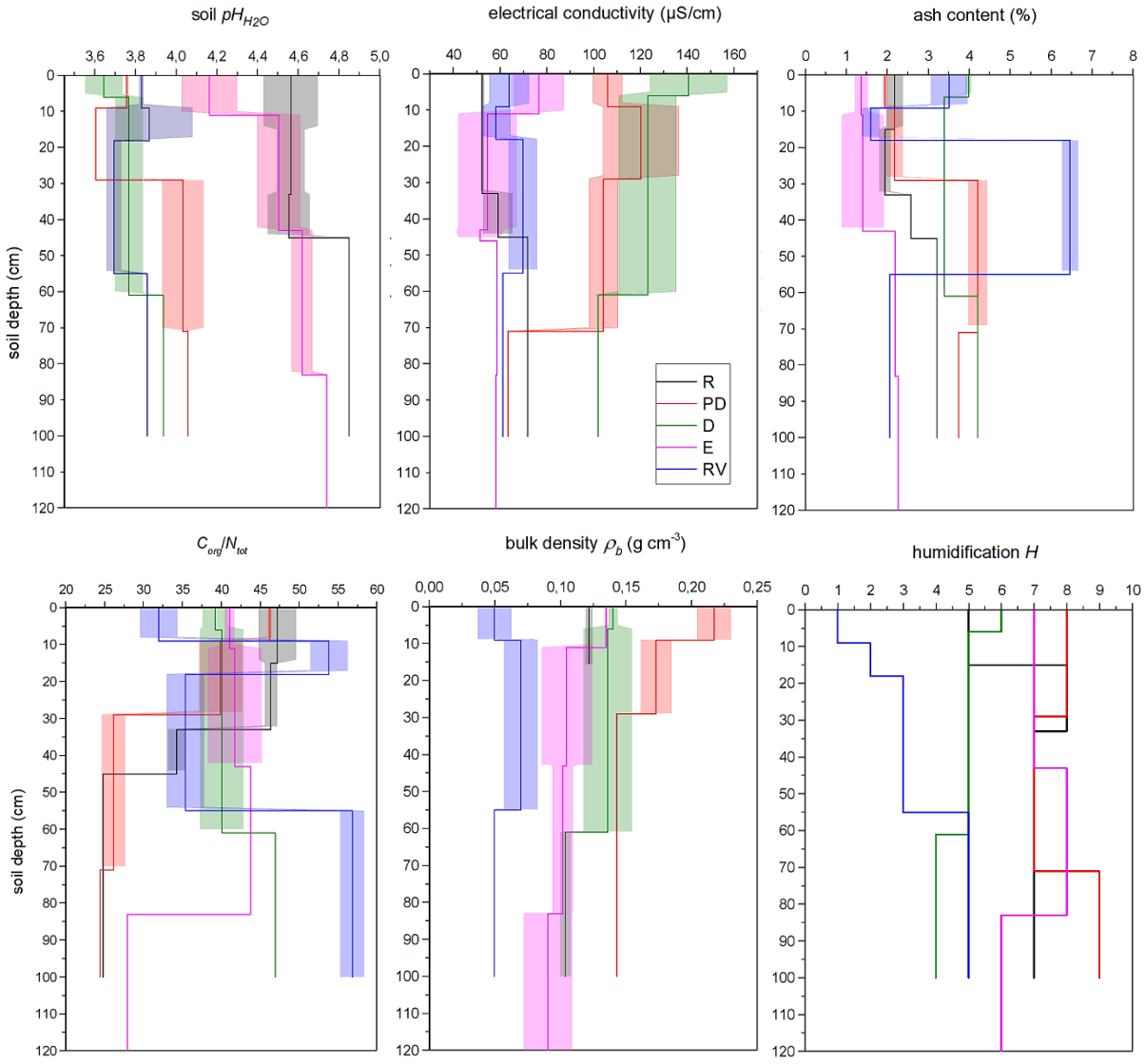


Figure 22. Trends in soil parameters for all study sites with depth. Shown are the soil acidity (pH_{H2O}), electrical conductivity (µS/cm), ash content (%), carbon to nitrogen ratio, bulk density (ρ_b, g cm^{-3}) and humidification grade H of soil samples from 0-120 cm depth. Lines represent mean values observed over measurement period, and filled areas represent standard deviations (n=3) for each soil horizon.

Investigations of bulk density (ρ_b) and humidification grade (H) reveal a strong relationship between these two variables (Figure 23). The determined humidification grades H showed very strong differences between the topsoils of rewetted and non-rewetted areas. Moreover, within the rewetted areas, the upper layer of RV study site had a significantly lower humidification grade of 1 compared to the topsoil of R study site, because of the presence of living mosses and as a consequence of fresh plant residues in the most upper peat horizon.

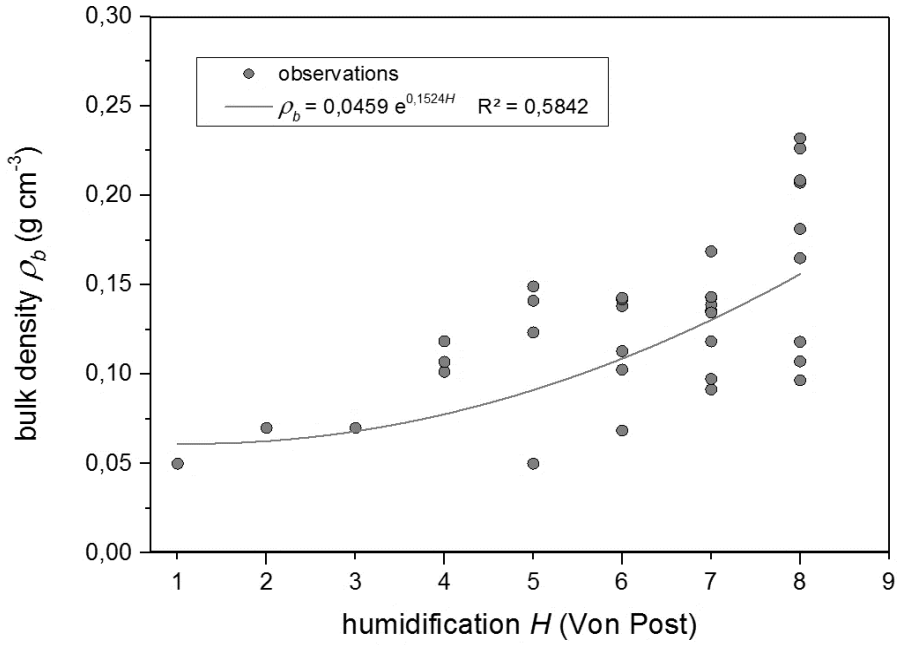


Figure 23. Soil bulk density in g cm^{-3} plotted against humification grade (n=29). Grey line displays an exponential fit (R^2=0.58).

The soil organic carbon stocks (C_{stocks}) estimated for the upper 100 cm soil show much variability between microtope plots (Table 10). The calculated mean SOC stock over the first 100 cm depth for sites R and E were 671.63 ± 2.95 t C ha^{-1} and 623.85 ± 2.93 t C ha^{-1}, respectively, with a minimum of 290.71 ± 2.03 t C ha^{-1} for the rewetted site RV and a maximum of 883.20 ± 2.96 t C ha^{-1} in dry peat dam site PD. The mean total soil N stock estimated over the upper 100 cm depth for all sites is 17.53 ± 0.84 t ha^{-1}, with a median of 15.41 ± 0.19 t ha^{-1}. Clear differences were identified between sites, with total soil N being over 4 times larger at the dry site (PD, 28.98 ± 0.33 t N ha^{-1}) than estimated for the rewetted site (RV, 6.52 ± 0.07 t N ha^{-1}). Similarly as for SOC stock, the main factor here was the soil bulk density and soil compaction in general.

Table 10. Bulk density, pH, carbon density, SOC stocks, nitrogen density, N stocks, microbial C and microbial N determined for all pedological horizons and as a total mean for the upper 100 cm depth for each investigation plot. Results are expressed as mean values ± standard deviation (n=3).

Depth	ρ_b	pH $_{CaCl2}$	ρ_c	SOC_{stocks}	N_{total}	N_{stocks}	C_{mic}	N_{mic}
[cm]	[g cm^{-3}]	[-]	[kg m^{-3}]	[t ha^{-1}]	[kg m^{-3}]	[t ha^{-1}]	[µg cm^{-3}]	[µg cm^{-3}]
R: Rewetted in 2009								
0-14	0.122±0.01	3.0	70.7±0.1	99.0±0.1	1.6±0.01	2.2±0.0	373.8±16.2	3.1±0.0
14-32	-	3.2	70.1±0.4*	126.2±0.7*	1.5±0.01*	2.8±0.0*	583.9±209.6*	1.1±0.0*
32-44	-	3.4	68.6±0.3*	82.4±0.3*	2.0±0.01*	2.3±0.0*	991.1±133.1*	0.0±0.0*
44+	-	3.6	65.1±0.3*	364.5±1.8*	2.6±0.01*	14.7±0.1*	827.0±50.9*	0.0±0.0*
Total amount for the upper 1 m of soil:				***671.9±2.9***		***22.0±0.2***		
PD: Dry peat dam								
0-8	0.222±0.01	2.7	127.1±0.1	101.7±0.1	2.8±0.01	2.2±0.0	356.7±2.3	7.6±1.0
8-28	0.184±0.02	2.7	105.7±0.1	211.4±0.0	2.4±0.00	4.9±0.0	339.6±2.5	2.5±0.1
28-70	0.152±0.01	3.2	82.8±0.3	347.6±1.0	3.0±0.02	12.8±0.1	743.5±8.1	17.6±5.1
70+	0.136±0.01	3.1	74.2±0.3	222.5±0.9	3.0±0.09	9.1±0.3	1295.1±13.2	20.4±2.0
Total amount for the upper 1 m of soil:				***883.2±3.0***		***29.0±0.3***		
D: Ditch refilled with peat								
0-5	0.141±0.00	2.9	72.6±0.1	36.3±0.0	1.90±0.02	0.9±0.0	570.7±24.1	12.3±3.0
5-60	0.138±0.01	2.9	70.7±0.1	388.6±0.2	2.22±0.62	12.2±3.4	720.9±29.2	21.3±2.8
60+	0.109±0.01	3.0	54.3±0.3	108.6±0.6	1.13±0.04	2.3±0.1	552.6±6.47	5.3±1.9
Total amount for the upper 1 m of soil:				***533.5±2.2***		***15.4±3.5***		
E: On-going extraction area								
0-10	0.136±0.00	3.1	74.2±0.7	74.2±0.7	1.82±0.03	1.8±0.0	368.8±7.1	9.3±1.0
10-42	0.102±0.01	3.4	60.6±0.1	193.8±0.1	1.35±0.01	4.3±0.0	341.7±16.3	0.0±0.0
42-82	0.107±0.01	3.4	63.8±0.1	255.4±0.1	1.26±0.01	5.1±0.1	699.3±18.9	0.0±0.0
82+	0.095±0.02	3.4	55.9±0.1	100.5±0.0	2.00±0.01	3.6±0.0	549.1±5.09	25.2±3.0
Total amount for the upper 1 m of soil:				***623.9±2.9***		***14.8±0.1***		
RV: Rewetted in 2004								
0-8	0.050**	2.8	25.5±0.1	20.4±0.1	0.92±0.04	0.7±0.0	378.9±38.3	10.6±2.5
8-17	0.070**	3.0	33.4±0.1	30.1±0.0	0.57±0.01	0.5±0.0	2010.3±14.6	89.4±4.6
17-54	0.070**	2.8	34.1±0.2	126.0±0.6	0.89±0.00	3.3±0.0	720.8±120.9	14.9±7.1
54+	0.050**	2.9	24.8±0.2	114.2±0.7	0.43±0.01	2.00±0.0	576.3±121.4	8.2±1.6
Total amount for the upper 1 m of soil:				***290.7±2.0***		***6.5±0.1***		

* for calculations was used value of bulk density determined for the first horizon (0-14 cm)
** data from Vanselow-Algan (2014)

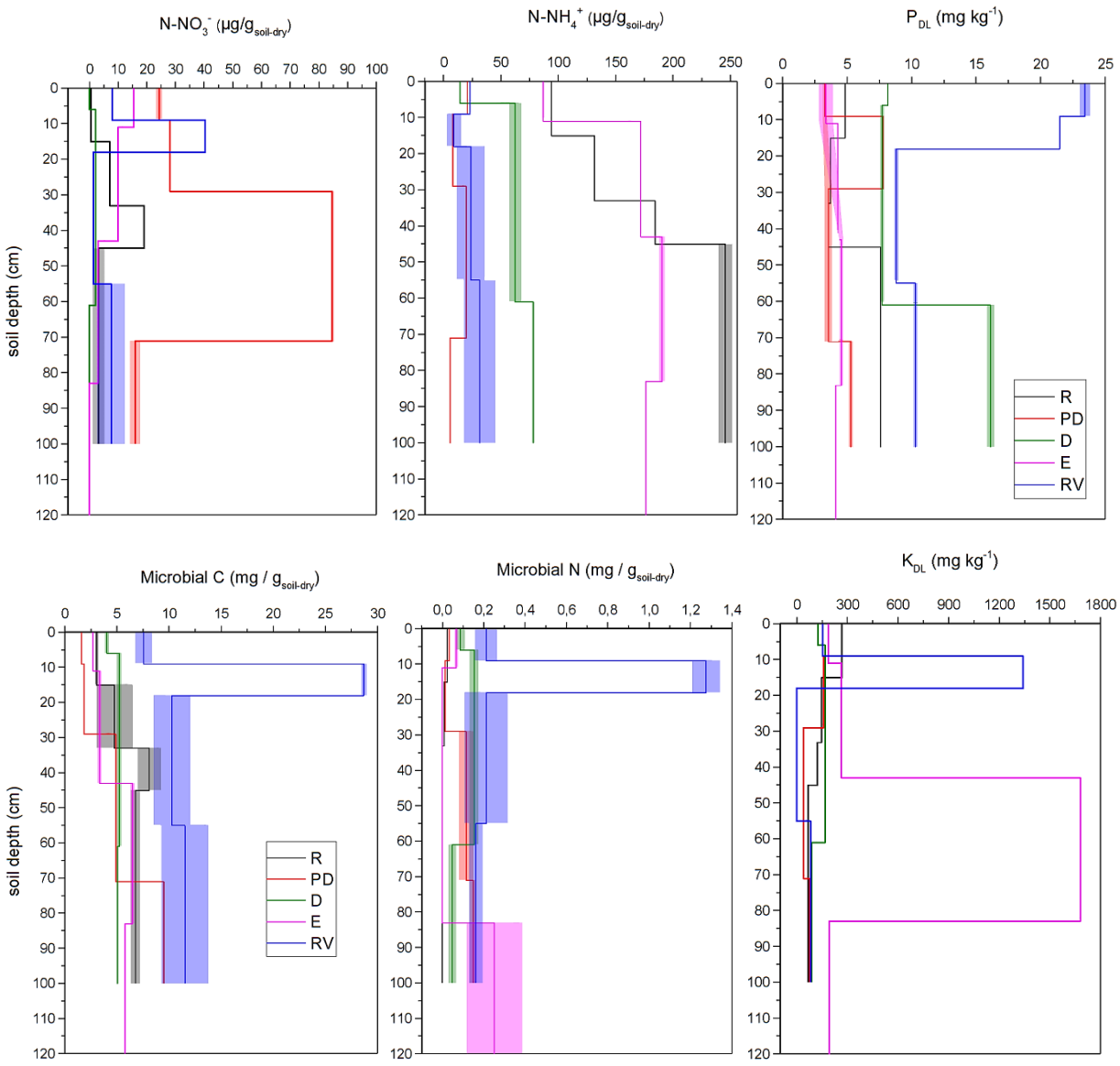


Figure 24. The soil nitrate, ammonium, plant-available phosphorus (P_{DL}) and potassium (K_{DL}) extracted with double lactate as well as soil microbial biomass contents of soil samples shown against soil depth (0-120 cm) for all study sites. Lines display mean values, and the filled areas display standard deviations (n=3) for each soil horizon.

In terms of soil-chemical properties, results show that the soil horizons from the rewetted sites are different to those in the drained areas (Figure 24). Investigations of plant-available phosphorus (P_{DL}) showed that vegetated sites (e.g. D and RV) have the highest levels of P_{DL}, with mean amounts of 10.68 ± 0.13 mg kg^{-1} and 16.02 ± 0.16 mg kg^{-1} respectively; whereas within bare profiles the mean values of the P_{DL} content were much lower, ranging from 4.09-5.00 mg kg^{-1}. Additionally, the higher P_{DL} content observed at Site RV displayed a different vertical distribution in comparison to other plots, clearly decreasing with depth. Moreover, a significant negative correlation of P_{DL} content with soil pH, C_{org} and bulk density was found at all sites. The contests of plant-available potassium (K) showed a higher scatter among all area units compared to P_{DL} ranging from 40.1 mg kg^{-1} to 1682.6 mg kg^{-1} of K_{DL} with a mean of

280.4 ± 1.8 mg kg^{-1} and a median of 152.6 ± 1.9 mg kg^{-1}, whereas the bare extraction plot E and the vegetated plot RV differed from other areas in having the highest K_{DL} contents.

Figure 24 shows a significant trend with depth in the contents of nitrate, nitrite and ammonium. The lower soil horizons have the highest NH_4^+ concentration, whereas the upper horizons have the highest NO_3^- concentrations, with the exception of Site PD where the highest concentrations of these chemical compounds are found in the middle horizon (Figure 24). The NO_3^- content was similar at the two rewetted sites, with higher N-NO_3^- amount in the topsoil of 2.2 and 1.9 $\mu g/g_{soil\text{-}dry}$ for RV and R respectively. The highest nitrate content of 12.5 and 2.6 µg N-Nitrate/$g_{soil\text{-}dry}$ was found at sites PD and E, respectively. In contrast, the content of N-NH_4^+ was the highest in non-vegetated areas, where maximum contents of up to 21 and 24 µg/ $g_{soil\text{-}dry}$ were found respectively for sites E and R. No detectable NO_2^- was found in any of the soil profiles.

The analyzed microbial biomass (C_{mic}; N_{mic}) showed a clear trends with depth as well as between study sites. The maximum C_{mic} was detected in the lower horizons, while the maximum N_{mic} was found in the topsoil of rewetted and vegetated sites. The mean C_{mic} contents were 6.92 ± 0.50 mg/$g_{soil\text{-}dry}$ with a median of 5.2 ± 0.2 mg/$g_{soil\text{-}dry}$. The C_{mic} distribution in the lower horizons was similar for all study sites, the highest amount being found in RV, which is mostly affected by vegetation and water level fluctuations. As seen with C_{mic}, the highest N_{mic} content was also found in the second horizon of RV, totaling 1.3 ± 0.1 mg/$g_{soil\text{-}dry}$.

It was generally found that plots affected by drainage had lower soil pH, C/N and microbial C and N contents than the rewetted plots. In general, the soil samples of all study sites were dissimilar to each other, but differed slightly in pH, carbon content, bulk density and electrical conductivity showing no statistical significance ($p > 0.05$).

4.3 Gas fluxes

4.3.1 Residual analyses and flux estimates

From a total of 1,623 chamber measurements, more than 75% of all flux curves were used for residual and further statistical analyses. The linear or non-linear flux model, which best represented the final slope for each measurement, was estimated by the *Akaike* Information Criterion with small sample correction (AIC_c, Kutzbach et al, 2007). The results on calculated AIC_c showed differences between different GHG. The majority of CO_2 fluxes for example were found to be better represented by a linear rather than an exponential fit (70%), although an F-test of the residual variances indicated that the exponential regression had a lower residual variance than the linear method. For methane and nitrous oxide emissions, in the majority of cases the calculated AIC_c value was smaller for the linear regressions (88% and 91%, respectively), which again indicates that this regression provides a better fit than the exponential regression. However, an F-test of the residual variance revealed no significant differences between linear and non-linear regressions for N_2O and CH_4 fluxes ($p = 0.2$; $p = 0.5$).

4.3.2 Carbon dioxide (CO_2)

Temporal variation and seasonal cycle

A strong seasonality in respiratory CO_2 fluxes (R_{eco}) was observed at all sites over the 2.5 year measurement period (2014-2016), with significant differences between seasons and between measurement years (Kruskal-Wallis one-way ANOVA, $p<0.001$ for both). The highest annual mean CO_2 flux and strongest temporal variability were measured in 2014, while in 2015 the mean annual CO_2 flux was 31% lower across the study area. Winter and spring CO_2 fluxes in 2016 were 15% lower than measured over the same period in 2014, whereas they were almost twice as high as measured in spring 2015. In terms of the individual study sites, only the CO_2 fluxes at Site R were not found to be statistically different between the measurement years ($p=0.14$). Distribution of R_{eco} fluxes in summer 2014 were not statistically different to summer 2015 ($p=0.91$). A detailed overview of the calculated ecosystem respiration fluxes for each of the monitoring sites and collars is shown in Figure 25.

All sites were found to be strong sources of CO_2 during summer periods, while wintertime emissions (December-February) were much lower, being near to zero at all monitoring plots. During the winter period (January 2016, E2), also highest rates of the CO_2 uptake (probably by a soil microcosm) were detected. Thus, the winter fluxes ranged between -0.9 to 3 µmol m^{-2} s^{-1} with some respiratory CO_2 peaks in vegetated sites after ice thawing. The temporarily higher fluxes in January were also observed in beginning of 2014, although these were less consistent. Furthermore, the highest R_{eco} rates at all plots occurred in late summer and early autumn (often in August and October). At all microtopes except site R, emissions from respiration were higher in autumn than during spring for both, 2014 and 2015, years (Fig. 25-26). The summer CO_2 emissions were decreasing during dry periods, and conversely CO_2 emission peaks followed large rain events. Additionally, the vegetated and non-vegetated D plots showed very similar seasonal trends in respiratory CO_2 fluxes (Kruskal-Wallis one-way ANOVA, $p < 0.001$) indicating a dependency of R_{eco} rates following seasonal trends in temperature.

As indicated by the grey shaded areas (Figure 25), high R_{eco} fluxes were observed during the month after collar installation (February 2014) with respect to the next month at all study sites, indicating a possible disturbance effect. Analysis of the data found that a linear regression model provided a better fit to the calculated CO_2 fluxes in the majority of cases (70%) when compared to an exponential regression model.

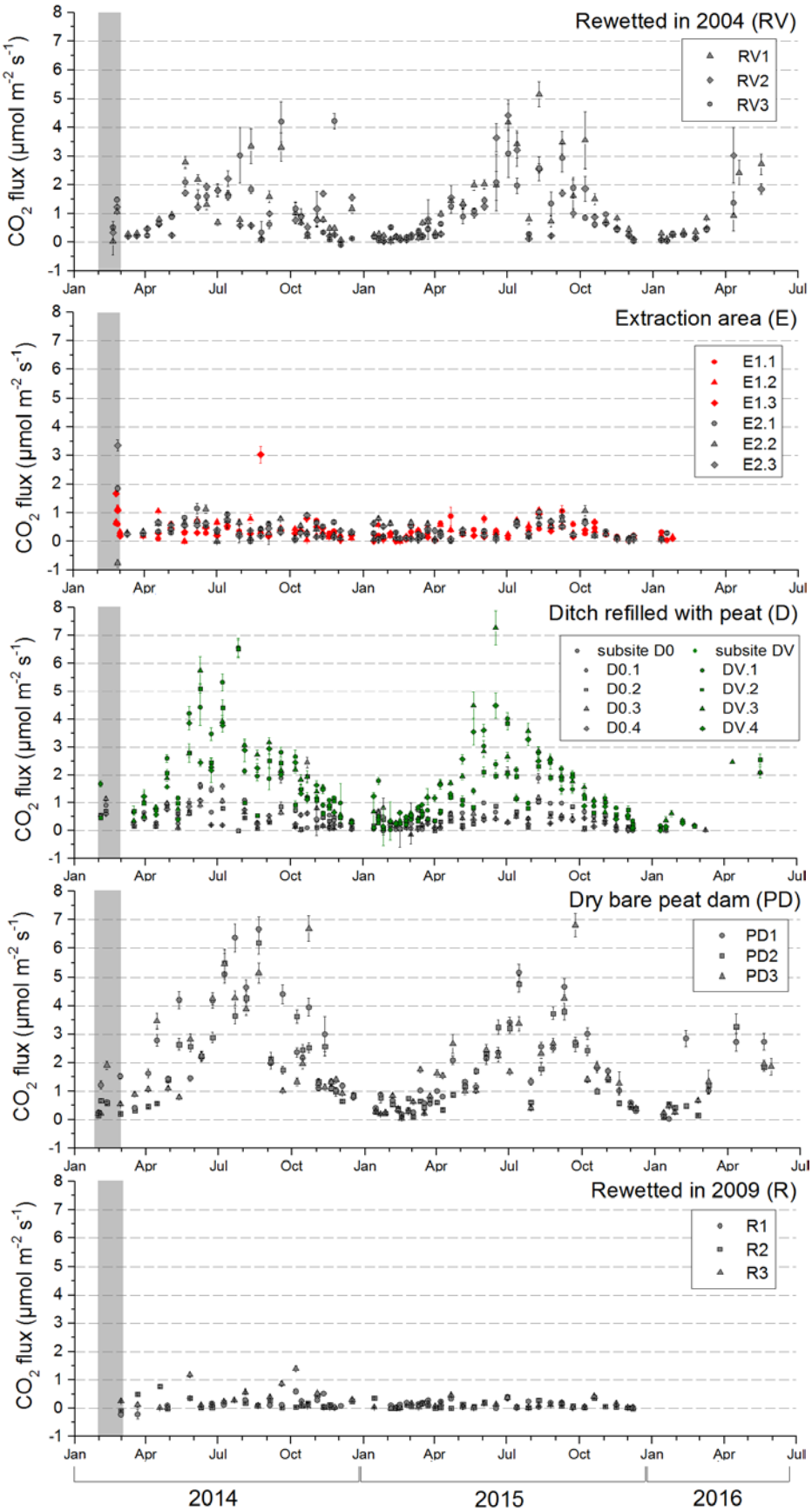


Figure 25. CO_2 fluxes (µmol m^{-2} s^{-1}) measured under dark conditions at all plots over the study period. Positive values indicate a loss of CO_2 to the atmosphere and negative values indicate ecosystem uptake. Sites without vegetation represent NEE, while vegetated sites represent only ER and cannot be directly compared to bare sites. Each replicate per plot is shown separately ± standard error. Shaded area indicates possible disturbance caused by collar installation.

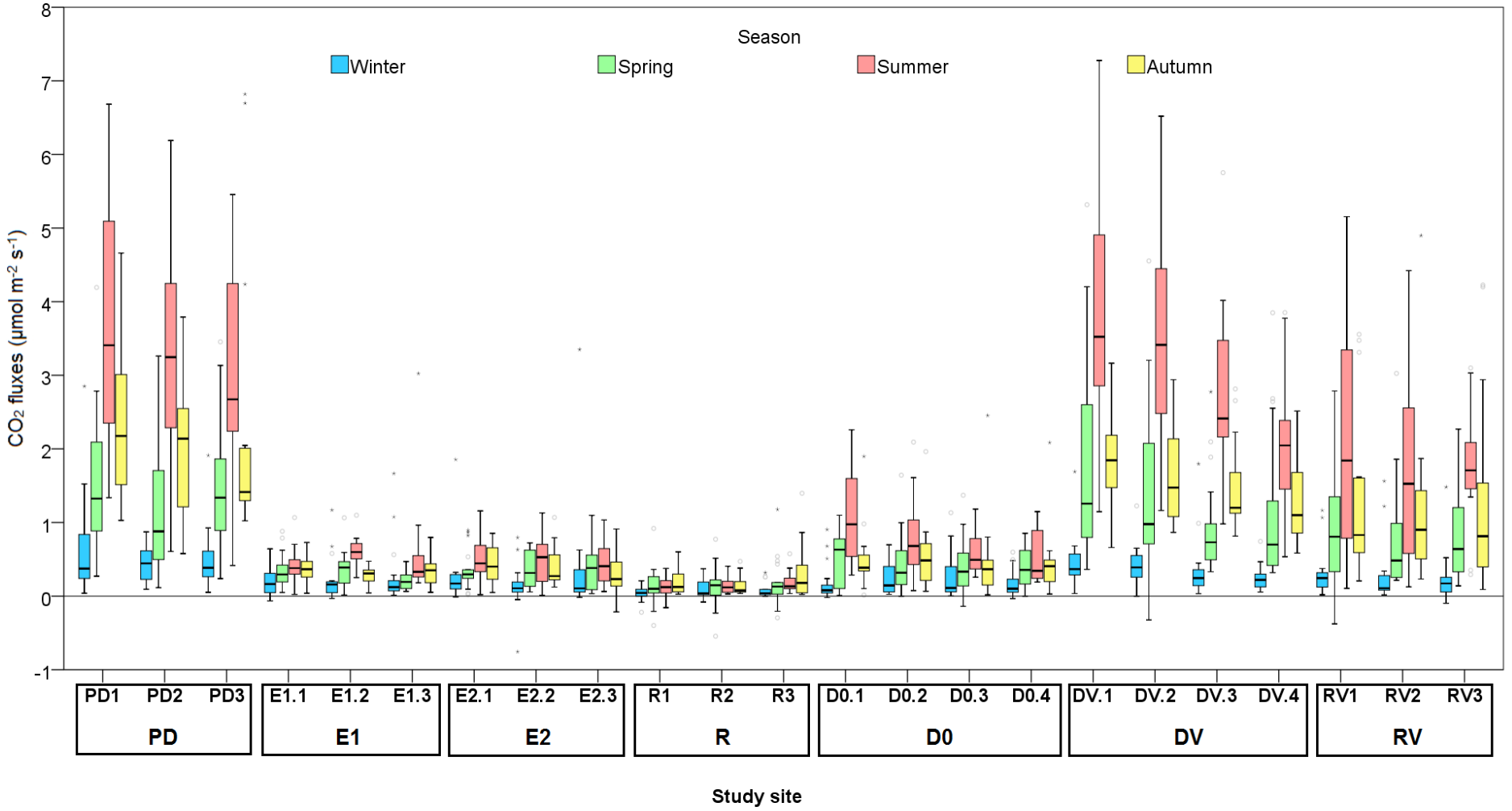


Figure 26. Boxplots of average CO_2 fluxes (µmol m^{-2} s^{-1}) for the four seasons shown for all replicates of study sites across the 2.5 year investigation. Boxplots represent temporal variability of CO_2 fluxes from all study sites (site abbreviation see in Table 1 and Figure 8). Sites without vegetation represent NEE, while vegetated sites DV and RV represent only ER part of NEE and cannot be directly compared to bare sites.

Spatial variability of R_{eco} fluxes between sites

Statistical analysis revealed significant differences in the measured respiratory CO_2 fluxes between monitoring sites (Kruskal-Wallis one-way ANOVA, $p < 0.001$). On annual scale, bare Himmelmoor study plots were identified to be sources of CO_2 emission, and the largest annual CO_2 emissions were found at Site PD, with the annual mean flux decreasing in magnitude as follows: PD > D0 > E2 > E1 > R.

The highest values of soil CO_2 uptake were measured at the on-going extraction area (E), where rates of up to -0.9 µmol m^{-2} s^{-1} were observed (subsite E2), followed by the subsites DV, R and RV (Figure 25). In contrast, no instances of CO_2 uptake by the soil were detected at any of the replicates at Site PD over the study period (Fig. 25-26).

The highest values of R_{eco} were detected at vegetated plots (Site DV, followed by site RV), where the maximum measured R_{eco} was 7.3 and 5.2 µmol m^{-2} s^{-1} respectively, with calculated annual average fluxes of 1.5 and 1.1 µmol m^{-2} s^{-1} for the two sites respectively.

The next strongest CO_2 source was the dry bare site PD (sites without vegetation represent NEE), while the lowest values of CO_2 release were observed at bare sites E and R. At Site PD, estimated daily CO_2 fluxes ranged between 0.03 to 6.9 µmol m^{-2} s^{-1} over the study period, with an average annual mean of 1.8 µmol CO_2 m^{-2} s^{-1}. In contrast to Site DV, annual R_{eco} fluxes at the non-vegetated area D0 were almost three times lower (with 0.47 µmol m^{-2} s^{-1} per year), however, clear CO_2 flux peaks of up to 2.1 µmol m^{-2} s^{-1} were observed during autumn 2014. At sites E and R, the largest CO_2 emissions were observed in late summer and early autumn 2014, with flux rates of 3.0 and 1.4 µmol m^{-2} s^{-1} respectively, with the majority of fluxes ranged around zero.

Influence of vegetation on R_{eco}

Measured respiratory CO_2 fluxes were compared to the vegetation type and cover at the two subsites DV and RV (see § 4.1.4). Respiration rates at both sites correlated strongly with the *Eriophorum* coverage (r^2=0.64) inside the collar (Figure 27).

At Site RV, *Eriophorum* cover increased from 1.2% to 9.3% over the study period, and the respiratory CO_2 fluxes showed a very strong dependence on the vegetation cover during each year as well as between years (Spearman's rank-order correlation, r_s= 0.77). At Site DV *Eriophorum* cover generally increased from collars DV4 to DV1 as well as during the second research year. Additionally, the correlation analysis found that temperature and vegetation cover were the two environmental variables which best explained variability in R_{eco}, indicating that the vascular plant coverage is one of the dominant controls on carbon dioxide (Spearman's rank-order correlation, r_s= 0.67).

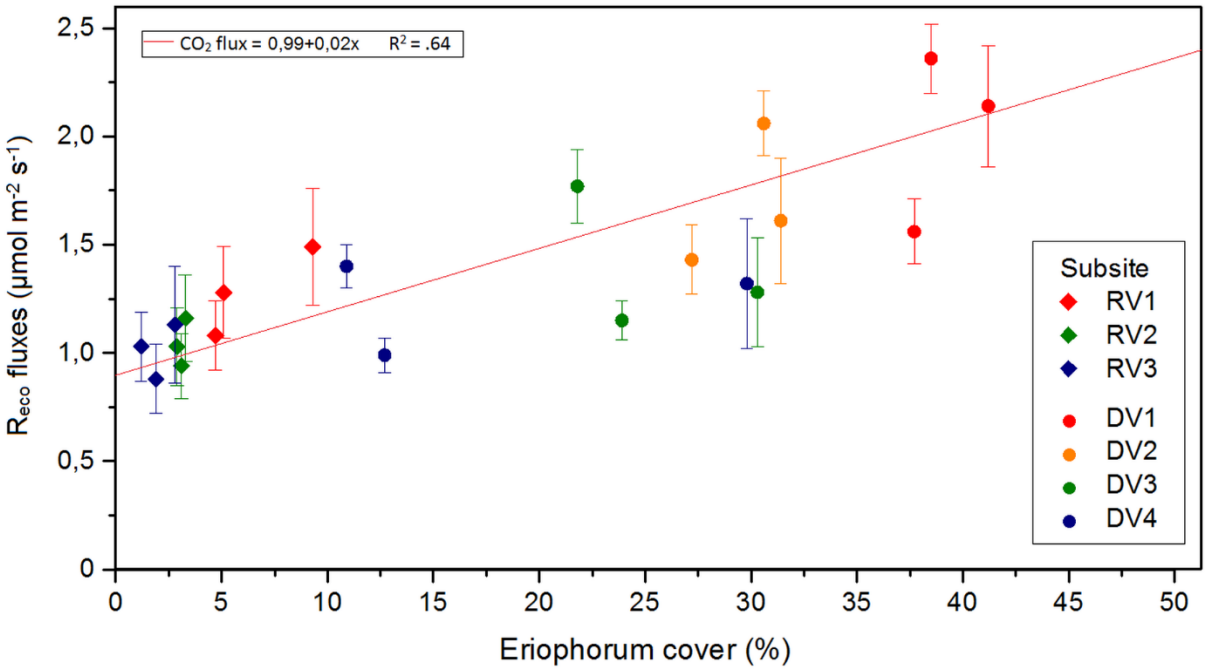


Figure 27. A positive relationship between mean annual R_{eco} flux and the vascular plants coverage at two vegetated plots RV (diamond shaped markers) and DV (round markers). Results from the three RV subplots (RV1, RV2, RV3) and the four DV subplots (DV1, DV2, DV3, DV4) are shown. The red line displays a polynomial fit (R^2=0.64).

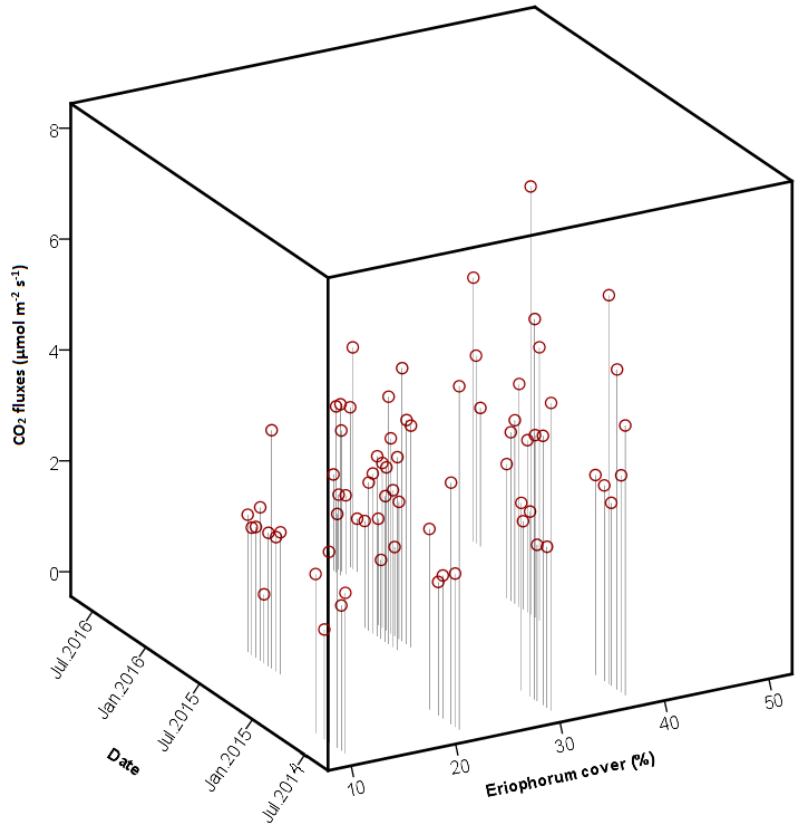


Figure 28. Relationship between summer R_{eco} fluxes and vascular plant coverage during monitoring at the four replicates from Site DV (Spearman's rank-order correlation, r_s= 0.33)

In addition to the vascular plant respiration at DV site, the soil respiration at non-vegetation D0 plot could be also affected by *Eriophorum* plants, first of all by detritus and fresh organic exudates supplied to the soil from plot DV, which is located 1 m near to D0. However, the clear CO_2 flux peaks observed at Site D0 could not be completely explained by a specific vegetation cover from DV due to insufficient study on changes in soil fauna, moisture content and organic matter availability.

4.3.3 Methane (CH4)

Temporal variation and seasonal cycle

CH_4 fluxes measured over the 2.5 year study period showed strong temporal variability with significant seasonal (Kruskal-Wallis one-way ANOVA, p=0.001) and inter-annual differences (p=0.001). The mean methane emissions observed in 2014 at all sites were significantly different from emissions in 2015 and 2016 ($p< 0.001$ and $p< 0.005$, respectively), but there was no significant difference in CH_4 fluxes between 2015 and 2016 for all sites ($p = 0.38$).

Comparing the different measurement years, 2015 exhibited the highest CH_4 emission rates, with an annual mean of 19.4 ± 4.5 nmol m^{-2} s^{-1}. In comparison, the mean annual methane flux across all sites in 2014 was around 15% lower, but they were clear differences between rewetted and non-rewetted sites. In general, the seasonal flux amplitudes for rewetted and vegetated plots (RV, DV, R) were more stable in 2014, exhibiting less spread than the drained sites (Figure 29).

In terms of seasonality, winter periods were characterized by the lowest methane fluxes and regular soil uptake, with flux rates ranging between -12.8 nmol m^{-2} s^{-1} (2014, site R) and 126.8 nmol m^{-2} s^{-1} (2014, subsite E2). Winter fluxes in 2014 were significantly larger from those in 2015 and were both significantly smaller than summer fluxes for all study sites ($p = 0.015$) with the exception of Subsite D0 and Site E. Additionally, the “wet” sites (DV, RV and R) showed no significant difference in the magnitude of summer and autumn methane fluxes ($p = 0.23$, $p = 0.19$, $p = 0.73$, respectively). Over the study period, the highest methane fluxes were measured in late summer and early autumn (mainly July and October), with a maximum flux of 353.3 nmol m^{-2} s^{-1} (June 2015) measured at Site RV. Summer methane emissions demonstrated a clear increase in wet periods, and emission peaks followed strong precipitation events. Additionally, vegetated plots DV and RV were found to exhibit a similar seasonal trend in methane fluxes ($p < .001$) indicating a dependency of emission rates following trends in vascular plants coverage.

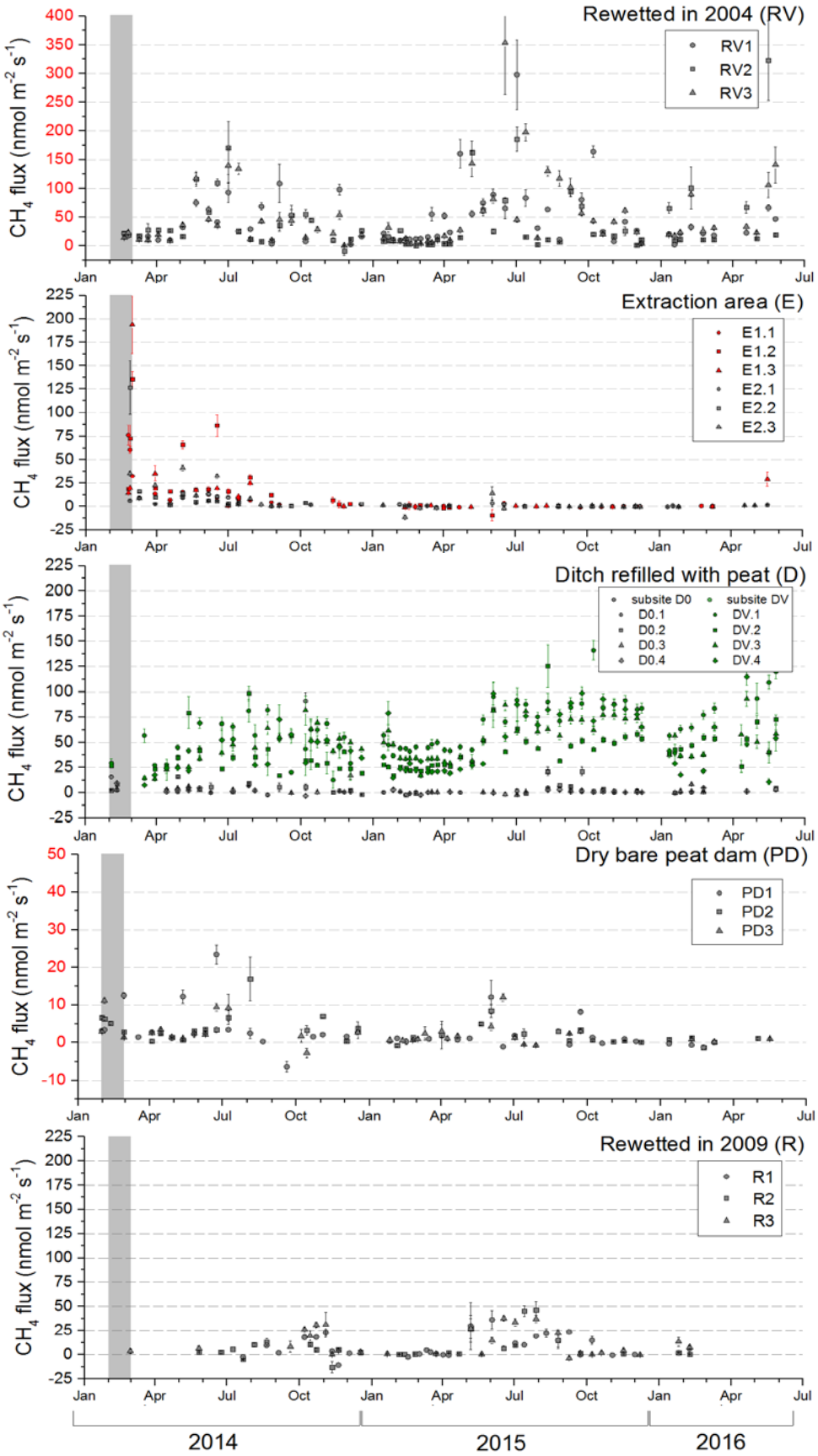


Figure 29. CH_4 fluxes (nmol m^{-2} s^{-1}) measured at all sites and replicates therein over the study period. Positive values indicate a loss of CH_4 to the atmosphere, and negative values indicate ecosystem uptake. It should be noted that plots RV and PD have deviating Y-Axis scales. The grey area indicates possible disturbance as a result of collar installation. Error bars indicate standard error of flux.

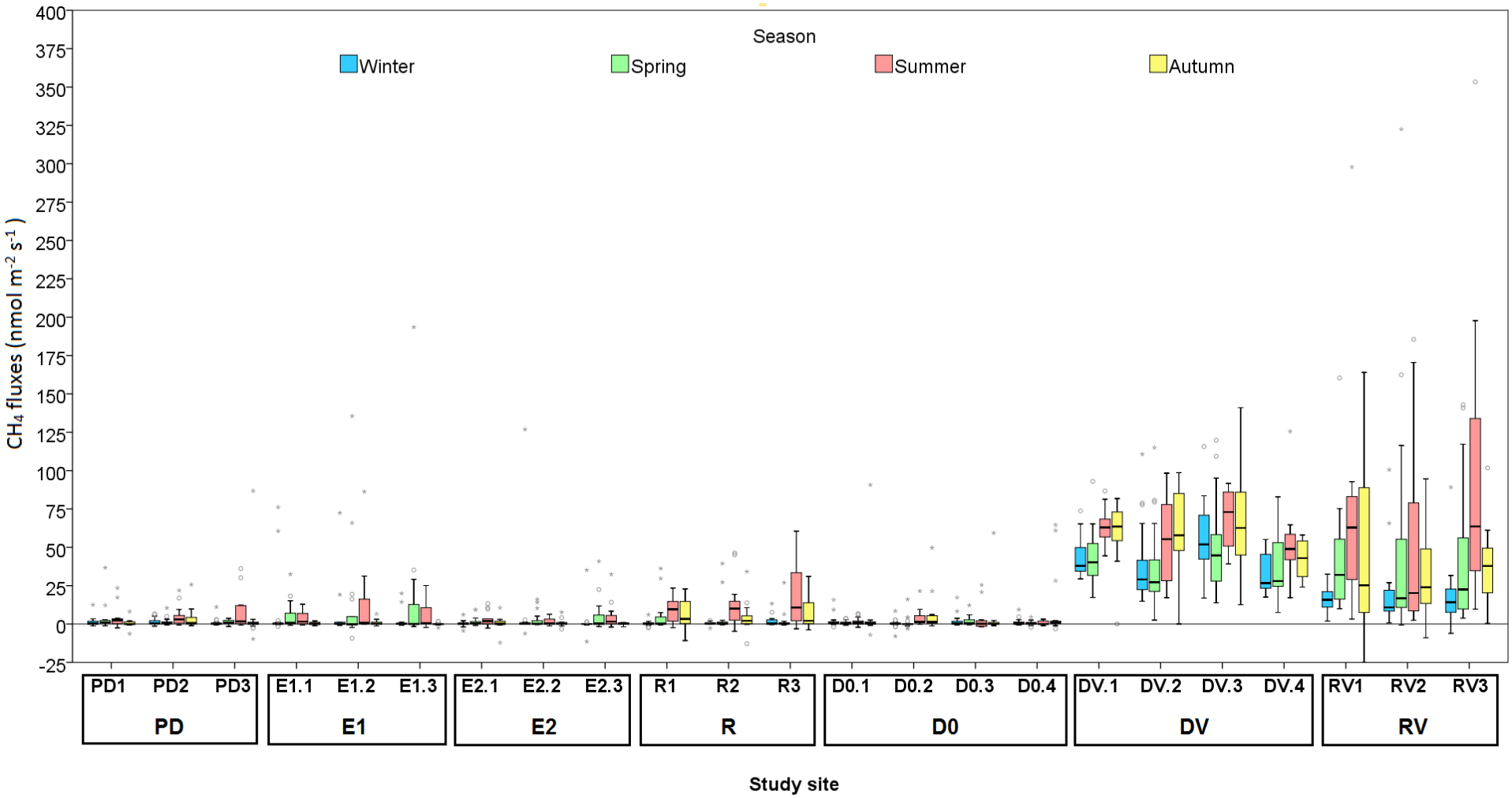


Figure 30. Boxplots of average CH_4 fluxes (nmol m^{-2} s^{-1}) for the four seasons shown for all replicates of study sites across the 2.5 year investigation. Boxplots represent temporal variability of CH_4 fluxes from all study sites (site abbreviation see in Table 1 and Figure 8). A strong seasonality in gas fluxes was found (Kruskal-Wallis one-way ANOVA, p =0.001).

Spatial variability of methane fluxes between subsites

CH_4 flux measurements reveal strong differences between study sites ($p < 0.001$). A more detailed overview of the calculated methane fluxes is provided in Figures 29-30. The largest differences were found between vegetated (RV and DV) and non-vegetated sites ($p < 0.001$), where fluxes at RV were found to be significantly smaller than fluxes at DV ($p = 0.012$). Overall, all Himmelmoor study plots were found to be sources of methane, with estimated mean annual CH_4 emissions from largest to smallest as follows: DV > RV > R > E1 > D0 > PD > E2. The highest values of CH_4 release were detected at Site DV, which had a mean annual flux of 50.3 nmol m^{-2} s^{-1}, followed by sites RV and R which had average fluxes of 42.7 nmol m^{-2} s^{-1} and 9.5 nmol m^{-2} s^{-1} per year respectively. The remaining study plots (E1, D0, PD, E2) exhibited generally low fluxes with some irregular and sporadic outbursts of methane. For example, although the estimated mean CH_4 emissions at Site PD were very low (2.7 nmol m^{-2} s^{-1}), clear methane peaks of up to 23.5 nmol m^{-2} s^{-1} were observed during the summer and autumn period. At sites E and R, the largest CH_4 emissions were detected during spring and summer 2014, in addition to the high peaks in flux following collar installation in February 2014, when irregular high fluxes of 193.5 and 60.5 nmol m^{-2} s^{-1}, respectively, were measured respectively at the two sites.

The highest values of soil CH_4 uptake were measured at the rewetted site (RV: -21.4 nmol m^{-2} s^{-1}), and the on-going extraction area (E2: -8.5 nmol m^{-2} s^{-1}), followed by the subsites R and PD (Figure 30). In contrast, Site DV was the only area in which no CH_4 uptake by the soil was detected in any of the replicates (Fig. 30), with fluxes ranging from 2.5 to 141.1 nmol m^{-2} s^{-1}.

Influence of vegetation on methane fluxes

Methane flux estimates for the two vegetated subsites (DV and RV) were compared to the vegetation type and cover at each plot as described previously (see §4.1.4). Analysis identified a strong correlation between the magnitude of the CH_4 flux and *Eriophorum* coverage inside each monitored collar, characterized during both vegetated plots, similarly as for ecosystem respiration fluxes (Figure 31).

At Site RV for example, methane fluxes showed a positive relationship with *Eriophorum* cover (Spearman's rank-order correlation, $r_s = 0.22$) (Figure 31). At site DV *Eriophorum* cover generally decreased from collars DV1 to DV4, but increased during the second research year, and the methane fluxes, similarly as R_{eco} fluxes, were thus related to the vegetation cover and number of plants inside the collar (Spearman's rank-order correlation, $r_s = 0.50$).

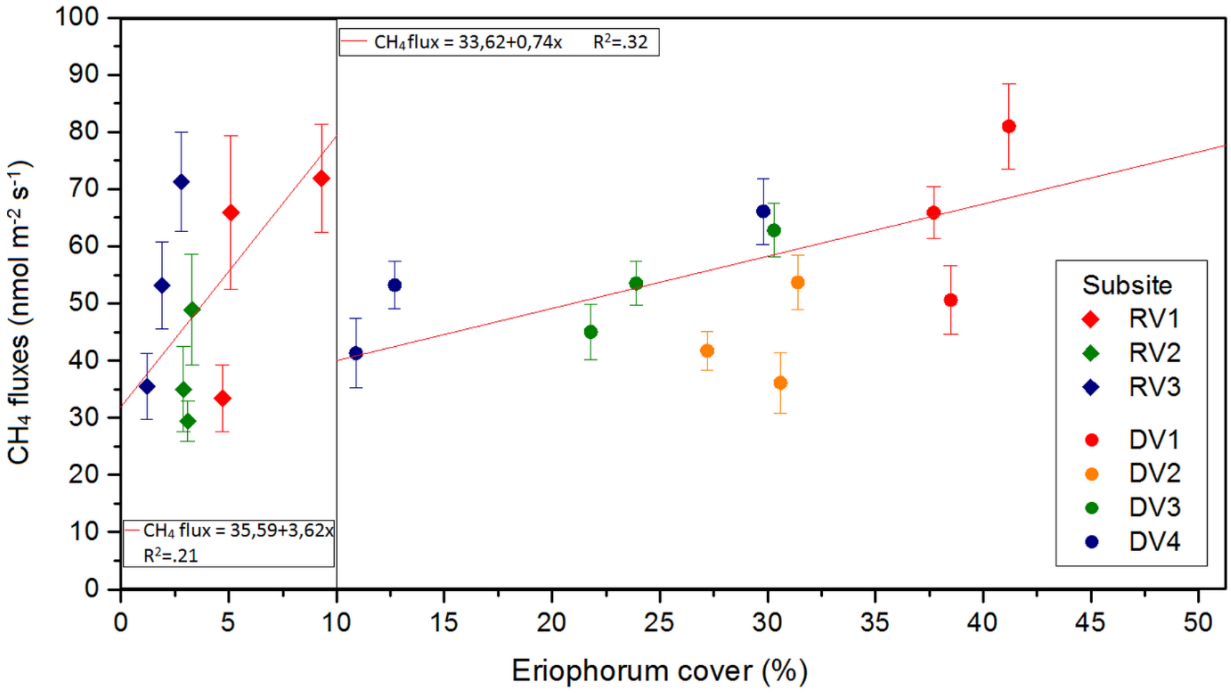


Figure 31. A positive relationship between mean annual methane flux and vascular plant coverage at the two vegetated plots. The results from three replicates from Site RV (RV1, RV2, RV3) and four replicates from Site DV (DV1, DV2, DV3, DV4) are shown. Red lines display linear regression functions (R^2= 0.21; R^2= 0.32). A detailed overview on plant number and vegetation coverage is given in §4.1.4.

4.3.4 Nitrous oxide (N_2O)

Seasonal and inter-year variability

Over the 2.5-years measurement period (Figure 32), the measured N_2O fluxes displayed strong seasonal and inter-annual variability (Kruskal-Wallis one-way ANOVA, $p < 0.001$, $p < 0.001$, respectively). The highest nitrous oxide emissions and the largest variance were seen in 2015, where rates of up to 13 nmol m^{-2} s^{-1} were observed. The highest mean annual flux however was measured in 2014, amounting to 0.8 nmol m^{-2}, which is 37% larger than the 2015 average. Fluxes measured in the winter and spring seasons of 2016 were also comparatively lower than in previous years, but being still similar those observed over the same period in 2015. Analysis shows N_2O fluxes in summer to be similar between measurement years, whereas significant differences were observed in winter ($p=0.004$). This observation was made for all individual sites except sites R and RV, where no significant seasonal difference in winter mean flux were found ($p=0.17$, $p=0.18$).

Figure 32 shows the seasonal variability of N_2O fluxes from 2014-2016. As can be seen in the figure, all monitoring sites were identified as significant sources of N_2O during the late summer and autumn, with mean emissions of 0.92 ± 1.3 nmol m^{-2} s^{-1}. Moreover, the summer N_2O emissions showed a clear increasing trend during the dry periods, and also emission peaks followed after large rain events and ice thawing, similarly as for the ecosystem respiration fluxes. In contrast, winter fluxes were generally lower, mean fluxes here were 0.39 ± 1.2 nmol m^{-2} s^{-1}; however, in contrast to all other sites, the two bare flooded areas (R, D0, DV) showed higher N_2O fluxes during winter 2014/15, but not during winter 2015/16. Statistical analysis revealed no significant difference between summer and autumn flux rates, although it did highlight the strong variability between summer, winter and spring emissions ($p < 0.001$).

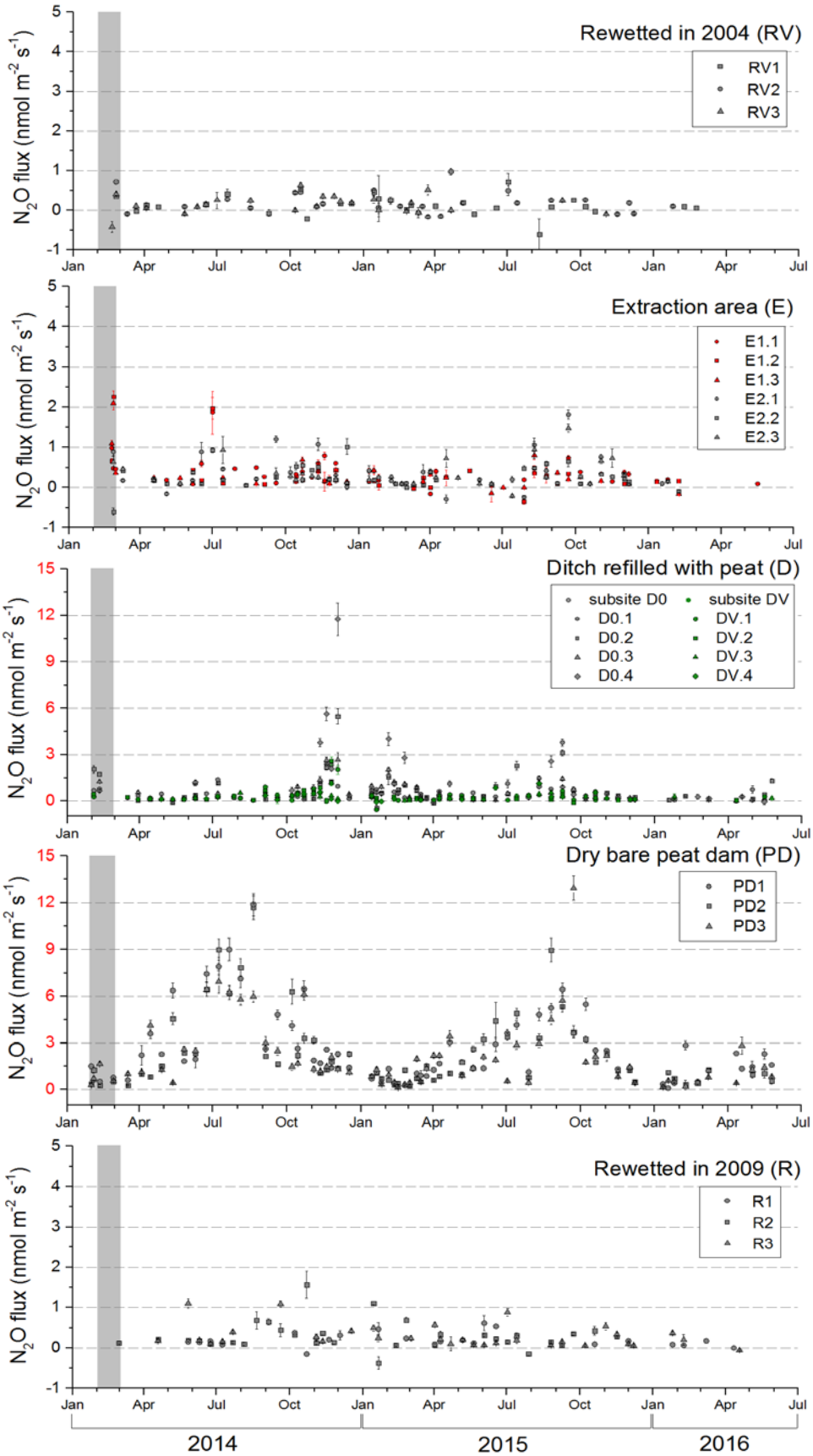


Figure 32. N_2O fluxes (nmol m^{-2} s^{-1}) measured at all study plots over the study period. Positive values indicate a loss of N_2O to the atmosphere and negative values indicate ecosystem uptake. Each replicate plot is shown separately. Note that plots D and PD have broader Y-Axis limits. Grey area shows possible effect of disturbance by inserting the collars. Error bars indicate standard error of flux.

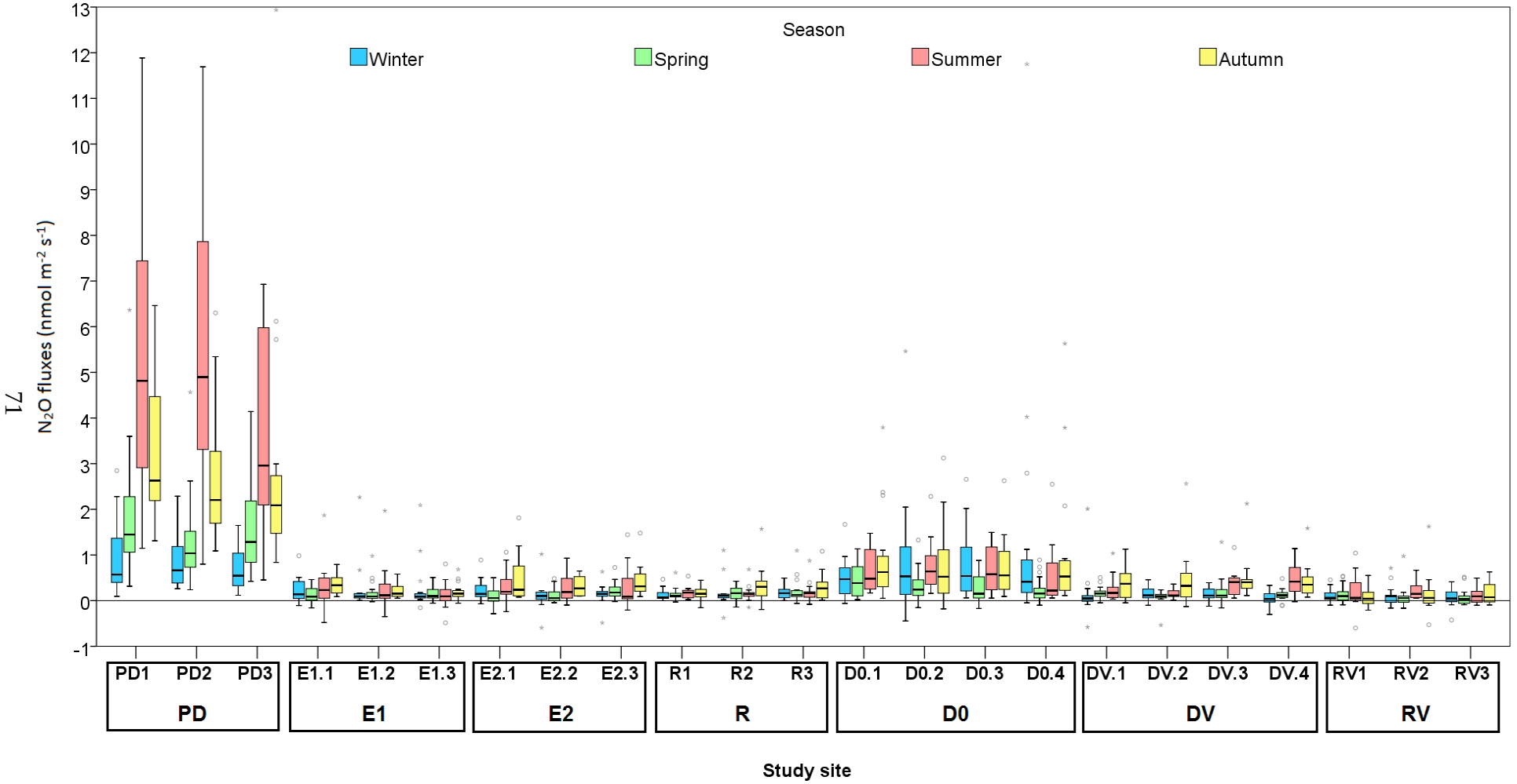


Figure 33. Boxplots of average seasonal N_2O fluxes (nmol m^{-2} s^{-1}) for the four seasons shown for all replicates of study sites across the 2.5 year investigation. Boxplots represent temporal variability of N_2O fluxes from all study sites (site abbreviation see in Table 1 and Figure 8). A strong seasonality in gas fluxes was found (Kruskal-Wallis one-way ANOVA, p =.001).

Spatial variability of N_2O fluxes between sites

The measured N_2O fluxes exhibited strong spatial variability, as shown in Figure 32 ($p< 0.001$). Site PD for example was associated with the highest mean annual flux rate of 2.3 nmol m^{-2} s^{-1}, which is an order of magnitude greater than the emissions observed from the rewetted plots. Particularly high fluxes (12.9 nmol m^{-2} s^{-1}) were also measured at plot D0, though this occurred in December 2014 due to irregular nitrous oxide peaks, and hence the average flux for this site is relatively low at 0.6 nmol m^{-2} s^{-1}.

Sites characterized by bare peat (PD and D0) were found to display clearly larger fluxes than other non-vegetated and vegetated sites ($p<0.001$), with fluxes at PD being statistically larger than at D0 site ($p<0.001$). Significant differences were also seen between the two vegetated plots, with significantly lower nitrous oxide emission rates found at site RV than at the other vegetated plot (DV). On the whole, all Himmelmoor study plots behaved as sources of N_2O emissions on the annual scale, with measured mean annual emissions at each site from largest to smallest as follows: PD > D0 > E2 > DV > E1 > R > RV.

Similarly to the PD plot, other study plots (E, RV, R) were generally characterized as sources of N_2O; however, mostly low or non-significant N_2O emissions were found on this sites (Figure 32, 33). Additionally, no significant difference was found between plots E1 and E2 ($p= 0.68$). However, two replicates (E1.2, E1.3) showed higher N_2O emissions of up to 2 nmol m^{-2} s^{-1} on July 2014, and, twice, significant negative fluxes were measured at plot E2 in February 2014 and 2015. Other pairwise comparisons of the study plots revealed no significant differences ($p>0.5$). In contrast, a clear N_2O uptake (on average of -0.1 nmol m^{-2} s^{-1}) was observed at the majority of plots (except Site PD) over the study period, predominantly at vegetated sites.

Influence of vegetation cover on nitrous oxide fluxes

In contrast to the strong relationship found between ecosystem respiration as well as methane flux and vegetation cover, the measured N_2O fluxes display weak to no significant correlation with the *Eriophorum* coverage inside each collar, as shown in Figure 34, however, a low correlation was found on the RV plot (Spearman's rank-order correlation, $r_s= .50$).

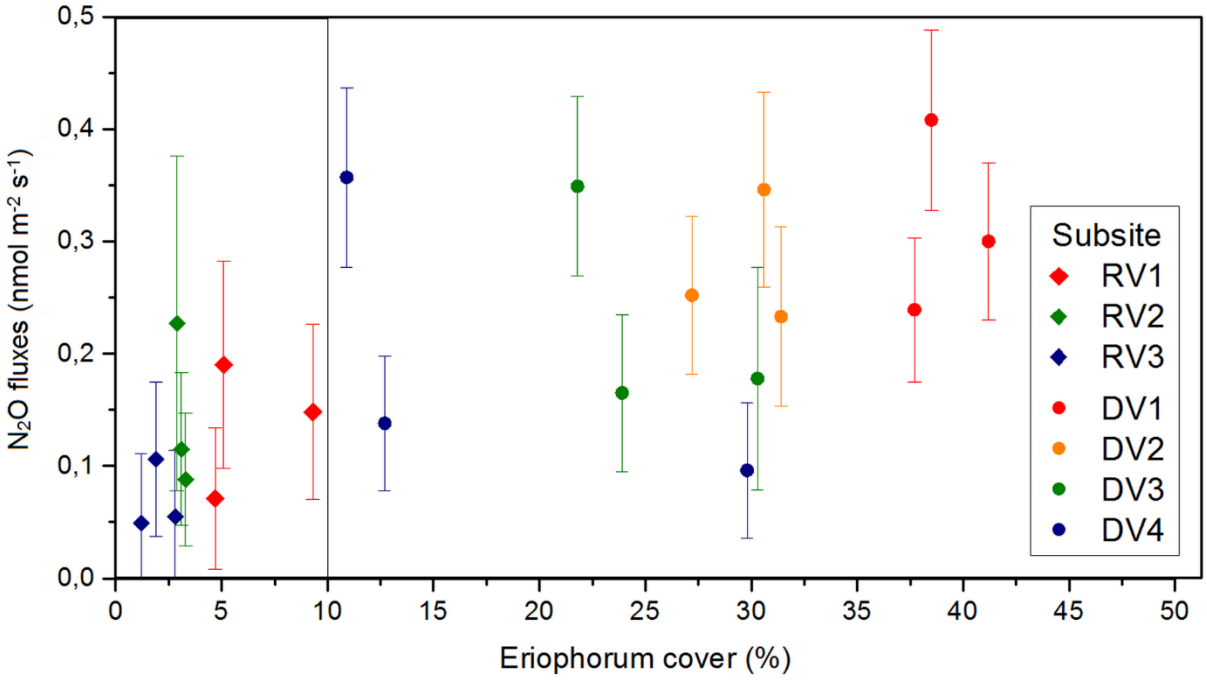


Figure 34. Vascular plant coverage versus mean annual nitrous oxide fluxes at the two vegetated plots, RV and DV, with three (RV1, RV2, RV3) and four (DV1, DV2, DV3, DV4) replicates respectively. A more detailed overview on plant number and vegetation coverage is given in §4.1.4.

4.3.5 Effect of soil and climatic conditions on GHG fluxes

The relationships between the GHG emissions and the environmental conditions for all study sites were determined using a Pearson and Spearman's rank-order correlations (r; r_s). The correlations with soil and air temperature, precipitation, soil redox potential, wind speed and water table level were determined for all study plots.

Impact of air and soil temperature

Fluxes of R_{eco}, CH_4 and N_2O were found to display a clear relationship with soil and air temperature, with the strongest positive correlations (r_s=0.75) being observed at sites PD, RV and D (Figure 35). At some study sites (e.g. sites R, E), due to low magnitude of methane fluxes, no significant correlation with soil temperature was found; however, at plot R, a slight dependency of the CH_4 fluxes was detected (r_s=0.4). The strongest correlations between soil temperature at all study plots and GHG fluxes were observed for temperatures at 10 and 20 cm depth.

Precipitation and water table level

Heavy rainfall events were found to have some impact on the magnitude of R_{eco} and N_2O fluxes at all study plots ($p < 0.001$), with larger emissions following periods of heavy precipitation. The majority of sites also exhibited a strong correlation between CH_4 flux and precipitation ($p < 0.001$), although this was not true for the two extraction area subsites (E1 and E2), where no significant correlation was observed ($p = 0.24$).

In contrast to the above, a strong negative relationship between GHG fluxes and water table level was observed at almost all sites (Figure 36). In general, this relationship was strongest for R_{eco}, followed by N_2O and lowest for CH_4. Some discrepancies are found amongst the data, with for example WTL not significantly correlating with the N_2O flux at sites RV and E2; or with the measured CH_4 fluxes at Site PD. Contrary, WTL was determined as important control factor on R_{eco} and N_2O emissions from plots PD ($r = -0.5$) and both subsites of the plot D ($r = -0.49$). On the E2 plot, only correlation for R_{eco} fluxes with WTL was evident ($p < 0.01$).).

Soil redox potential

In general, soil redox potential displayed a weak relationship with GHG fluxes for the whole study area, although at some plots a significant positive relationship ($p = 0.03$) was identified between redox potential at 10 cm depth and R_{eco} and N_2O fluxes, whereas a negative relationship was found for methane fluxes. On the dry PD plot, ecosystem respiration and nitrous oxide fluxes displayed a significant correlation to soil redox potential at all depths, with the largest coefficient observed at 5 cm depth ($p < 0.001$). No significant correlation between redox potential and methane was identified, and also at plot D no significant correlations between ecosystem respiration fluxes and redox potential at any depths on either vegetated or non-vegetated subsites were found.

At Site R, no significant relationship between the measured nitrous oxide fluxes and soil redox potential was found, while R_{eco} and methane flux were found to be significantly related to redox potential at 5 and 10 cm depth. For the other rewetted plot (RV), only CO_2 fluxes were found to correlate with daily means of redox potential at 20 cm depth ($p = 0.03$). Meanwhile, the daily average redox potential at 5 cm depth was the abiotic variable which best explained variations in N_2O and R_{eco} emissions at the E2 plot. In contrast, no significant relationship between any of the GHG fluxes and soil redox potential was found at plot E1 (Figure 37), mainly due to a low number of redox potential measurements at plots RV and E used for correlation ($n = 42$).

Wind speed

For the majority of plots, analysis found no strong relationship between wind speed and gas flux. Sites PD and D were the only sites where a significant correlation could be identified, namely between wind speed and both R_{eco} and N_2O fluxes.

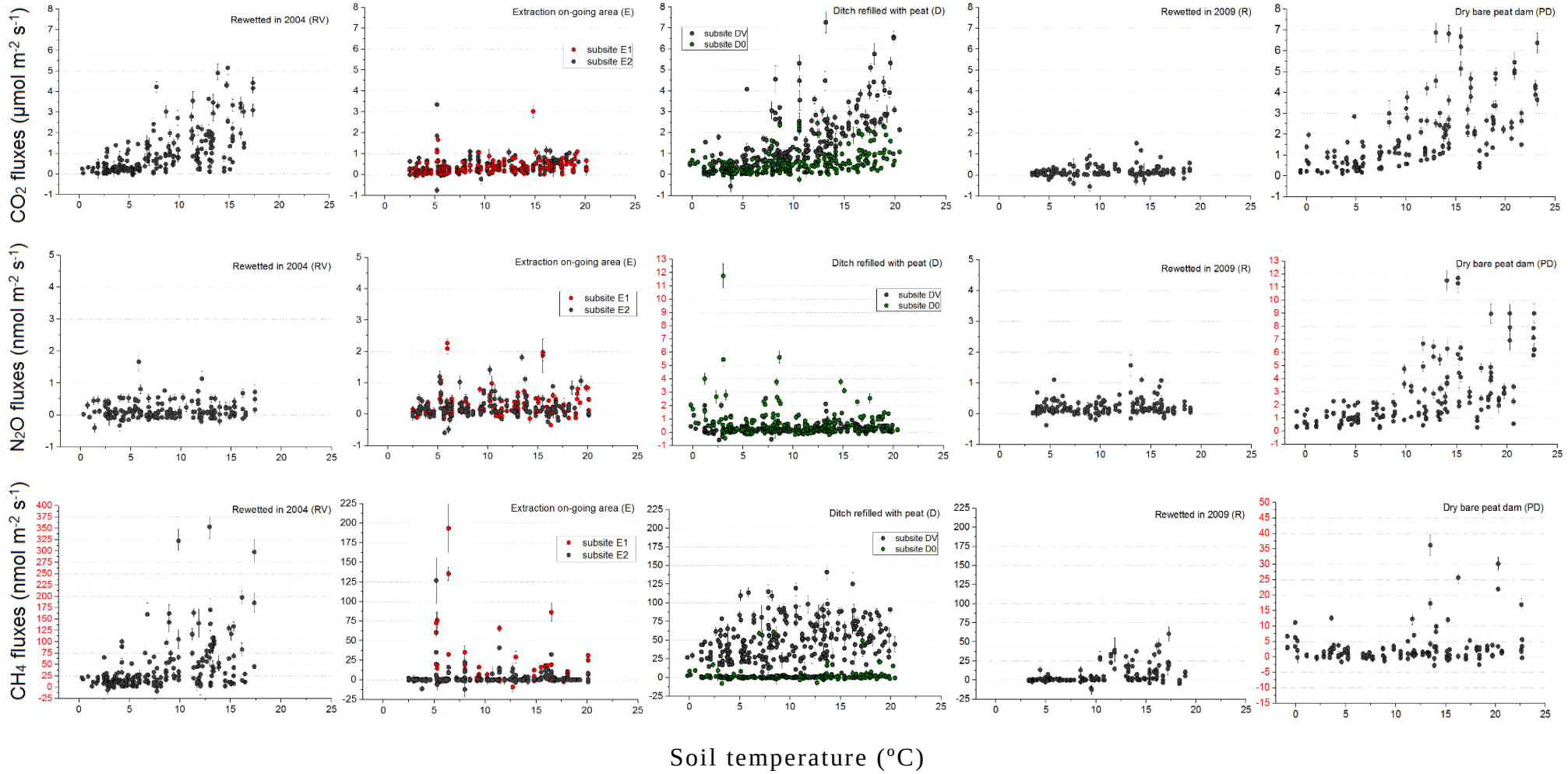


Figure 35. R_{eco} (upper panel), nitrous oxide (mid panel) and methane (lower panel) fluxes at all study sites plotted against of soil temperature at 10 cm depth. Data of all replicates are shown separately with their standard deviation. Note: broader Y-axis limits are highlighted in red.

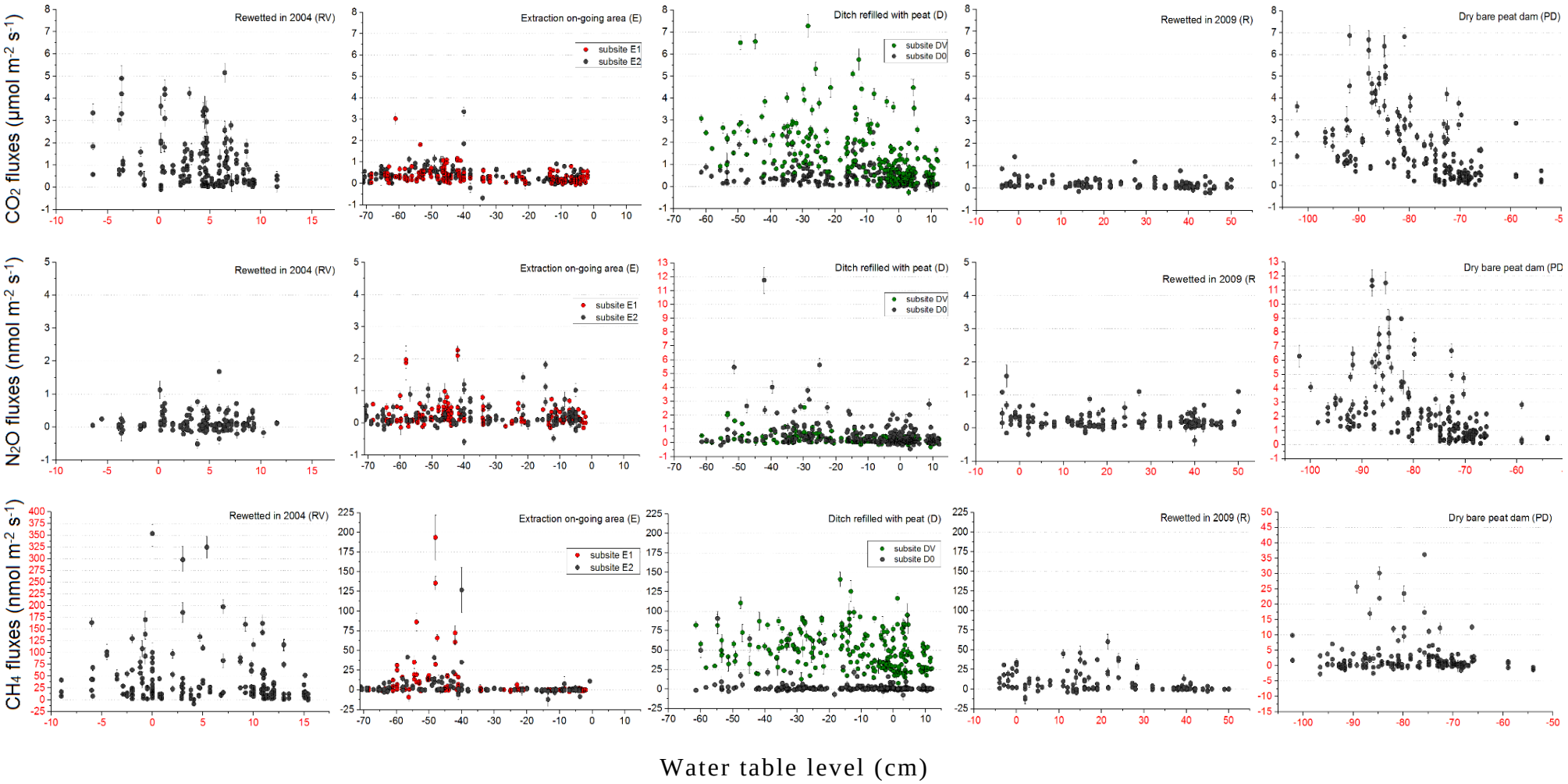


Figure 36. R_{eco} (upper panel), nitrous oxide (mid panel) and methane (lower panel) fluxes at all study sites plotted against water table level. Data points from all replicates are shown alongside their standard deviation. Note: broader Y- or X-limits are highlighted in red.

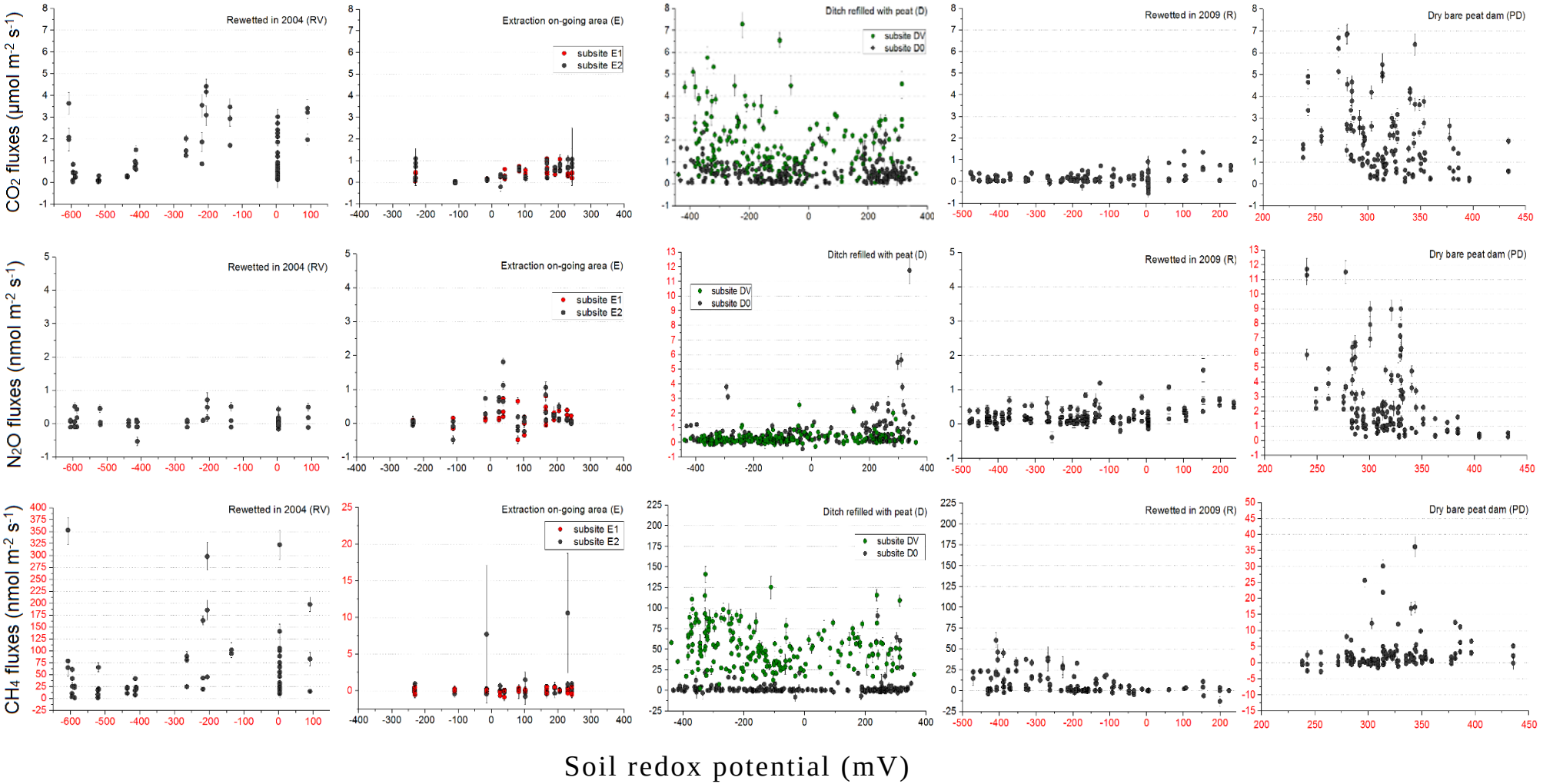


Figure 37. R_{eco} (upper panel), nitrous oxide (mid panel) and methane (lower panel) fluxes at all study sites plotted against soil redox potential (mV) at 10 cm depth. Data from replicates are shown alongside their standard deviation. Note: broader Y- or X-axis limits are highlighted in red. For plots E and RV fewer redox measurements were made than for other plots (n=42).

4.3.6 Correlation of the N_2O and CO_2 fluxes

Overall, a significant correlation between CO_2 and N_2O emissions was identified for all sites except the subsite D0 ($p = 0.001$). The strongest correlation between these fluxes was observed at the dry peat dam site (PD). Furthermore, the observed degree of correlation for this study was very similar to previous studies of van Asperen (2015). The CO_2:N_2O emission ratio here varied yearly, as is shown in Figure 38. In 2012 the ratio was higher with 1:2.8 than in 2014 (1:1.8) and in 2015 (1:1.6). In contrast, at the rewetted site (R), the CO_2:N_2O ratio remained constant over the study period at 1:8.

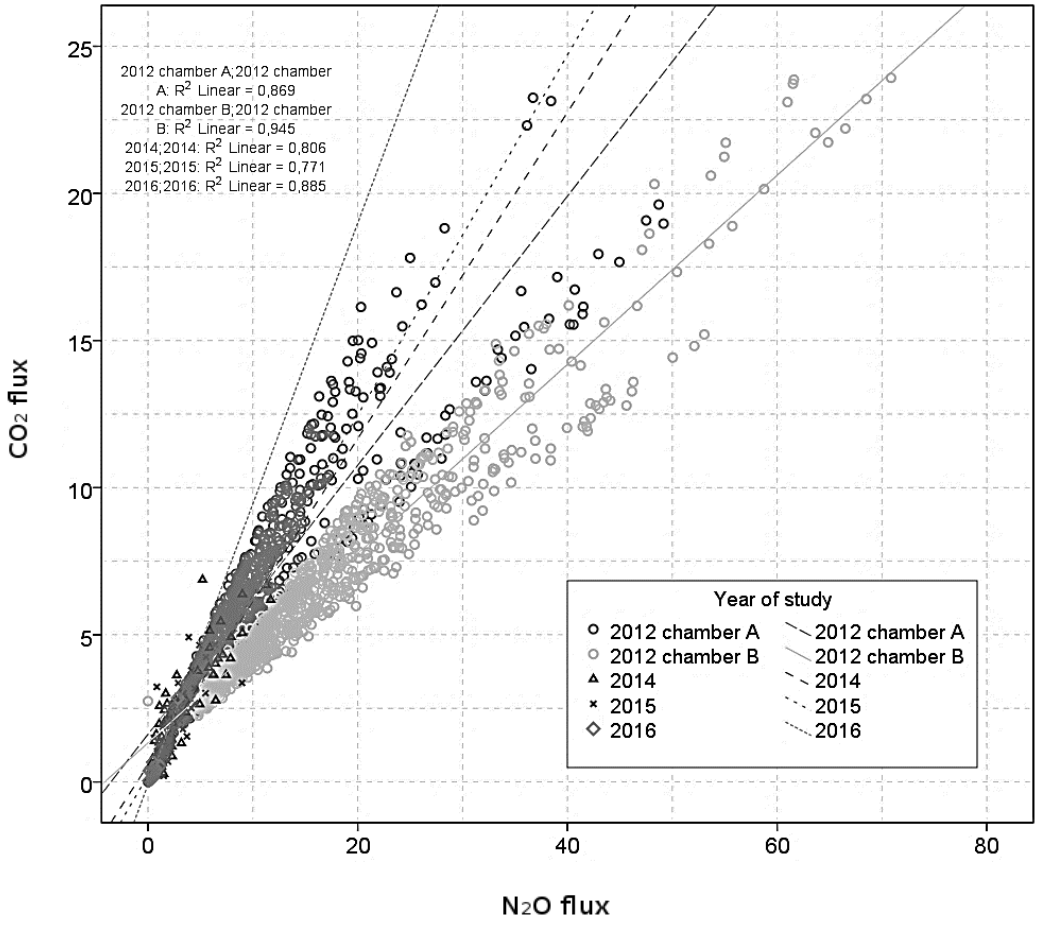


Figure 38. The relationship between CO_2 and N_2O fluxes using closed metal chambers (triangles, crosses and diamonds; this research) and a FTIR-flux chamber set up (open circles; data for chamber A and B from Hella van Asperen, 2015) for Site PD for years 2012, 2014, 2015 and 2016.

Correlation between R_{eco} and N_2O displayed a similar clear trend during measurement period for other study sites and indicated gas sources that respond similar to environmental variables on the PD plot such as soil temperature, WTL, soil redox potential etc. However, for CO_2 also sources and sinks higher in the canopy are expected, and hence not a strong correlation between these gases can be assumed.

Analysis found the correlation between R_{eco} and CH_4 fluxes was significant only at the rewetted and vegetated plots. This is likely due to the irregular emissions and low CH_4 fluxes observed at the bare plots (D0, E and PD), whereas the strongest relationship between CH_4 and CO_2 fluxes was identified for the vegetated plots RV and DV ($p < 0.001$).

4.3.7 Annual GHG emissions

The annual GHG fluxes were calculated by multiplying the daily average flux by the length of each season (in days) to derive a seasonal gas budget. By estimating the average daily fluxes from three/four replicates, it was assumed that nighttime methane and nitrous oxide fluxes did not differ from the daytime fluxes, as has been described in previous studies (Vanselow-Algan, 2014; van Asperen, 2015). In contrast, for R_{eco} it was found that daytime fluxes at one of the vegetated plots (RV) were up to three times higher than their nighttime counterparts (Vanselow-Algan, 2014), and hence it is possible that annual R_{eco} fluxes from the vegetated plots are overestimated.

Additional sources of uncertainty in the annual flux estimates which should be considered are: (1) the non-continuous sampling conducted in the first half of 2014 (January-May); and (2) the unusually large fluxes measured in February 2014, which were presumably caused by soil disturbance after collar installation, and likely resulted in an overestimation of the winter fluxes in 2014. As plots D0 and DV have different covering area over D site (20% and 35%, respectively), their total annual GHG emissions were calculated only for the subsite's coverage of 121,610 m^2 from the whole amount of 221,100 m^2.

Table 11. Mean GHG emissions for the years 2014 and 2015 for all Himmelmoor study sites converted into CO_2-equivalents according the Global Warming Potentials and Global Temperature Potentials (GWP, GTP for a 100-year period). Values given are annual means calculated from three to four replicates ± standard deviation.

Subsite	Year	GHG Potential used for calculation	R_{eco}	CH_4	N_2O	**Sum**	Area
			[t CO_2-eq ha^{-1} year^{-1}]				[m^2]
R: Area rewetted in 2009							
R	2014	GWP	2.86±1.12	0.95±0.68	1.25±0.48	5.07±2.29	
		GTP		0.31±0.22	1.25±0.48	4.42±1.83	
	2015	GWP	1.85±0.50	1.50±0.53	1.06±0.32	4.41±1.35	
		GTP		0.49±0.17	1.06±0.32	3.39±0.99	
Total mean annual emission per whole area:						***38.45±14.79***	*81,300*
PD: Dry peat dam							
PD	2014	GWP	30.74±4.69	0.59±0.23	12.95±2.18	44.28±7.10	
		GTP		0.19±0.07	12.91±2.17	43.84±6.94	
	2015	GWP	22.46±4.24	0.35±0.15	9.12±2.19	31.94±6.58	
		GTP		0.11±0.05	9.09±2.18	31.67±6.47	
Total mean annual emission per whole area:						***552.07±98.53***	*144,900*
D: Ditch refilled with peat							
DV	2014	GWP	24.41±3.81	7.07±0.65	1.33±0.44	32.81±4.91	*77,385*
		GTP		2.29±0.21	1.33±0.44	28.02±4.47	
	2015	GWP	18.72±2.87	9.27±0.65	0.86±0.20	28.86±3.73	
		GTP		3.00±0.21	0.86±0.20	22.58±3.29	
D0	2014	GWP	7.73±1.69	0.95±0.60	4.28±2.47	12.95±4.76	*44,225*
		GTP		0.31±0.19	4.26±2.47	12.30±4.35	
	2015	GWP	5.45±1.07	0.40±0.25	2.85±0.78	8.70±2.10	
		GTP		0.13±0.08	2.84±0.78	8.42±1.92	
Total mean annual emission per whole area:						***286.46±48.30***	*121,610*
E: On-going extraction area							
E1	2014	GWP	5.98±1.69	1.70±0.73	1.36±0.52	9.04±2.94	
		GTP		0.55±0.24	1.36±0.52	7.89±2.45	
	2015	GWP	4.67±0.88	0.06±0.72	0.87±0.27	5.59±1.87	
		GTP		0.02±0.23	0.87±0.27	5.55±1.39	
E2	2014	GWP	6.09±1.58	1.07±0.59	1.40±0.38	8.57±2.55	
		GTP		0.35±0.19	1.40±0.38	7.84±2.15	
	2015	GWP	4.72±1.05	0.11±0.15	1.29±0.42	6.12±1.62	
		GTP		0.04±0.04	1.29±0.42	6.04±1.52	
Total mean annual emission per whole area:						***225.62±67.72***	*307,800*
RV: Area rewetted in 2004							
RV	2014	GWP	13.98±2.86	5.96±1.86	0.70±0.20	20.65±4.92	
		GTP		1.93±0.60	0.70±0.20	16.61±3.67	
	2015	GWP	15.86±2.69	8.34±2.85	0.57±0.26	24.78±5.81	
		GTP		2.70±0.92	0.57±0.26	19.13±3.88	
Total mean annual emission per whole area:						***192.02±45.32***	*84,550*

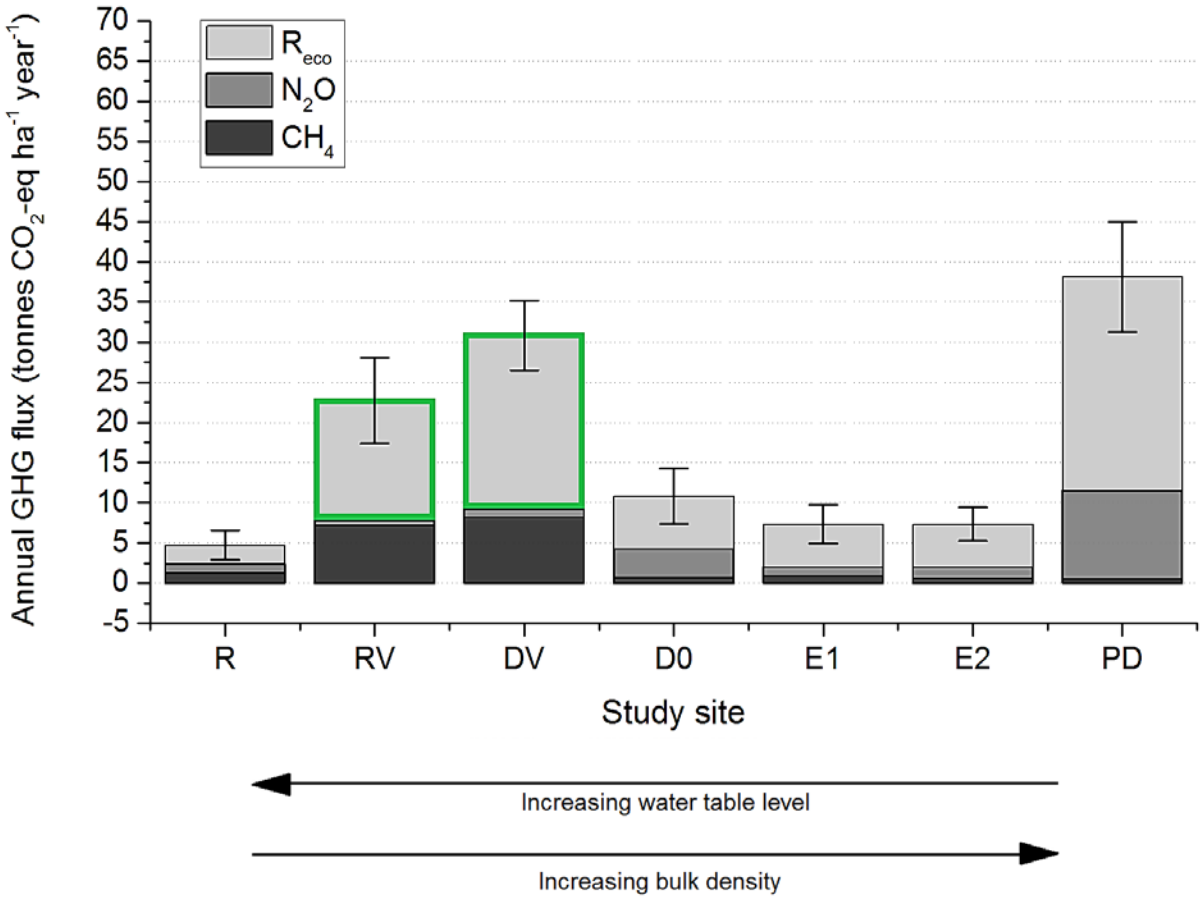


Figure 39. Mean annual GHG fluxes calculated for ecosystem respiration, methane and nitrous oxide released from three to four replicates of all study plots. Values are expressed as tonnes CO_2 equivalents over 100-year period. Sites without vegetation represent NEE, while vegetated sites represent only ER part of NEE and cannot be directly compared to bare sites. Green areas indicate soil heterotrophic and autotrophic respiration from the vegetated sites RV and DV.

For both years, the emissions including all three gases were considerably higher for plot PD in comparison to the other bare plots (Table 11); however no significant (ANOVA) differences in potentials between the all plots were detected. In general, all study sites showed positive annual GHGs, however, the lowest annual GHG emission were identified at the restored areas. The mean annual GHG emission over the investigation area, including the CO_2, N_2O and CH_4 emissions ranged between 4.7 ± 1.8 t CO_2-eq ha^{-1} $year^{-1}$ (R plot) to 38.1 ± 6.8 t CO_2-eq ha^{-1} $year^{-1}$ (PD plot) and thus, both min and max values characterized for non-vegetated bare peat plots differing in water table level and soil properties (Figure 39).

In view of the total area of the Himmelmoor, the emissions of R and E plots, extrapolated to the amount of hectares, are considerably lower than emissions of other sites. Upon the whole, annual GHG emissions were driven mostly by the CO_2 exchange (50% to 74%), and the contribution of N_2O and CH_4 ranged from 3% to 33% and from 1% to 32%, respectively.

4.4 Incubation experiments

4.4.1 Physical-chemical characteristics of the incubation material

Soil incubation material and soil solution

Soil properties of the incubation material taken in 2015 and 2016 from the on-going extraction area (E) and area rewetted in 2004 (RV) were found to be very similar (§ 4.2.2). Both soil profiles were described as Ombric Fibric Histosols and demonstrated a similar stratigraphy to the profiles dug in 2016. Three soil sampling horizons from plot E, one mixed sample from the two upper horizons at plot RV, as well as the soil samples used for soil solution extractions are shown in Tables 12 and 13.

Table 12. Soil description of the soil pit and sampled horizons at Site E.

E: ongoing extraction area
Location: Himmelmoor (53°44.488’ N, 009°50.713’ E) Date of sampling: 23.03.2015 Water level: -40 cm Vegetation: absent

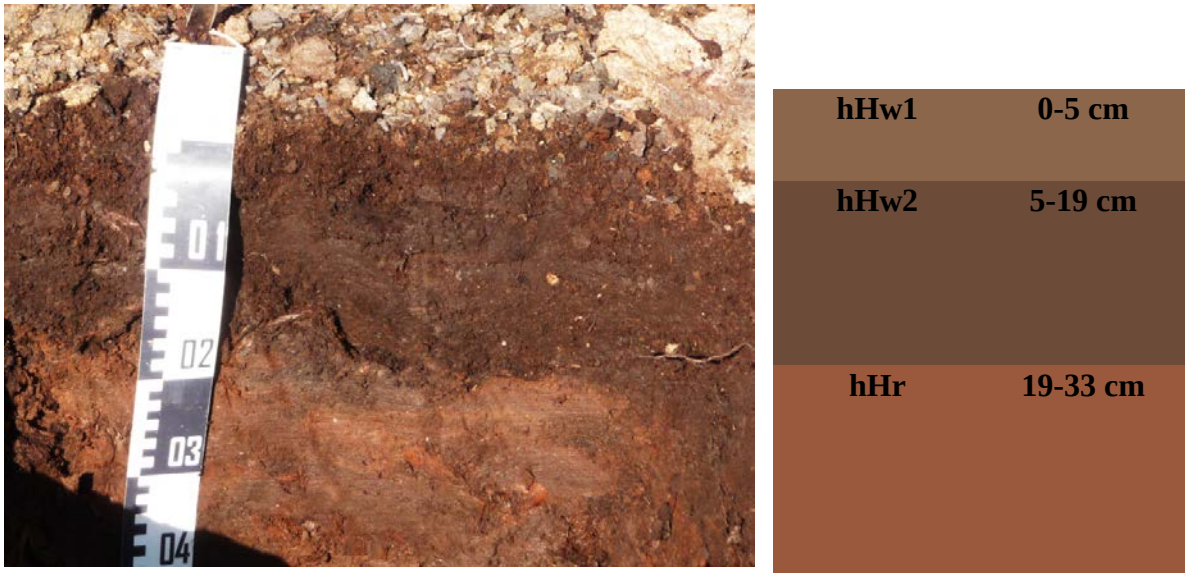


Figure 40. Soil pit at Site E (Vybornova, 2015)

Soil type:

- German classification: erodiertes Normhochmoor aus organogenem Sphagnum Torf (eHH: og-Hhs-qh)
- WRB classification: Ombric Fibric Drainic Histosol
- US Soil Taxonomy: Typic Sphagnofibrist

Table 13. Soil description of the soil pit and sampled horizons at Site RV.

RV: area rewetted in 2004	
Location: Himmelmoor (53°44.652’ N, 009°50.817’ E) Date of sampling: 23.03.2015 Water level: +11 cm, all horizons were situated below the water level Vegetation: *Sphagnum, Eriophorum angustifolium, Erica tetralix, Molinia caerulea*	

Figure 41. Mixed soil sample from the two upper horizons at Site RV (Vybornova, 2015)

Soil type:

- German classification: Normhochmoor aus umgelagertem, durchmischten Sphagnum Torf (HHn: om-Hhs-qhi)
- WRB classification: Ombric Fibric Histosol
- US Soil Taxonomy: Typic Sphagnofibrist

Figure 42. Soil samples from Site RVt used (i) for extraction of the first soil solution and DOC (left picture, 23.03.2015), and (ii) for extraction of the second soil solution (right picture, 02.06.2015, source: Laure Höppli).

As shown in Table 14, the samples taken from Site E showed different chemical, but similar physical soil properties compared to the mixed sample Site RV. Soil pH values at both sites were characterized as acidic in all sampled horizons. The highest pH_{H2O} value of 4.9 was found in the upper E horizon (0-5 cm), and decreased with depth at both sites. In contrast, the electrical conductivity was found to increase with the depth at Site E and was almost twice as high in the topsoil of Site RV than in the topsoil of Site E. The water content also increased with depth at Site E, and was the highest in the RV mixed sample, which contained up to 91% water according to the soil wet mass. The highest water content of the max. WHC was found in the RV-1 sample, where it amounted to 78.4%.

Table 14. Physical and chemical properties of the sampled soil horizons, and the two filtrates of the second soil solution used for the incubation experiment. Both filtrates were filtered through glass filter funnel with pore size of 100µm (DOC+M) and one half of the filtrate was autoclaved (DOC, sterile). Shown are mean values of laboratory replicates ± standard deviation.

Parameter	**Unit**	**Soil sample**				**Soil solution**	
		E-1 (0-5 cm)	**E-2 (5-19 cm)**	**E-3 (19-33 cm)**	**RV-1 (0-15 cm)**	**100 µm**	**100 µm sterile**
C_{org}	(%)	54.7±0.1	54.7±0.1	54.4±0.1	50.0±0.2	12.8	4.7
N_{tot}	(%)	1.3±0.1	1.1±0.2	1.1±0.2	1.3±0.3	1.3	0.5
C/N	-	44.4	49.9	48.7	40.0	10.1	9.7
Water content	(%)	74.8±1.4	85.8±1.3	88.9±0.9	90.9±2.5		
$WC_{soil\text{-}dry}$ of max.WHC	(%)	47.0±2.1	58.8±12.6	61.6±5.7	78.4±19.7		
soil pH_{H2O}	-	4.9	4.8	4.8	4.1		
electrical conductivity	(µS/cm)	25.1	29.4	40.7	47.1		
C_{mic}	($mg/g_{soil\text{-}dry}$)	3.36±0.09	4.22±0.83	4.90±0.34	9.35±1.46		
N_{mic}	($mg/g_{soil\text{-}dry}$)	0.03±0.02	0.01±0.01	0.01±0.01	0.08±0.02		
NH_4^+	($mg/g_{soil\text{-}dry}$)	0.24±0.01	0.21±0.03	0.23±0.01	0.02±0.00		
	(mg/l)					0.54	0.46
NO_3^-	($mg/g_{soil\text{-}dry}$)	0.01±0.00	0.001±0.00	0.002±0.00	0.00		
	(mg/l)					0.00	5.49
NO_2^-	($mg/g_{soil\text{-}dry}$)	0.00	0.00	0.00	0.00		
	(mg/l)					0.00	0.00
DOC	(mg/l)		48.56±4.31		38.75±4.65		
DN	(mg/l)		7.38±2.19		1.22±0.18		

Chemical analysis showed similar contents of C_{org} and N_{tot} in the sampled soil horizons, and calculated C/N ratios ranged from 40-49. In contrast, the microbial C and N concentrations were very low at Site E, decreasing with depth and ranging between 3.4 to 4.9 $mg/g_{soil\text{-}dry}$ and 0.03 and 0.01 $mg/g_{soil\text{-}dry}$, respectively, and both concentrations were twice as large in the RV-1 horizon. No nitrite was identified in any of the samples, and the nitrate and ammonium concentrations were the highest at Site E.

A nitrate concentration of 5.49 mg/l was found in the autoclaved filtrate 100-µm sterile, whereas the 100-µm-filtrate showed the higher concentration of NH_4^+. In addition, the concentrations of organic carbon and nitrogen were more than twice as high in the 100-µm-filtrate, despite the C/N ratio being the same for both filtrates (Table 14).

Soil properties of peat material after incubation

A change in the soil properties of aerobe and anaerobe samples was identified after the incubation experiments. The microbial carbon content showed a reduction of between 7% and 40% for the majority of the aerobe treatments after incubation, with the highest losses seen in the E-3 and RV-1 horizons. In contrast, the microbial N concentration was found to increase in all treatments with 100-µm-filtrate, and in all samples from the E-3 horizon increased by up to 5 times. Additionally, the C_{org} content of all aerobically incubated samples showed the largest reduction in samples from the upper E-1 horizon. The C/N ratio showed a considerable increase in RV-1 samples with up to a 22% increase, and a 7-20 % reduction was observed in all treatments of Site E samples.

All anaerobe samples also showed increased microbial C concentrations, which were up to 10 times higher for E horizons (E-1-100-µm-sterile; 42.3 mg C_{mic} /$g_{soil\text{-}dry}$) and twice as high for RV-1 horizon (18.3 mg C_{mic} /$g_{soil\text{-}dry}$). Moreover, in comparison to the aerobe experiment, this increase was very similar between all treatments. The microbial N content displayed the largest increase in the most anaerobe treatments of up to 90% for E horizons (E-3-control; 0.6 mg N_{mic} /$g_{soil\text{-}dry}$), whereas a lower increase was found for the samples from the rewetted plot (RV), where up to a 70% increase was observed (0.4 mg N_{mic} /$g_{soil\text{-}dry}$).

4.4.2 Aerobic and anaerobic C mineralization: gas production potentials

Only C (CO_2 + CH_4) mineralization rates are reported in this work since all oxic and anoxic incubations showed very low or even negligible N_2O production over the whole incubation period, with measured gas peaks in the bottles under the lowest standard GC concentration.

Cumulative CO_2 production and production potentials from control treatments

The aerobe CO_2 production potential (P_{CO_2}) differed significantly between control samples from extraction site E and the rewetted site RV. All five replicates sampled from Site RV showed significantly higher P_{CO_2} than replicates from Site E (Figure 43). Similarly, the anaerobe production of CO_2 from the control treatment at Site RV was larger than from control samples taken at different depths from Site E.

Aerobe and anaerobe cumulative CO_2 production curves and their calculated linear regressions are shown in Figure 44. Statistical analysis revealed that gas production of CO_2 normalized to soil organic carbon content was significantly higher in the aerobic (red scatter plots) than in the anaerobic (blue scatter plots) incubations. In addition, the mean cumulative CO_2 production rates showed a similar tendency in the relative differences between horizons for both aerobic and anaerobic samples, and continued to increase until the end of the incubation period. Both aerobic and anaerobic CO_2 production rates in all sampled E and RV horizons were highest at the beginning of the incubation, showing a clear reduction within the first twenty days of the experiment.

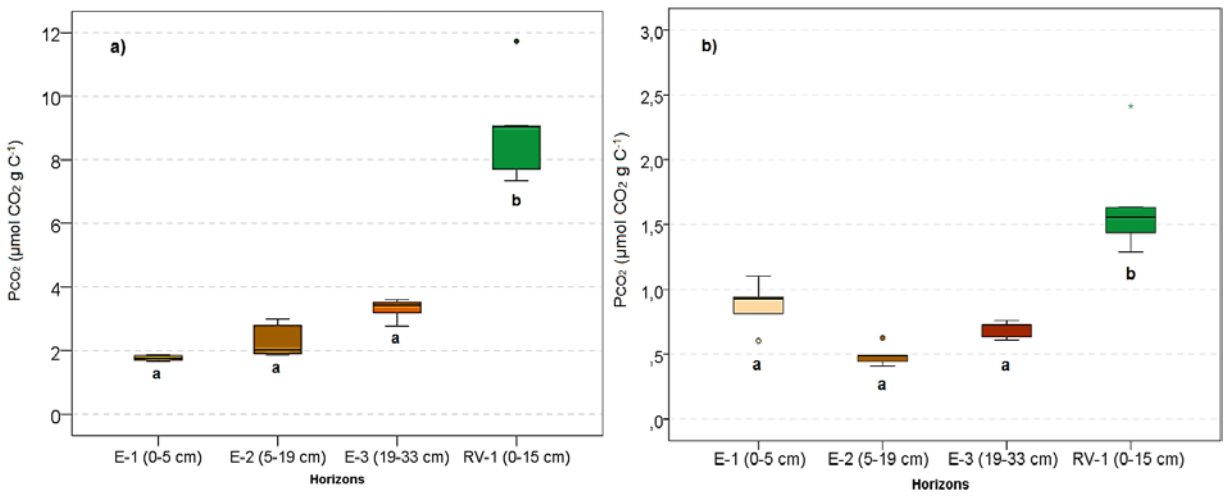


Figure 43. CO_2 production rates referred to g soil organic C^{-1} of the control aerobe (left picture) and anaerobe (right picture) incubations with soil material from the three sampled horizons of the drained E and the rewetted RV sites. Homogeneous subsets determined by Tukey´s test are displayed with a or b.

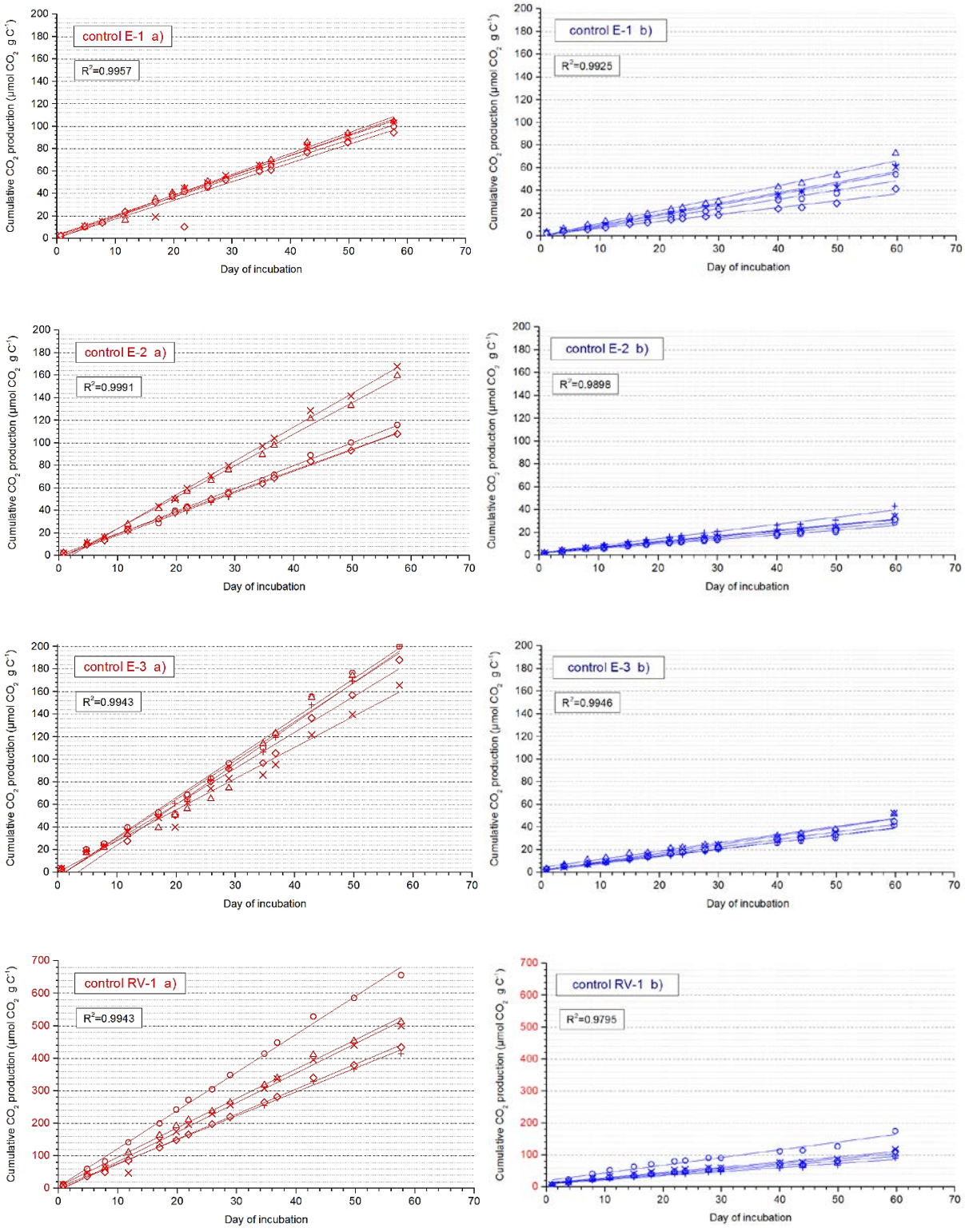


Figure 44. Production curves of carbon dioxide (in µmol CO_2 g of soil C^{-1}) of all control a) aerobe (red) and b) anaerobe (blue) horizons with respect to the duration of incubation. The scatter plots show the amount of CO_2 measured over the incubation period for five replicates, and linear regression lines for each replicate, alongside their corresponding coefficient of determination R^2. It should be noted that plots RV-1 a) and b) have different Y-Axis scales.

Influence of soil depth on CO_2 production from site E control samples

Aerobe control samples taken from three depths at Site E revealed significant differences ($p<0.001$) in CO_2 production between the lower (E-3) and two upper horizons (E-1, E-2) (Figure 44 and 45). The incubation experiments demonstrated an increasing carbon dioxide production with depth, with mean P_{CO2} in the third horizon (E-3) being twice that observed in the first horizon (E-1). Control anaerobe incubations displayed a different behavior, in that the production of CO_2 decreased over depth, and only rates of production in the first horizon were significantly higher than those observed in the middle horizon (E-2) ($p=0.01$).

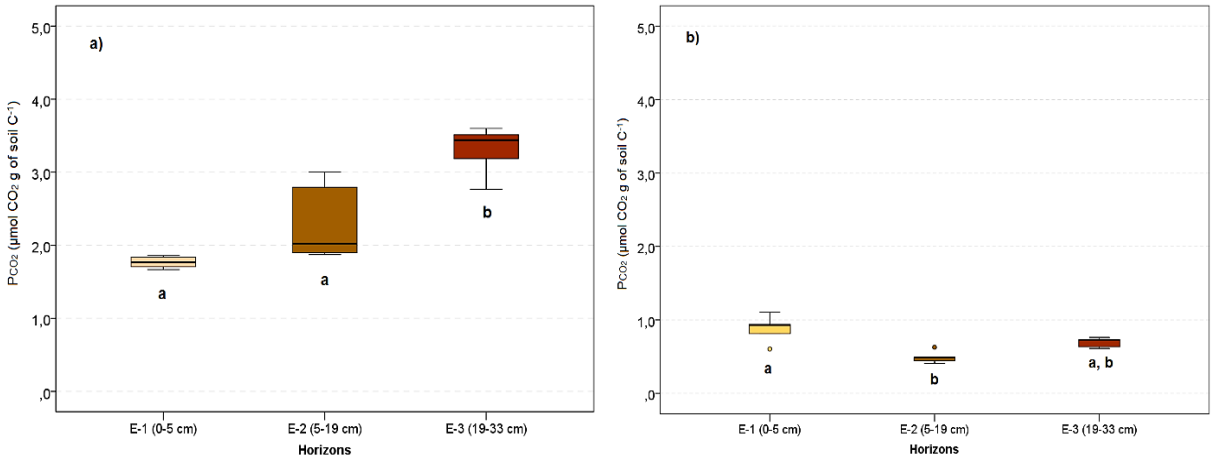


Figure 45. CO_2 production rates referred to g soil organic C^{-1} of the control aerobe (left picture) and anaerobe (right picture) incubations with soil material from the three different depths of the site E. Homogeneous subsets determined by Tukey´s test are displayed with a and b.

Cumulative CH_4 production and production potentials from control treatments

All aerobe control samples showed low to no significant methane production over the 133 day incubation period, with measured gas in the bottles under the lowest standard GC concentration of 1.79 ppm. Anaerobe production rates of control samples from the middle and lower E horizons were also low, whereas samples from E-1 and RV-1 demonstrated a distinctly higher cumulative CH_4 production (Figure 46), increasing over time until the end of the 366 day incubation period. Calculated methane production rates for RV-1 replicates amounted to 0.02 ± 0.003 µmol CH_4 g of soil C^{-1}, and up to 0.5±0.04 µmol CH_4 g of soil C^{-1} for the first 3 months of incubation; and from 0.23±0.01 µmol CH_4 g of soil C^{-1} and up to 0.5±0.01 µmol CH_4 g of soil C^{-1} for the next 9 months. Calculated production rates for the E-1 horizon were up to 10 times lower than RV rates, with a mean of 0.002 ± 0.001 µmol CH_4 g of soil C^{-1}.

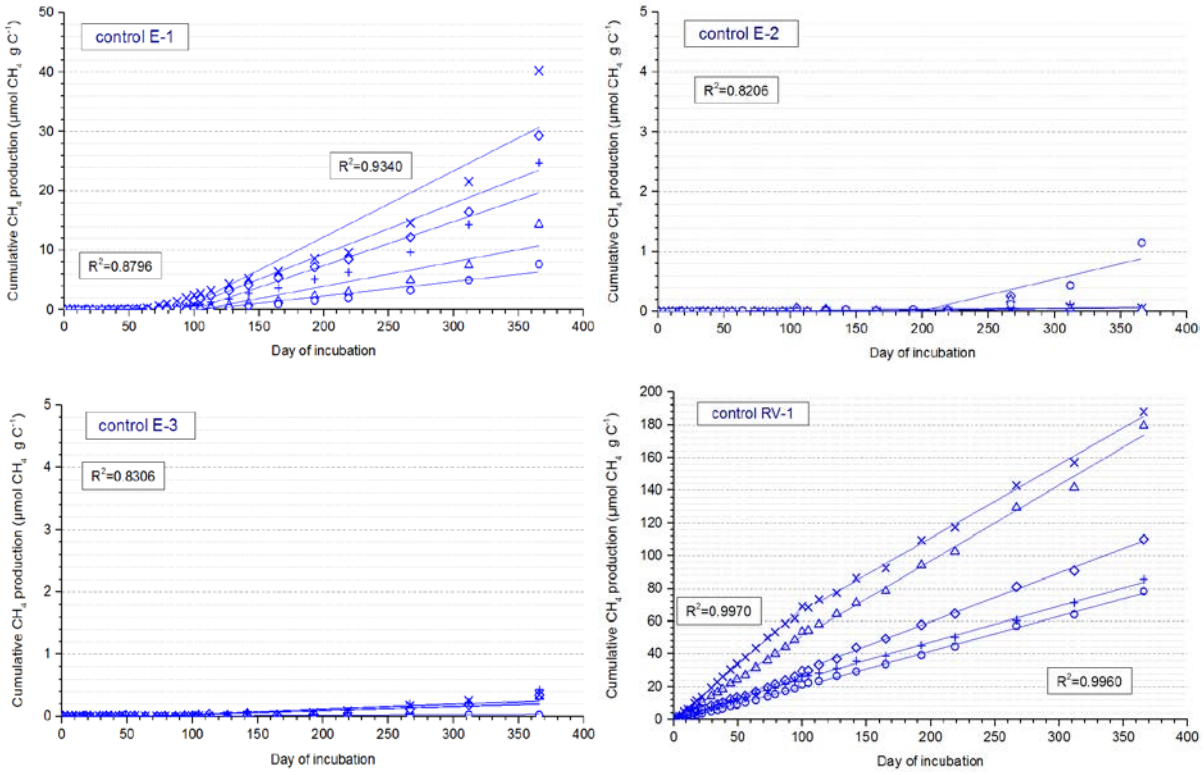


Figure 46. Production curves of methane (in µmol CH_4 g of soil C^{-1}) from control anaerobic incubations with samples taken from the E-1, E-2, E-3 and RV-1 horizons with respect to the duration of incubation.: Scatter plots show the amount of CH_4 measured in five replicates, and the linear regression lines for the initial three months, and subsequent nine months of incubation for each replicate, alongside their corresponding coefficients of determination R^2. It should be noted that plots E-1 and RV-1 have different Y-Axis scales.

4.4.3 Effect of treatments with added sterile and non-sterile soil solution filtrates on CO_2 and CH_4 production of samples from different depths at Site E

Cumulative CO_2 production and production potentials from filtrate treatments

For the aerobe incubations, no significant differences in the carbon dioxide production of samples were found between the control- and filtrate addition treatments (100 µm and 100 µm-sterile) for the lower E-3 horizon (Figure 47), and for samples from the middle E-2 horizon incubated with filtrates 0.7 µm and 0.2 µm. In contrast, the samples from the upper E-1 horizon demonstrated significantly higher production rates for the controls than for the filtrate 100 µm-sterile addition treatment. No difference between both treatments with filtrates was found. Overall, aerobe control samples and aerobe samples with added filtrates showed significantly higher CO_2 production rates compared to the anaerobe samples, and an increasing difference between the two incubation types was observed with depth (Figure 47).

Anaerobe incubations of soil material from the upper horizon showed no significant difference in the CO_2 production from control and filtrate treatments. In contrast, the middle horizon was found to have significantly higher carbon dioxide production in all filtrate addition treatments (sterile and non-sterile). The treatment with sterile filtrate had the greatest gas production rates over the first two months, and the same effect was seen in soil samples taken from the E-3 horizon.

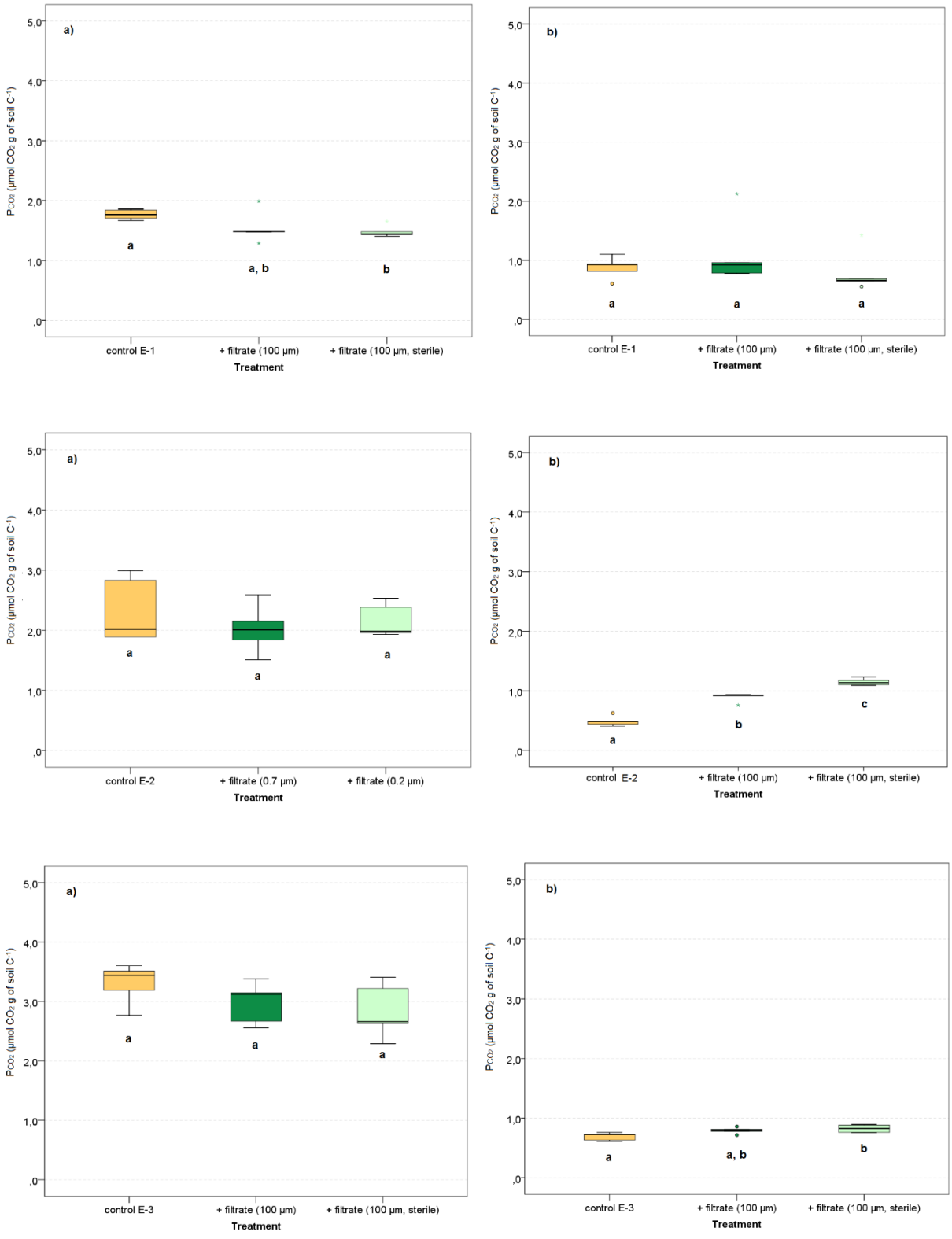


Figure 47. CO_2 production rates of the aerobe (left pictures) and anaerobe (right pictures) incubations over the first two months of incubation with soil material from the three sampled horizons at extraction site E: the upper – E1, middle – E2, lower – E3 horizons. Homogeneous subsets determined by Tukey´s test are displayed with (a), (b) and (c) using the Tukey test. It should be noted that no incubation for samples from E-2 with filtrates 100 µm and 100 µm sterile was made and that only the results of first incubation with filtrates 0.7 µm and 0.2 µm are shown.

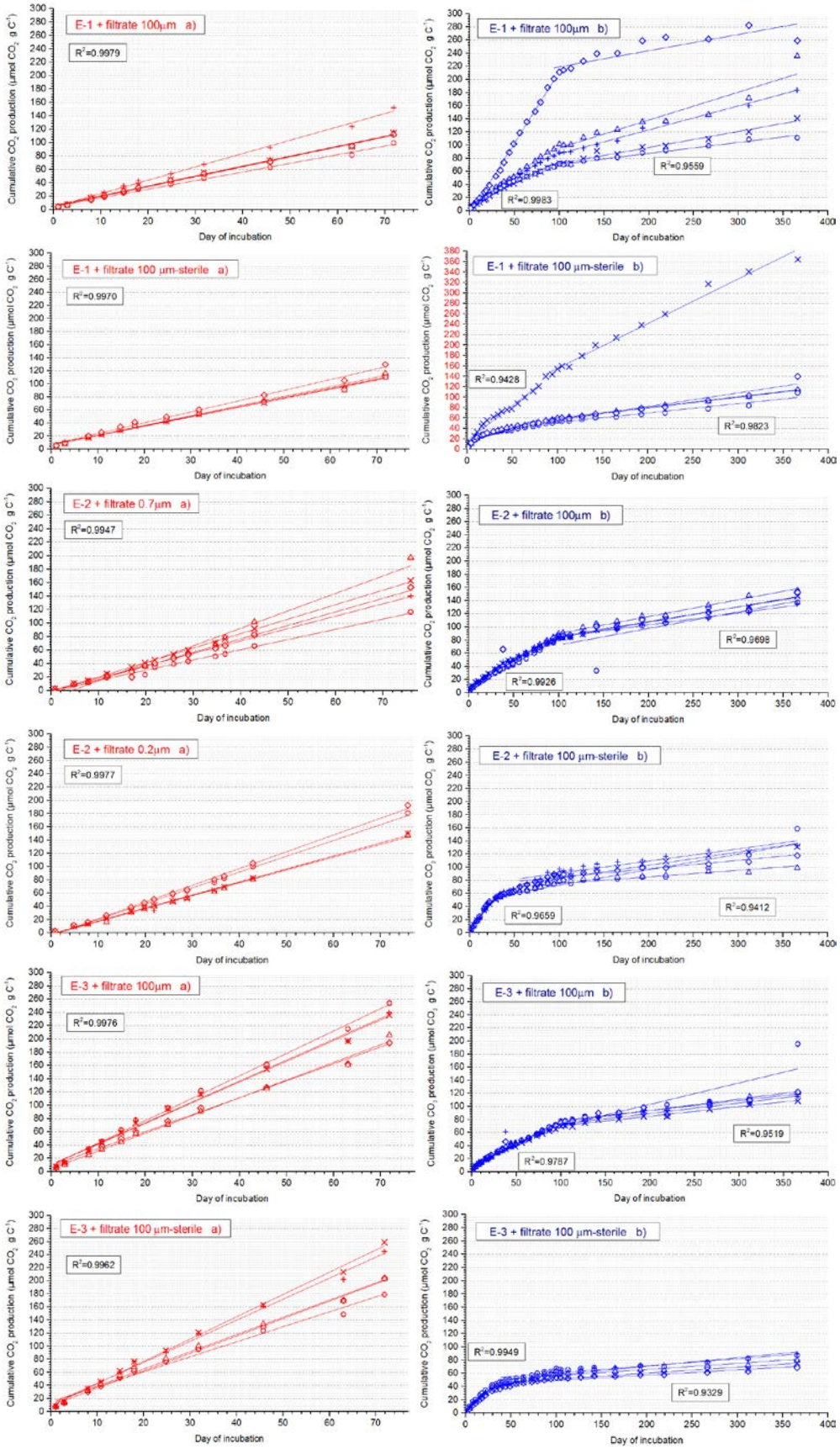


Figure 48. Production curves of carbon dioxide of added sterile and non-sterile filtrates for a) aerobe (red) and b) anaerobe (blue) incubated horizons at site E. Scatter plots show the amount of CO_2 produced in five replicates, and linear regression lines calculated for each replicate. It should be noted that anaerobic plots have broader X-Axis limits.

Cumulative CH_4 production and production potentials from filtrate treatments

Aerobe incubations showed very low to no significant methane production from samples taken from the three E horizons. In contrast, the anaerobe incubated samples demonstrated significantly higher production rates in the bottles incubated with replicates of filtrate "100 µm" compared to the control and "100 µm sterile" treatments, although for the lower E-3 horizon a strong difference was observed in the effects of both filtrates ($p<0.001$). The control samples from the horizon E-1 were not significantly lower than samples with added filtrates (Figure 49), and additionally, the control samples from horizon RV-1 were substantially higher than all filtrate treatments from Site E (Figures 50-51).

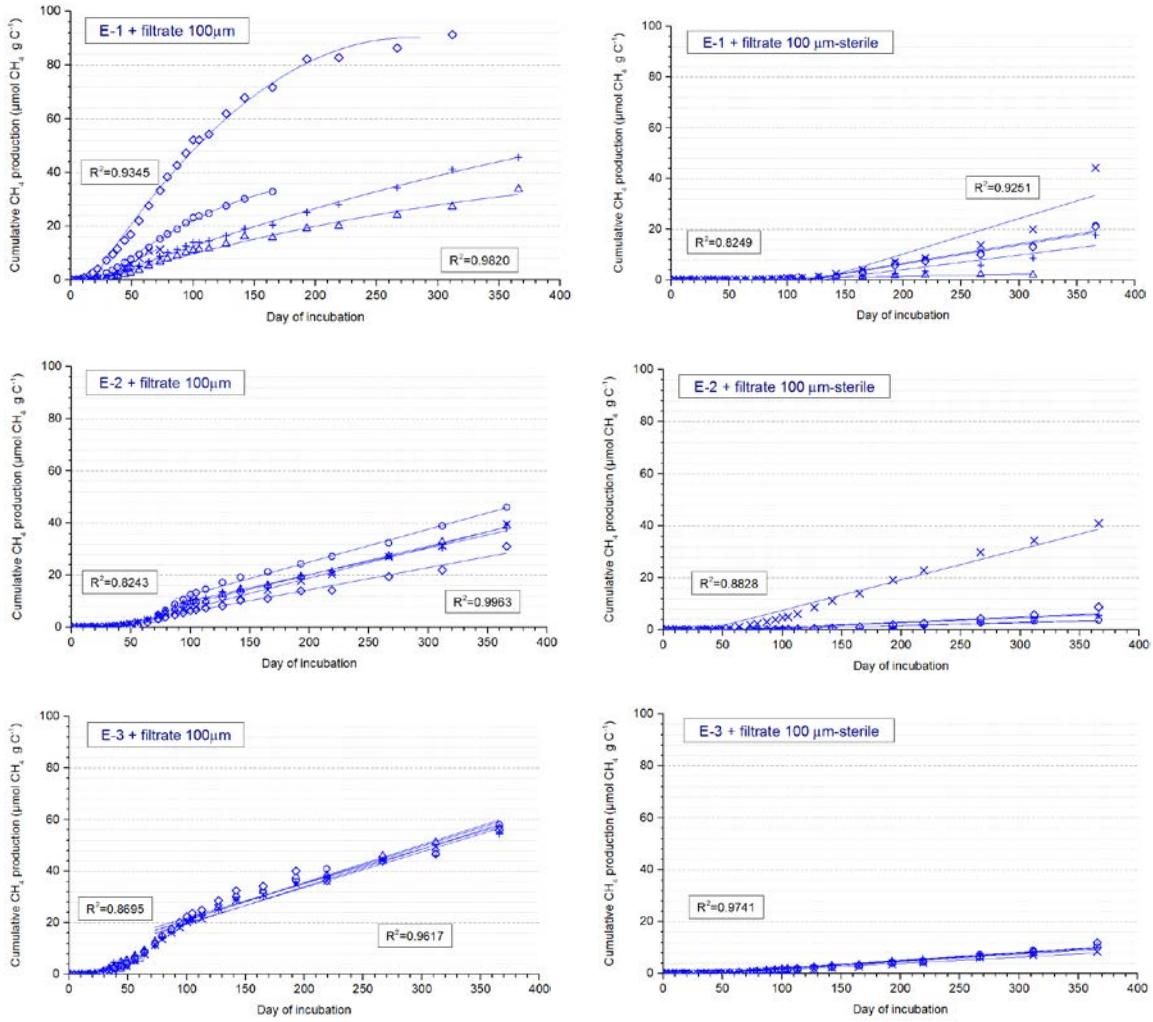


Figure 49. Production curves of methane of control anaerobe samples from the RV-1 horizon with respect to the duration of incubation. Scatter plots show the amount of CH_4 measured in five replicates, and linear regression lines calculated for days 0-60 and 60-430 of the incubation for each replicate, along with their corresponding coefficient of determination R^2.

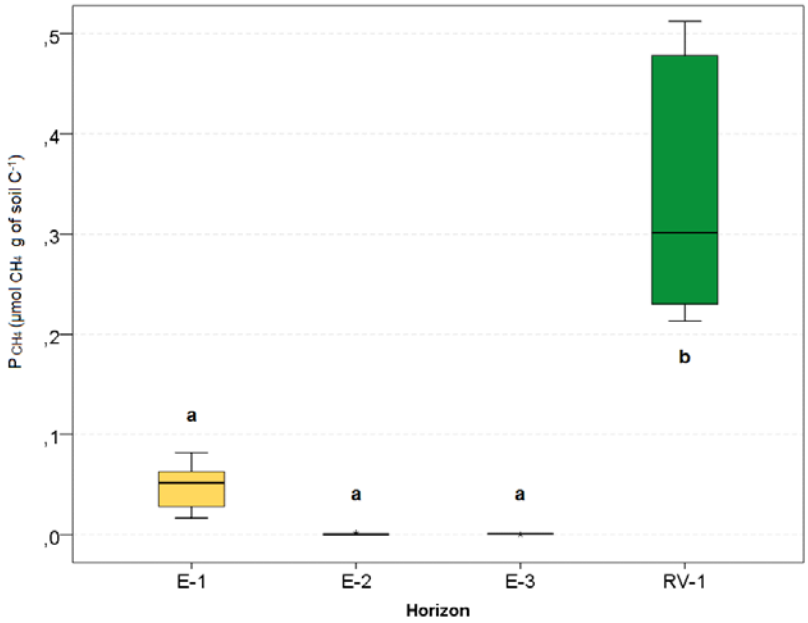


Figure 50. CH_4 production rates of the anaerobe incubations with soil material from the three sampled horizons of the extraction site E and one sample horizon of the RV site. Homogeneous subsets determined by Tukey´s test are displayed with (a) and (b).

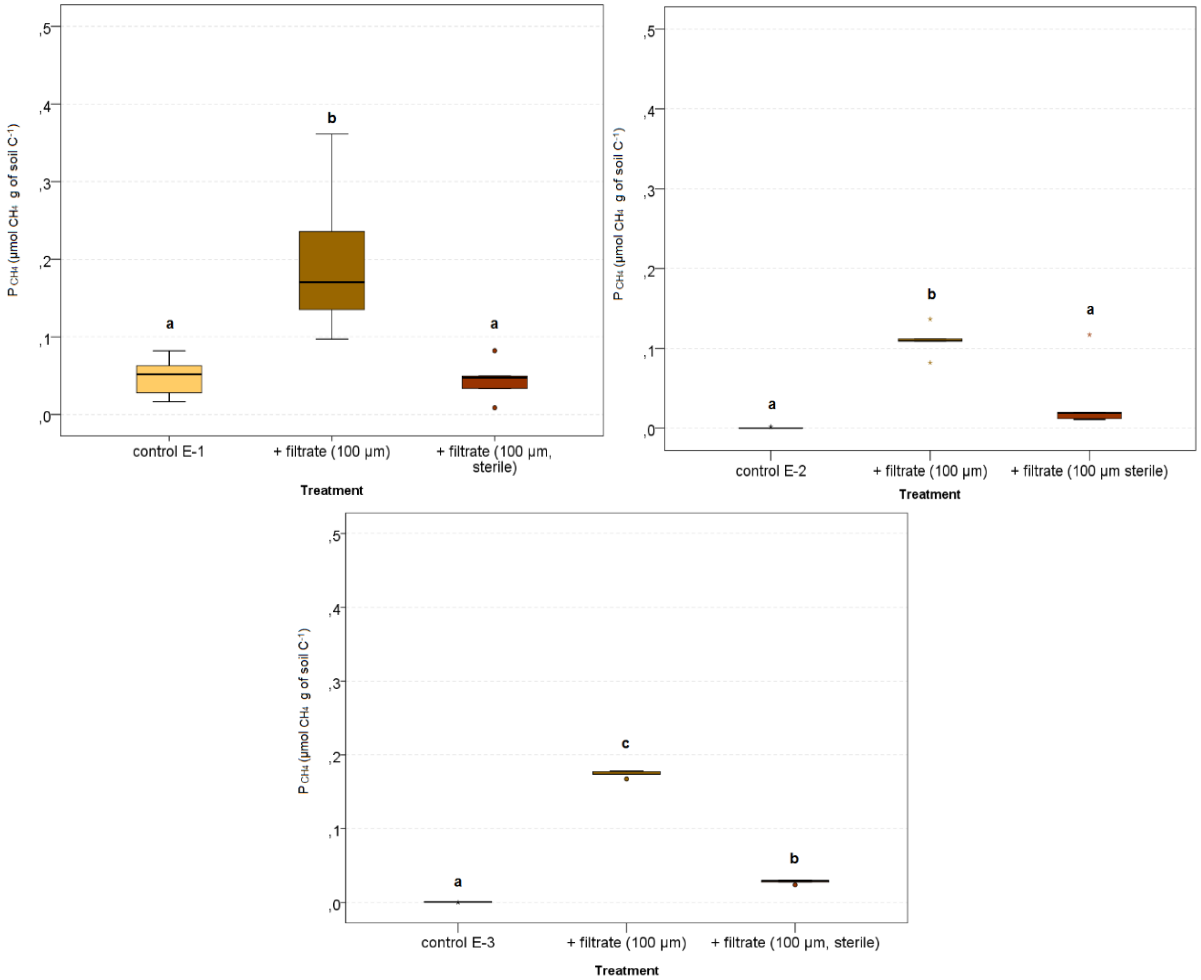


Figure 51. CH_4 production rates of the anaerobe incubations with soil material from the three sampled horizons of the extraction site E (E-1 upper left, E-2 upper right, E-3 lower picture) showing control and filtrate addition treatments. Homogeneous subsets determined by Tukey´s test are displayed with (a) and (b).

4.4.4 The effect of drying and rewetting on aerobe CO_2 production from Site E

The aerobe CO_2 production rates of incubated E-1 samples (0-5 cm) from both treatments after drying and rewetting showed a significant difference between the treatments, with a strong increase in the gas production with increasing degree of drying before rewetting and incubation (Figure 52). The mean CO_2 production potential of the control upper horizon amounted to 0.02 ± 0.003 µmol CO_2 g of soil C^{-1}, and was respectively 30% and 50% lower than rates of incubated samples that were dried to 25% and 10% of max. WHC. Furthermore, results from the first incubation showed that the addition of both filtrates from the first soil solution (0.7 and 0.2 µm) did not result in any increase in gas production of either the non-dried or dried samples of the E-1 horizon. Further results on the aerobe incubations can be found in Höppli (2015).

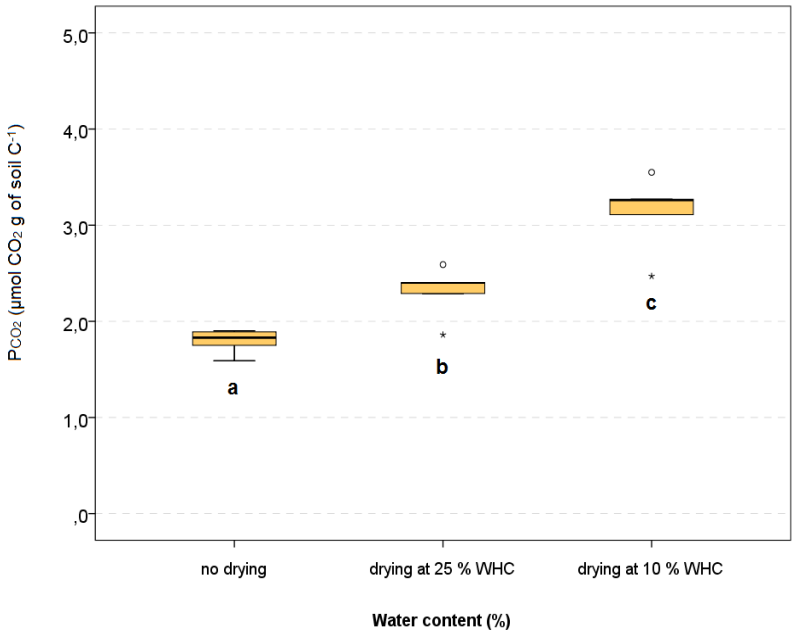


Figure 52. Aerobe CO_2 production rates of the incubation with non-dried soil material and two treatments with a different degree of drying from the first sampled horizon of Site E.

5 Discussion

5.1 Effect of rewetting on soil properties

The special rewetting management at the Himmelmoor bog has increased water table levels by an average of ~50 cm at the rewetted site R after 5 years of closing ditches compared to water table levels at the extraction site. Flooding has resulted in a more stable water level above the soil surface, less temporal variation in soil temperature, lower soil pH, lower humidification grades, and a reduced bulk density of peat samples. For example, the bulk density of surface peat in the former extraction area is by 10 % higher than in the site R after 5 years of rewetting, and by 60 % higher than in the site RV after 10 years of rewetting. However, the dry peat dam site (adjacent to the rewetted area) was characterized by denser (45% more dense) and more strongly decomposed (H8) peat compared to the rewetted area. This change is in agreement with the findings of Couwenberg (2011), who observed peat compaction and subsidence after drying due to a loss of supporting pore water pressure. In the Himmelmoor bog, after the peat dams were constructed, the peat was compacted using machines in order to make the peat wall more stable, to prevent water flow through it, and to support peat rewetting. A number of studies have suggested that peat subsidence and loss of carbon are the main consequences of peat oxidization and drainage (Ketcheson and Price, 2011; Hooijer et al., 2012). However, the majority of investigations describe the positive impacts of peat dams, most of all the rapid re-establishment of high water levels and restoration of past hydrological conditions (Jaenicke et al., 2010; Ketcheson and Price, 2011; Gonzalez et al., 2013; Ritzema et al., 2014). Two additional consequences of compaction identified also in this study were a higher soil carbon and nitrogen density observed at the peat dam and drained area, which result from increased peat oxidation and decomposition under the dry conditions. More than 25 years after closing the ditches in the Himmelmoor and 5 years after flooding, soil conditions still show the impact of former management practices. According to Schimelpfenig et al. (2013), this outcome can be expected, and the full restoration of soil conditions after ditch refilling can take decades to centuries to occur.

Rewetting was found to have a significant impact on microbial biomass and total nitrogen concentration. Rewetted bare and vegetated plots were characterized by microbial C and N concentrations in the upper horizons, which were 25-70 % higher than found in the drained plots. Previous studies on microbial activity report similar findings, with soil rewetting causing an immediate increase in bacterial growth and microbial biomass; and it being observed that the resulting growth rates are similar or even higher than those measured in permanently moist soil (Borken et al., 2003; Iovieno and Bääth, 2008). Furthermore, the increased rates of carbon dioxide production found in the present study after dry peat samples were rewetted are likely due to the strong positive correlations observed between microbial activity, water content, and degree of rewetting. However, no direct relationship between microbial activity and respiration rates was observed, which is likely due to a shift in the microbial community structure (Fierer and Schimel, 2002; Pietikäinen et al., 2005). Additionally, the drying-rewetting stress induced

a significant increase in soil microbial C and N concentrations, which were still detectable after 3 months of incubation (after drying of 10% of max. WHC).

Analyses of the dissolved organic carbon and nitrogen contents in the soil pore water reveal lower DOC (by 20 %) and lower DN (by 84 %) concentrations in rewetted site compared to extraction area. Previous studies observe similar DOC concentrations (from 30 to 110 mg/l) to those found here in rewetted and drained bogs in Europe (Glatzel et al., 2003; Wallage et al., 2009; Höll et al., 2009; Armstrong et al., 2010), as well as in the Himmelmoor itself (Gross, 2014). Moreover, recent published data suggests that peat rewetting may return DOC losses from the former extracted or degraded peatlands to levels recorded at pristine mires (Armstrong et al., 2010). Thus, despite the short initial peak of DOC contents in drainage water (Worall et al., 2007), which occurred in response to the peat disturbance, the rewetting shows a total reduction in DOC up to 70% (Gibson et al., 2009; Strack et al., 2011).

5.2 Temporal and spatial variation in GHG fluxes

The results of this study have revealed significant heterogeneity in GHG emissions between the investigated land-use types within the Himmelmoor bog complex. These are discussed below firstly for carbon dioxide and methane, and then for nitrous oxide.

5.2.1 Carbon dioxide and methane

Rewetted extraction areas and ditches

As a result of the opaque chamber measurement technique adopted in this study, only ecosystem respiration fluxes could be examined as no data on net ecosystem exchange of CO_2 (NEE) were available. The rewetted re-vegetated study site (RV) exhibited R_{eco} emissions that were up to five times larger (max flux: 5.2 µmol CO_2 m^{-2} s^{-1}) than observed at the non-revegetated rewetted site R. R_{eco} rates for the RV area in the present study are three times lower than that reported by Vanselow-Algan (2015). For example, in 2011 R_{eco} emissions ranged from 0 to 13 µmol CO_2 m^{-2} s^{-1}, whereas in 2010 the maximum emission was considerably higher at 20 µmol CO_2 m^{-2} s^{-1} (Vanselow-Algan, 2015). Hence, in comparison to the 2011 data, the R_{eco} rates observed in the present study are relatively low. Although the location of the four collars and coverage of “Sphagnum/Cotton grass” at the 2011 subsite differed from that examined in this study, it seems reasonable to assume that this area has been successfully rewetted after a period of 10 years in that it is becoming a weaker source of CO_2 and CH_4, and a stronger C sink. Both a reduction in soil R_{eco} and increased CO_2 uptake at non-vegetated sites have been reported by a number of authors in response to raised water levels (Komulainen et al., 1999; Chimner and Cooper, 2003b; Riutta et al., 2007). The observed changes in R_{eco} fluxes from rewetted sites with vegetation presented here are again similar to that found at other sites (Petrone et al., 2003; Schimelpfenig et al., 2013; Wilson et al., 2016), and are significantly lower than the mean annual R_{eco} flux observed from managed organic soils (7.7 - 17.7 µmol CO_2 m^{-2} s^{-1} Petersen et al., 2012). Additionally, R_{eco} emissions at the RV site and rewetted bogs with *Eriophorum*

vegetation from other studies are higher than observed at newly rewetted microtopes and even pristine bogs (Lafleur et al., 2003). For example, a non-vegetated site in Sweden, which was rewetted 15 years ago was found to emit up to 4.2 µmol CO_2 m^{-2} s^{-1}, with significantly lower R_{eco} fluxes than observed at the rewetted extraction site vegetated with *Eriophorum angustifolium* where the maximum flux was 11.1 µmol CO_2 m^{-2} s^{-1} (Jordan et al., 2016). After 15 years of rewetting, Günther et al. (2015) reported that the R_{eco} of a site revegetated with vascular plants in north-eastern Germany had decreased, with a maximum flux up to 3 µmol CO_2 m^{-2} s^{-1} and a net GHG balance similar to that of pristine mires.

The second vegetated plot, DV, exhibited R_{eco} fluxes which were at the higher end of the range reported for rewetted peatlands with vascular plants in Europe (0.2-8.9 µmol CO_2 m^{-2} s^{-1}, Elsgaard et al., 2012; Olafsdottir, 2015; Beyer and Höper, 2015; Gatis et al., 2016). R_{eco} flux rates for site DV were similar to those determined for organic soils under agricultural management (Maljanen et al., 2004; Jacobs et al., 2007; IPCC, 2014), whereas they were considerably lower than R_{eco} fluxes from nutrient-rich drainage ditches and fen areas, which can emit up to 30 µmol CO_2 m^{-2} s^{-1} (Stanley, 2015; Oertel et al., 2016). Furthermore, the mean annual ecosystem respiration of the D0 area (326.1 g CO_2 m^{-2} a^{-1}) was in the line with data from other bare ditches. Hyvönen et al. (2013) for example found growing season CO_2 emission rates of 111 – 391 g CO_2 m^{-2} at wet bare, but non-refilled drainage ditches in eastern Finland. The above fits in with results from a number of other studies, which found that previously dry drained study sites continued to act as a CO_2 source during the initial years after rewetting (Petrone et al., 2003; Waddington et al., 2010). The low CO_2 emissions observed at the ditch without vegetation (D0) and the rewetted plot (R) in this study are likely caused by the anoxic conditions which are unfavorable for decomposition processes (Landry and Rochefort, 2012). The magnitudes of CO_2 emissions from the rewetted location (R) and ditch without vegetation (D0) are within the range of those reported in the literature for temperate restored cutaway bogs in Europe with emissions of 466 g CO_2 m^{-2} a^{-1} (Drösler, 2008; Wilson et al., 2007) as well as those reported previously for the Himmelmoor, with Van Asperen (2015) observing CO_2 emissions of up to 1.6 µmol m^{-2} s^{-1} at site R. The results presented in the current study show that rewetting of the abandoned extraction area has significantly reduced R_{eco} fluxes by up to 56 %. The results presented in this thesis therefore support **hypothesis H1a;** that five years of rewetting leads to significantly lower CO_2 emissions than are observed at the extraction area.

In terms of temporal trends, although CH_4 emissions were highly variable, no clear annual trend could be detected. The large spatial and temporal variability of CH_4 emissions from rewetted peatlands has been reported in other studies, being linked to factors such as site-specific water table level (Couwenberg et al., 2011), time since rewetting (Tuittila et al., 2000; Waddington and Day, 2007), and soil environmental conditions (Günther et al., 2015). According to previous studies, microbial respiration and CH_4 production in areas with vascular plant cover can be enhanced by an increased supply of easily decomposable organic substrates from the vegetation (Finér and Laine, 1998; Minkkinen, 1999). Although the additional transport pathway for methane through the aerenchymae tissue to the atmosphere may have a negative effect on the

overall GHG balance (Whalen, 2005), peat areas vegetated with *Eriophorum* spp. may act as more efficient accumulators of carbon and providers of oxygen compared to bare peat or moss-covered areas (Wilson et al., 2013; Strack et al., 2014). These areas thus support microbial activity, the re-colonization of former peatland vegetation (Bhullar et al., 2014), and peat formation (Lee et al., 2016). Recent studies show that despite methane emissions increasing shortly after rewetting, in the longer term CH_4 emission rates are similar to that of undrained peatlands (Hiraishi et al., 2014). Additionally, the rewetting of bare nutrient poor peat-cut areas can lead to low CH_4 emissions until the vegetation cover re-establishes (Tuitila et al., 2000; Waddington and Day, 2007). During the present study, CH_4 emissions were generally low at the rewetted site R, high at vegetated sites, and negligible at bare drained plots. Mean methane emissions at the rewetted site R (9.5 nmol m^{-2} s^{-1}, max 49 nmol m^{-2} s^{-1}) are comparable to those reported for other rewetted peatlands, where CH_4 emission rates range from 4 – 125 nmol CH_4 m^{-2} s^{-1} (Wilson et al., 2013; Cooper et al., 2014; Günther et al., 2015; Karki et al., 2016). However, due to the re-establishment of plants with aerenchymatous tissue, it is likely that CH_4 emissions from the rewetted area will significantly increase over time. In comparison to rewetted site R, high summer methane emissions (up to 353 nmol CH_4 m^{-2} s^{-1}) were detected at the rewetted and vegetated with *Eriophorum* RV site, although mean annual emissions are still in line with that reported for other rewetted temperate bogs (Järveoja et al., 2016). A similar difference in mean annual and seasonal CH_4 emissions was found at the refilled ditch site between vegetated (DV) and bare (D0) subsites. Both plots were identified as sources of CH_4, and the ditch with *Eriophorum* plants had a mean methane emission of 50.3 nmol m^{-2} s^{-1}, which is up to five times higher than emitted annually at the adjacent bare plot. An increase in CH_4 emissions after re-colonization by vascular plants has been reported in the literature (Ruitta et al., 2007; Couwenberg et al., 2011), with Waddington and Day (2007) finding an increase from 0.4 to 50.2 nmol m^{-2} s^{-1} after re-colonization at a restored peatland in Canada. These results suggest that vegetation composition and the presence of vascular plants are good proxies for estimating methane fluxes from rewetted peatlands. The data presented above support the **hypotheses H1c** that CH_4 emissions are significantly increased after rewetting; and **H3b** that ditches vegetated by vascular plants are CH_4 hotspots. Although these findings suggest that rewetting of the RV area did not achieve its aim to decrease net radiative forcing effect, it is evident that there has been reduction in CH_4 emissions at the site since the study of Vanselow-Algan (2014), where after a 4 year period mean annual CH_4 fluxes decreased from 197.5 in 2011 to 42.7 nmol CH_4 m^{-2} s^{-1} in 2015, respectively. This decrease is responsible for a net-reduction of GHG emissions at the site, and suggests that it is possible to achieve a near-natural GHG balance. In terms of the bare rewetted site R, future CH_4 emissions will depend on the plant species which re-colonize this area, and hence on the supply and amount of fresh root exudates and litter (Ström et al., 2003). Furthermore, the fact that the majority of the observed CH_4 uptake at rewetted and drained sites occurred over the summer and autumn periods may point to biological soil uptake, which is usually larger during periods characterized by warmer temperatures (Pfeiffer, 1998; Van Asperen, 2015).

Table 15. A review of CO_2 and CH_4 fluxes measured at natural, degraded and rewetted bogs. Values reported in the existing literature are compared alongside those found in the present study (Vybornova, 2017).

Type of peatland	Location	GHG species	Annual emissions	Study
Pristine bog	Germany	CO_2 (t C ha^{-1} a^{-1})	-1.3 - 0.1	Beetz et al., 2013
	Canada		2.3	Glatzel, 2016
Bog under extraction	Germany	CO_2 (t C ha^{-1} a^{-1})	1.3 ± 0.1	Beyer&Höper, 2015
	Canada		1.8	Glatzel, 2016
	Temperate bog		0 - 0.6	Couwenberg, 2011
	Germany		1.5	**Vybornova, 2017**
Bog under agriculture	Denmark	CO_2 (t CO_2 ha^{-1} a^{-1})	0 - 78	Elsgaard et al, 2012
Drained bare areas	Temperate climate	CO_2 (t CO_2 ha^{-1} a^{-1})	10.27	Wilson et al., 2016
	Estonia		7.22	Järveoja et al., 2015
	Germany		5.5 - 7.0	**Vybornova, 2017**
Rewetted extracted peatlands	Temperate climate	NEE (t CO_2 ha^{-1} a^{-1})	-0.84	Wilson et al., 2016
	Germany	NEE (t CO_2 ha^{-1} a^{-1})	0.6 ± 1.43	Vanselow-Algan, 2014
	Estonia	CO_2 (t CO_2 ha^{-1} a^{-1})	4.8	Järveoja et al., 2015
	Canada		9.8	Glatzel, 2016
	Germany		2.45	**Vybornova, 2017**
Pristine bog	Germany	CH_4 (kg C ha^{-1} a^{-1})	2.8 - 5.8	Beetz et al., 2013
	Canada	CH_4 (kg C ha^{-1} a^{-1})	50	Glatzel, 2016
Bog under extraction	Germany	CH_4 (kg C ha^{-1} a^{-1})	0.9 ± 1.3	Beyer&Höper, 2015
	Canada	CH_4 (kg C ha^{-1} a^{-1})	10	Glatzel, 2016
	Germany		26.4	**Vybornova, 2017**
Drained bare areas	Temperate		0.1	Couwenberg, 2009
Rewetted	Canada	CH_4 (kg C ha^{-1} a^{-1})	250	Glatzel, 2016
	Canada		106	Christen et al., 2016
	Temperate		37 – 127	Couwenberg, 2009
	Germany		33.7 - 156	**Vybornova, 2017**

Based on fluxes measured using the eddy covariance (EC) technique, Holl (2016) found that the flooded area in the Himmelmoor had lower annual cumulative CO_2 emissions (35 - 40 %) and higher annual cumulative CH_4 emissions (51 - 84 %) than the extraction mining area over the period 2012-2013. These EC based estimates have similar mean values to those found in this thesis using the chamber method. Additionally, the same seasonal trend in R_{eco} was identified in the two studies, with the highest fluxes (up to 10 µmol CO_2 m^{-2} s^{-1}) being observed in June-July for vegetated ditch plots; and maximum fluxes (up to 2.5 µmol CO_2 m^{-2} s^{-1}) observed in late summer for the rewetted and extraction areas. As was found in the present study, Holl (2016) also identified significant differences in annual CO_2 balances for the two

measurement years. Additionally, the GHG emissions calculated by Holl (2016) for the whole former extraction area in 2012-2013 were up to 3 times higher than estimated in the present study. Despite these similarities, differences in annual gas emissions were also found between the two studies, which are likely due, firstly, to the different measurement techniques employed and, secondly, to the EC footprint that included not only flooded areas, but also dry extraction areas and peat dams in one sector. Annual cumulative CO_2 fluxes from extraction areas measured by Holl (2016) for example, were two to three times higher than observed in this study for plot E (5.4 ± 1.3 t CO_2 ha^{-2}), although flux estimates from the two studies are similar for rewetted bare sites (mean annual rate 27.8 ± 4.4 mol CO_2 m^{-2}, 32.9 ± 5.8 mol CO_2 m^{-2} for Holl et al. (2016) and this study, respectively). The fact that the rewetted site, adjacent ditch and dam area were investigated separately in this study could explain the observed differences in GHG emissions between the two studies. In contrast to the findings presented above for the Himmelmoor peatland, Marushchak et al. (2016) reported that methane fluxes measured by the EC technique were lower than those calculated using chamber measurements in subarctic Russian tundra due to site-specific differences in land cover type within and outside of the EC footprint.

Dry peat dam and extraction area

Overall, a clear difference in R_{eco} fluxes was observed between the study plots throughout the study period. The strongest seasonal temperature trend in R_{eco} fluxes was observed at sites RV, DV, and the dry peat dam site (PD). The measurements revealed that fluxes from the PD were the highest across all microsites; and were usually over 10 times higher than fluxes from the bare rewetted plot R. Mean R_{eco} rates calculated for Site E (0.35 µmol CO_2 m^{-2} s^{-1}) are similar to those reported for other abandoned and extraction areas. For example, Shurpali et al. (2008) reported that emissions at a bare peat cut area in eastern Finland ranged from 0.3 to 0.71 µmol CO_2 m^{-2} s^{-1}, and that three times lower R_{eco} was three times lower here than in areas cultivated with reed canary grass. Järveoja et al. (2015) observed that R_{eco} fluxes from an abandoned bare peat extraction area (mean flux 0.83 µmol CO_2 m^{-2} s^{-1}) were 50% higher than from areas which had undergone 3 years of restoration with *Sphagnum* and *Eriophorum* vegetation.

There are only few studies in the literature, which examine GHG fluxes from peat dams. Ketcheson and Price (2011) report that peat dams, due to their high bulk density, may result in large water pressure differences across the dam, and, hence, in hydraulic lifting and water flow through the dam. Although summer CO_2 emissions from the PD measured in the present study (0.5 to 7 µmol m^{-2} s^{-1}) are lower than the fluxes observed in 2012 for the same plot (Van Asperen, 2015), they are substantially higher than the flux values reported for other dry ombrotrophic bogs (Blodau et.al., 2007), managed peatland areas (Alm et al, 2007), and the extraction area of the Himmelmoor. Similarly as in present study, Van Asperen (2015) observed large CO_2 fluxes of maximum 20 µmol m^{-2} s^{-1} in 2012 on the peat dam plot, which were 10 times higher than from the rewetted plot R. These findings support **the hypothesis, H2a,** that peat dams are hot-spots of CO_2 emissions. The most likely reasons for the spatial variability in

ecosystem respiration between the peat dam and extraction areas are the higher WTL in the extraction area, as well as differences in peat properties at this site, such as a lower microbial biomass and lower peat oxidation.

5.2.2 Nitrous oxide (N_2O)

The measurements described in the present study reveal a high spatial variability in N_2O fluxes within the study sites, as has been reported previously for other peatland microtopes (Landry and Rochefort, 2012). For example, this thesis identified clear differences in N_2O fluxes from the peat dam site across replicate collars located only 2 m apart. In addition, chamber measurements from one collar at the adjacent rewetted site clearly show lower N_2O emissions than the two other collars, which points to consistently lower levels of biological activity at this location. This spatial heterogeneity in flux magnitude is an important aspect to consider when upscaling flux measurements.

As described above for CO_2, consistently lower N_2O emissions were measured at the flooded area than at the peat dam and ditch without vegetation. Compared to the existing literature, the N_2O emissions observed at the PD area (up to 13 nmol m^{-2} s^{-1}) are higher than those reported for other drained bogs and peat extraction areas (Maljanen et al, 2010; Jordan et al., 2016). The reason for this difference is unclear, although it is possible that high rates of nitrification and the consequent high nitrate (NO_3^-) content of the peat samples (Aerts 1997; Brumme et al. 1999), as well as the dry conditions (Schiller and Hastie, 1994) and low soil pH (Aerts, 1997) may promote increased N_2O emissions, inhibiting both the amount of N_2O reductase and N_2 production. Wrage et al. (2001) hypothesize that the dry oxic conditions in peat dams favor nitrification processes, because nitrate is substrate for peak N_2O emissions produced by denitrification (Stolk et al., 2009). This hypothesis is supported by the high nitrate concentrations found in peat from the PD site (§4.2.2), as well as the perfect for soil microorganisms C:N ratio of 25 in the middle and lower horizons as the site is nutrient rich (C:N<25 Maljanen et al, 2010). The nitrous oxide fluxes observed during this research at the PD plot are clearly lower (5-6 times) than fluxes measured in 2012 at the same site (van Asperen, 2015), which shows a reduction in N_2O emissions over time (Figure 38). In contrast, mean N_2O emissions from the rewetted and vegetated plots (38 and 45 µg m^{-2} h^{-1} respectively) are 3-10 times lower than observed at the drained areas, however, they are similar to those observed in the literature (Karki et al., 2015; Jordan et al., 2016). The above findings support the **hypotheses H1b,** that N_2O emissions from formerly extracted, bare peat soils are significantly lower after 5 years of rewetting; and **H2b,** that peat dams are hotspots of N_2O emissions.

The occurrence of a temporal trend in CO_2 and N_2O emissions was identified in the present study. At the peat dam, CO_2 and N_2O fluxes showed a clear reduction over the course of the study period, and were markedly lower than those observed in 2012 (van Asperen, 2015). A consistent decrease in the CO_2:N_2O emissions ratio over time was evident. While it is possible

that spatial variability caused part of the difference in the magnitude of observed fluxes between 2012 and 2014, the most likely reason for the observed decreasing trend in emissions at the peat dam is that this site has not yet reached 'equilibrium' in terms of its CO_2:N_2O emissions ratio. The bare peat dams were constructed in 2009, at which point conditions became favorable for decomposition (oxic), resulting in high CO_2 and N_2O fluxes. Over the years, it is likely that the amount of fresh readily available carbon and nitrogen has decreased, resulting in a subsequent reduction of CO_2 and N_2O fluxes, and explaining the large difference in observed flux magnitude over the years 2012, 2014, 2015 and 2016. Another possible explanation for the reduction in N_2O emissions over time could be the higher C:N ratio (40-46:1; Klemedtsson et al., 2005) which was measured in peat samples from the upper soil horizons. The C:N ratio in the studied peat samples from the topsoil is higher than 25:1, meaning that less nitrogen is available here and that the decomposition process in the upper 30 cm of peat is slower. The fact that the upper 28 cm of peat at the PD site is strongly humified might also support the above theory. The changing ratio of CO_2 to N_2O indicates that the timeframe of the 'equilibration process' may be different for the two gases. Moreover, the high amount of CO_2 produced and emitted to atmosphere during mineralization acts as an important index of the soil biological activity.

Table 16. A review of N_2O fluxes measured at natural, degraded and rewetted peatlands and this study. Values reported in the existing literature are compared alongside those found in the present study (Vybornova, 2017).

Type of peatland	Location	Annual N_2O emissions (kg N ha^{-1} a^{-1})	Study
Pristine bog	Germany	0.0 - 0.2	Beetz et al., 2013
	Poland	-0.05 – 0.2	Juszczak & Augustin, 2013
	Canada	0.0	Glatzel, 2016
Bog under extraction	Germany	1.3 ± 0.7	Beyer and Höper, 2015
	Boreal peat soils	2.0 - 2.2	Couwenberg, 2011
	Canada	0.0	Glatzel, 2016
	Finland	1.86	Maljanen et al., 2004
	Germany	1.65	Vanselow-Algan, 2014
	Germany	0.9-1.95	**Vybornova, 2017**
Bare peat	Sweden	0 – 3.68	Jordan et al., 2016
	Estonia	1.18	Järveoja et al., 2015
	Germany	1.46	**Vybornova, 2017**
Peat dam	Germany	12	**Vybornova, 2017**
Rewetted	Canada	0.0	Glatzel, 2016
	Sweden	0 – 1.2	Jordan et al., 2016
	Germany	1.1	**Vybornova, 2017**
Range	European bogs	0 – 11.9	Oertel et al., 2016
	German fens	5.3 – 14.0	Augustin et al., 1998

Summer fluxes observed at the ditch area were about one order of magnitude higher than the observed winter fluxes, and again lie within the range reported in the literature (Table 16). The hypothesis that ditches re-filled with peat are hot-spots of N_2O emissions at bare sites (**H3a**) can therefore be accepted. Lower N_2O fluxes at the rewetted sites are expected, and it is possible that very low emissions may remain undetected due to the chamber volume or short chamber closure time (Vanselow-Algan, 2014). In this study, almost 30% of all N_2O fluxes were below the detection limit of the Agilent gas analyzer and loss of gas concentration from syringes. Moreover, although the closed chamber method is simple to use and relatively inexpensive, it is well known that chamber-based N_2O fluxes estimates are likely to underestimate the annual flux. Previous research suggests that N_2O emissions are released in the form of pulses with very short peaks, taking mostly a few days or weeks (Stolk et al. 2009; Hastings et al. 2010). Hence it is likely that measurements with a frequency of less than one to two weeks will not correctly characterize emissions over the year. Barton et al. (2015) note that these large peaks mostly follow rewetting or spring-thaw events, which could contribute two-thirds of the total peak emissions. These findings are in agreement with the present study, where the observed N_2O fluxes showed a decreasing trend during drought periods, and peak emissions were observed after precipitation and spring-thaw events. On the other hand, nighttime N_2O fluxes are known to be lower than their daytime counterparts (Mosier et al., 1997; Van Asperen, 2015), especially during precipitation events and periods characterized by large temperature fluctuations (Maljanen et al., 2002). This difference is another possible cause for the flux overestimation in the present study, as only daytime gas fluxes (10 am - 2 pm) were investigated. The calculated N_2O fluxes therefore likely overestimate than the actual daily mean flux.

In their study of the Himmelmoor, Vanselow-Algan (2015) found that although N_2O fluxes from the extraction and rewetted sites in 2011 were negligible, the fluxes from the industrial extraction site (max 1.6 nmol m^{-2} s^{-1} during summertime) were significantly higher than observed at the rewetted site. Similar trends were identified during this study, whereby the extraction area emitted 14% and 56% more N_2O annually than the former extracted areas after 5 and 10 years rewetting, respectively. Research suggests that N_2O emissions may be of minor importance under anoxic conditions, but still possible denitrification may result in higher N_2 release, and hence in higher gaseous N losses under anaerobic conditions (Velty et al., 2007; Ussiri and Lal, 2013).

5.2.3 Controls on GHG fluxes

The temporal variation in GHG emissions identified for all study sites was driven mostly by soil temperature and changes in the water table level. In general, soil temperature (at 10 and 20 cm depth) was the abiotic control which was best able to explain variations in the examined gas fluxes. Non-linear relationships between these variables suggest that higher GHG fluxes can be expected under higher soil temperatures. Overall, it was identified that the CO_2 and N_2O fluxes were not significantly correlated to redox potential, although they correlated with water table level with a stronger negative relationship observed at the PD and D sites during the growing

season. In contrast, the CH_4 fluxes displayed no strong relationship with either WTL or air temperature. These results are consistent with previous studies, including those in the Himmelmoor, which suggest that for different microtopes on rewetted and drained plots, GHG emissions may most strongly depend on air and soil temperature, being less dependent on water table level (Parmetier et al., 2009; Vanselow-Algan, 2014; van Asperen 2015). For example, Muhr (2011) found no strong relationship between CO_2 fluxes and WTL in a nutrient-rich peatland in Germany, with some dependency identified only for the upper 15 cm of topsoil. In drained Finnish peatlands, Mäkiranta et al. (2009) were again unable to identify a significant relationship between WTL and R_{eco}, whereas their results do confirm the strong dependency of R_{eco} emissions on soil temperature discussed above. Recent studies have revealed the importance of soil temperature in the topsoil and soil surface, identifying these as the most important controls on GHG emissions (Kuzyakov and Gavrichkova, 2010). The authors note that soil temperature is especially important for CO_2 and CH_4 emissions because temperature affects assembled soil-biological processes, heterotrophic soil respiration, plant roots, and microbial biomass. Moreover, wintertime fluxes may contribute significantly to annual N_2O emissions in organic soils, mostly during freeze-thaw cycles (Koponen et al., 2006; Maljanen et al., 2007; Alm et al., 2007), as was also found on the ice-covered plots in the present study.

5.3 GHG production potentials from peat soils at the extraction and rewetted areas

5.3.1 Production potentials of carbon dioxide

The CO_2 production rate of control samples (oxic and anoxic incubations) from the extraction area was between two to four times lower than in the peat samples from the rewetted area. The results also showed that C mineralization rates under oxic conditions were up to five times higher for all samples when compared to anoxic conditions, as has been described previously in the literature. A number of authors studying northern ecosystems have found anoxic conditions to be associated with two to six times lower C emissions than observed under oxic incubations (Moore and Dalva, 1997; Glatzel et al., 2004; Schädel et al., 2016). The aerobic and anaerobic CO_2 production rates of samples from the rewetted site in this study amounted initially to 9.5 ± 1.9 and 2.1 ± 0.5 µmol CO_2 g C^{-1} d^{-1} (during first 2 months, see Fig. 44), respectively and slowed over time by 31 - 78 % during the final phase of incubation. Similarly, samples taken from different depths at the extraction site showed decrease in CO_2 production rates with time and were on average 50-70% lower than observed at the rewetted plot. Mineralization was fastest in the samples taken from initially anoxic soil horizons and subsequently incubated under oxic conditions, and vice versa, when samples from the upper oxic layer were incubated under anoxic conditions. The response in CO_2 production to the oxygen availability and to intense drying-rewetting is again similar to that found previously by other authors (Waddington et al., 2001; Borken et al., 2003; Moyano et al., 2013) and corresponds to suggestion that repeated fluctuations of the WTL and oxic-anoxic conditions

may contribute significantly to faster substrate mineralization compared to stable WTL position. Overall, these incubation study results are in agreement with ecosystem respiration dynamics for both study sites, which were calculated using chamber measurements in the field, as well as values reported previously for drained and rewetted bogs (Moore and Dalva, 1997; Beyer and Höper, 2015).

According to the estimated area and carbon density of the upper 100 cm of peat, the mean CO_2 production potential of the rewetted area under anoxic conditions amounts to 2.3 t CO_2 ha^{-1} yr^{-1}, whereas the extraction area has the potential to emit an average of 8.0 t CO_2 ha^{-1} yr^{-1} if mean WTL is no higher than -30 cm. Related to possible peat drying effects in the field, peat from both areas showed a significant increase in mean CO_2 production under oxic conditions of up to 29.5 and 39.6 t CO_2 ha^{-1} y^{-1} for rewetted and extraction sites, respectively. These production potentials are closely related to annual CO_2 emissions estimated for the dry peat dam plot, where the upper 60-70 cm of peat are situated above the range of annual fluctuations in WTL. The CO_2 production potentials estimated during this study (aerobe mineralization rates 0.04- 0.2; anaerobe 0.02- 0.04 mg CO_2 $g_{soil\text{-}dry}^{-1}$ d^{-1}) lie towards the lower range of that reported in other studies. For example, Glatzel et al. (2004) found that respiration activity was enhanced from a topsoil of revegetated and a rewetted site in Canada compared to recently restored areas. The authors report fluxes of 0.04 to 1.05 mg CO_2 g_{soil}^{-1} d^{-1} under aerobic conditions and 0.01 to 0.29 mg CO_2 g_{soil}^{-1} d^{-1} under anaerobic conditions; which are both up to five or six times higher than the fluxes calculated in the present study. Oxic incubations of peat samples from natural and drained peatlands in Canada demonstrate varying CO_2 production potentials (0.02 to 0.7 mg g^{-1} d^{-1}), and the undisturbed site being characterized by the highest values (Waddington et al., 2001). In the present study, the CO_2 production ratio between oxic and anoxic treatments remained constant throughout the experiment. The anoxic CO_2 production from peat soil at the rewetted area remained consistently closer to the rates produced by peat from the extraction area, whereas oxic incubated samples revealed significantly higher CO_2 production rates at the rewetted area than at the extraction area. One possible explanation for this result could be a stimulating effect of oxygen availability on the mineralization of more labile C in samples from the RV. Although the soil properties of both study sites differ, it was hypothesized that the vegetation cover and anoxic conditions at the RV site would be the two major factors determining the enhanced microbial respiration here. Previous research has confirmed that easily decomposable organic matter released from plant roots in the form of root exudates is immediately used by soil microorganisms and emitted as CO_2 during respiration (Estop-Aragonés and Blodau, 2012), hence affecting the soil microbial community and microbial biomass. Findings in the present Himmelmoor study support this observation, and enhanced microbial C and N concentrations were found in the rewetted area than in the drained plot. Moreover, the lower DOC and ammonium concentrations measured at the RV plot in comparison to extraction area may indicate a significant uptake of these nutrients by the vascular plant vegetation as has been reported previously for the Himmelmoor (Pfeiffer, 1998; Vanselow-Algan, 2014).

It is well known that peat mineralization occurs more rapidly under oxic rather than anoxic conditions (e.g. Gorham 1995; Brown 1998), and hence it was hypothesized that respiration in peat samples from the rewetted plot would be enhanced during oxic incubation due to the higher amount of organic substances available for mineralization. Evidence from the present study suggests that the high CO_2 production potentials of peat samples from the RV site, and hence the microbial activity, were triggered by the availability of fresh, easily decomposable organic substances and microorganisms. A possible explanation for this finding is that although the DOC- and nutrient content were lower at the RV site than at Site E, the substances released by plants in the form of root exudates (glucose, citrate, amino acids) play a more important role in determining microbial activity and enhance peat CO_2 production. Research suggests that the root exudates (also called rhizodeposits), which contain amino acids, carbohydrates, root tissue and fatty acids (Basiliko et al., 2012), may either significantly stimulate the rate of mineralization of soil organic C (Kuzyakov, 2002; Hamer and Marschner, 2002) or reduce it largely due to microbial consumption of the available labile substrate (Cheng, 1999). However, the importance of plant activity and the effect of vascular plant exudates on peat decomposition and microbial respiration rates in wetlands are still not sufficiently understood.

The impact of adding fresh organic substrate to the anoxic incubated samples from site E was an increase in CO_2 production in the second and lower horizons of the soil. This increase can be explained by the enhanced anaerobe activity of added methanogens as well as by possible anaerobic CH_4 oxidation coupled to denitrification (Raghoebarsing et al., 2006; Smemo and Yavitt, 2011). Unexpectedly, no significant change in CO_2 production were observed under oxic conditions after the addition of fresh filtrates in comparison to the control treatments. In fact, some inhibitory effect of filtrate addition on oxic CO_2 production was observed in the peat from the upper soil horizon. One possible explanation for why the additional fresh substance from the rewetted site did not trigger oxic soil respiration activity was that decomposition of the peat in the studied samples was paused due to the presence of fresh labile C substrates that also affected total respiration rates. Other explanations given at the beginning of the experiment were that these topsoil horizons from site E are limited in the amount of living microorganisms or that no living aerobe and anaerobe microorganisms were presented in either filtrate of the soil solution. However, as the incubation with replicates containing only 2 ml of non-sterile filtrate (designed to check for the presence of living microorganisms) showed a weak increase in aerobe and anaerobe CO_2 as well as CH_4 production during the whole incubation experiment, both of the above suggestions are rejected. Alternatively, it is possible that the microorganisms from the RV site, especially methanogenic archaea, could not adapt to the long-term oxic conditions. This suggestion is supported firstly by the decreasing ratio of microbial biomass measured at the beginning and the end of the 134-day incubation, and secondly by similar findings for peat samples from ombrotrophic bogs in Canada (Brown, 1998).

The role of fresh labile C on the CO_2 and CH_4 production potential in the extraction area is complex due to the linked processes of methanogenesis and methanotrophy. Despite this complexity, it was possible to examine this role by studying drying-rewetting events, whereby

the fresh organic substance is released through cell lysis due to osmotic shocks. During the two month experiment, mean aerobe CO_2 release from the samples affected by drying-rewetting stress were 26-63% higher than in the control experiment. Similar results have been reported previously for different soils in response to the drying-rewetting process, indicating a shift in microbial community structure and microbial adaptation to the moisture conditions found in the field (Fierer and Schimel, 2002; Ioveino and Baath, 2008). Generally, the anoxic incubations conducted in the present study revealed the significant effect of adding sterile filtrate (with labile C) to the control samples on CO_2 and CH_4 production from the initially anoxic horizons; however, no CO_2 production increase was measured in the topsoil. This can be explained by the highest population of methanogens in wetlands being situated mostly in the first 10 cm below the water table, as well as the high concentration of important methanogenic substrate contained in the DOC fraction (Liu et al., 2011). As the mean annual WTL in site E is situated below the surface (at -30 cm), it is possible that the strong oxic conditions in this upper peat layer caused a shift in microbial community structure, and that any subsequent stress will cause a reduction in the amount of microbial activity and biomass. Large amounts of microorganisms die during periods of drought (Wolf et al., 2010), and the released cell organic substances and nutrients could be mineralized immediately after rewetting due to the low C:N ratio which was observed (Van Gestel et al., 1993). In general, the incubation experiment has shown that peat extraction and drainage results in low microbial activity and low respiration rates at site E. **The hypotheses H5a and H5b**, regarding the dependency of CO_2 production potentials from a fresh, easily decomposable soil organic matter supply and soil microbial activity, are thus supported. However, some important aspects of the relationship between microbial metabolism and soil respiration in drained peat under dry oxic conditions remain unclear. For example in this study, although estimated laboratory peat respiration rates correlate with results from the field measurements; as the respiration measured at the RV site also contains the autotrophic respiration component by *Eriophorum*, these results are not comparable.

5.3.2 Production potentials of methane

The measurements of CH_4 produced during the 365-day anoxic incubation indicated a significant difference in gas production between peat samples from the extraction and rewetted sites. The anaerobic CH_4 production of all anoxic incubated samples was positively correlated with CO_2 production ($r = 0.88$, $p < 0.001$), with the highest correlation coefficients being observed in all samples from the RV site, indicating changes in the availability of easily decomposable C sources (Moore and Dalva, 1997; Estop-Aragones and Blodau, 2012). The suggestion that there is a lack of methanogens in the extraction area is supported by the low CH_4 production rates observed in the control samples compared to the high production rates observed in samples with added living microorganisms from the RV site. Recent studies have reported that peatland rewetting and the occurrence of vascular plants can affect the composition of methanogenic microorganisms and decrease peat DOC concentration in

comparison to bare areas or *Sphagnum*-dominated microtopes (Leroy et al., 2017). Similarly as seen with microbial respiration rates, Knoblauch et al. (2015) showed that areas vegetated by vascular plants in the polygonal tundra in northeast Siberia exhibit higher CH_4 emissions due to in situ CH_4 production in the rhizosphere of vascular plants, as well as the dominance of plant-mediated CH_4 transport. In agreement with previous studies, anoxic incubation experiments with samples from the Himmelmoor showed that CH_4 production rates were 1 to 2 order of magnitude faster in the samples from rewetted area than from extraction area (0.01-0.5 and to 2.6 µg $g_{soil\text{-}dry}^{-1}$ d^{-1} for the E and the RV area, respectively) and increased with time non-linearly. Production rates lie within the range reported by previous studies for northern peatlands (Moore and Dalva, 1997; Yavitt et al., 1997; Blodau and Moore; 2003). Previous literature for example, reported values of up to 1 µg CH_4 $g_{soil\text{-}dry}^{-1}$ d^{-1} for pristine mires, 0.1 µg CH_4 $g_{soil\text{-}dry}^{-1}$ d^{-1} for harvested areas, and 816 µg CH_4 $g_{soil\text{-}dry}^{-1}$ d^{-1} for inundated revegetated sites after several months of anoxic incubation (Glatzel et al., 2004). Hornibrook (unpublished) determined the CH_4 production potentials from different rewetted peat soils by anoxic incubation to be in the range of 0 to 18.4 µg $g_{soil\text{-}dry}^{-1}$ d^{-1}, with the highest potentials being identified in the wet natural area, which had a high root biomass compared to other sites and is directly related to findings reported in the present study.

In comparison to the upper horizon, the control samples from the middle and lowest horizons of site E did not show any significant CH_4 production, with only several low peaks in CH_4 production occurring throughout the incubation experiment. Additionally, in comparison to the control samples, the enhanced response in CH_4 production activity after the addition of both filtrates indicates that the most important factors affecting methanogenesis throughout the depth profile are anaerobic conditions and the amount of easily decomposable organic matter. The last one acts as a trigger for microbial activity and mainly for living methanogenic archaea. During oxic incubations no effect of filtrate addition on CH_4 release could be observed. Similarly as seen with CO_2 production, the anaerobe cumulative CH_4 production potentials of the rewetted site were at least ten times higher than observed in the extraction area, on average amounting to 4.7 t CH_4 ha^{-1} yr^{-1}. At the extraction area, CH_4 production was the fastest in the upper horizon and began much earlier in this layer compared to lower parts of the profile, indicating more decomposable organic matter quality. A higher organic matter quality was indicated by the higher potential CO_2 production there, and in the rewetted area additionally due to root and litter inputs close to the soil surface (Estop-Aragones and Blodau, 2012; Treat et al., 2015). These incubation study results are in agreement with values reported previously for northern bogs (Blodau and Moore, 2003).

The results reported in the present study have shown that both aerobic- and anaerobic CO_2 production as well as anaerobic CH_4 production were related to the degree of decomposition, with a high humification grade resulting in lower gas emissions. This finding is in agreement with Glatzel et al. (2004), who suggested that the humification grade may be a significant control of CO_2 and CH_4 production. In this respect, both **hypotheses H4a** and **H4b** - that aerobic and anaerobic microbial production potentials of CO_2 and CH_4 in the soil material from the peat

extraction sites are lower than in re-vegetated peatlands - are accepted, and the results indicate a significant relation between GHG release estimates from laboratory incubations and field-based measurements.

5.4 Estimation of annual GHG fluxes for the Himmelmoor and global peatlands

An estimation of the annual GHG fluxes for the Himmelmoor peatland was made based on the measurements from 2014-2015. Annual emissions were calculated from the mean fluxes measured during each year at the three to four replicate plots, and the annual CH_4 and N_2O fluxes were converted into CO_2-equivalents according to the Global Warming Potentials of both gases (34 and 298 respectively) (Myhre et al., 2013). The mean annual GHG emissions for a 1-hectare area were estimated as: 123 ± 43 kg CH_4, 3.01 ± 1.1 kg N_2O, and 2.4 ± 0.8 t CO_2 for rewetted areas; 130 ± 16 kg CH_4, 7.8 ± 3.3 kg N_2O, and 6.6 ± 1.4 t CO_2 for ditches; and 17.7 ± 10.8 kg CH_4, 20.6 ± 4.3 kg N_2O and 16.0 ± 2.9 t CO_2 for drained sites. In general, all studied areas of the peatland were found to act as GHG sources, with the total mean emissions over the two year period varying from 4.7 t CO_2-eq ha^{-1} yr^{-1} at the rewetted plot to 38.1 t CO_2-eq ha^{-1} yr^{-1} at the peat dam. A relationship of all annual GHG emission budgets estimated at the Himmelmoor study sites in terms of annual C and N losses is provided in Figures 53-54. Total GHG emissions for the study area were dominated by the CO_2 fluxes (50-74% for different sites); an observation which has been reported previously for many drained and rewetted bogs and fens (Shurpali et al., 2010; Elsgaard et al., 2012; Renou-Wilson et al., 2014; Eickenscheidt et al., 2015; Günther et al., 2015; Karki et al., 2016).

Very similar results for rewetted sites were described in Järveoja et al. (2015) who reported total GHG balances of 4.14 and 3.83 t CO_2-eq ha^{-1} yr^{-1} for Estonian bog after two years of rewetting with high and low WTL, respectively. However, their estimated balance for bare peat study site (10.21 t CO_2-eq ha^{-1} yr^{-1}) was much lower than the results obtained in the present study for the peat dam in the Himmelmoor. Karki et al. (2016) estimated annual CH_4 emissions for a fen peatland in Denmark to be 2.1 t CO_2-eq ha^{-1} yr^{-1}. This value reveals disparity in fluxes from different European peatlands, as it is almost twice as high as methane emissions estimated for the rewetted R site in the Himmelmoor.

In contrast to the rewetted Site R, annual GHG fluxes measured on the peat dam are clearly larger than values reported for other drained and harvested areas (with fluxes of up to 22.6 and 31.4 t CO_2-eq ha^{-1} yr^{-1}, respectively) and on the data from 100 studies in Nordic countries (Maljanen et al, 2010).

Size of the former drained area in the Himmelmoor: 142 ha

Size of the studied area: 74 ha

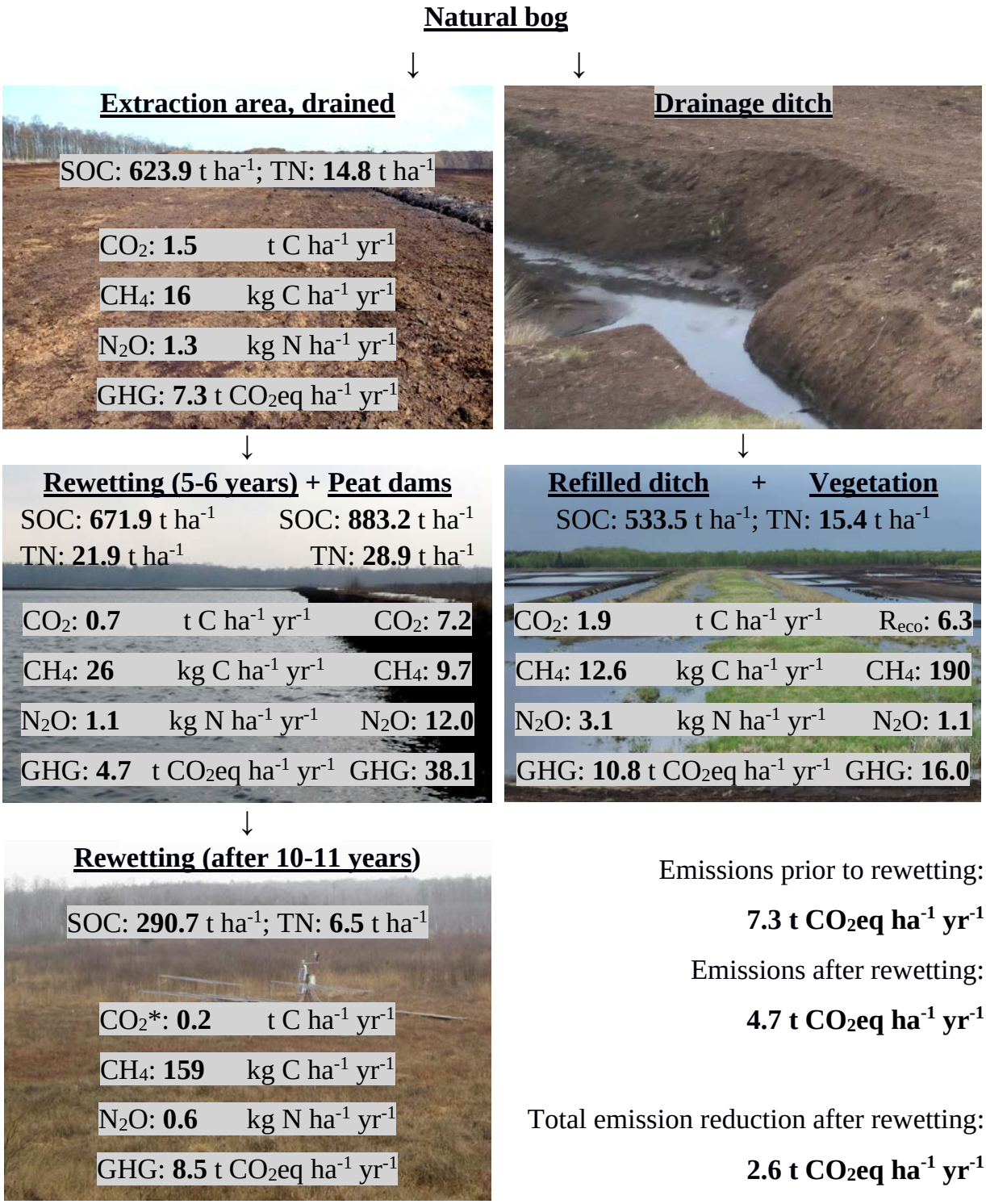


Figure 53. Carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O) fluxes from differently managed microtopes in the Himmelmoor, as well as their SOC and SON pools estimated for the upper 1 m depth. The CO_2* emission value for the rewetted RV area was taken from Vanselow-Algan (2014). It was assumed that gross primary production for bare areas was negligible, and that NEE was equal to R_{eco}.

The results of this study have shown that after only a five-year period, the strong warming impact of GHG emissions from the E area was reduced by more than 36% as a result of the rewetting process. The estimated mean annual GHG for the whole former extraction area and for the investigated area (weighted by area in ha) amounted to 10.5 and 13.9 t CO_2-eq ha^{-1} yr^{-1}, respectively. This is equivalent to 1029 t CO_2-eq yr^{-1} according to total studied microtopes area of 74 ha. Based on these findings, the **hypothesis H1d** that peatland rewetting leads to a significantly reduced (although still positive) radiative forcing effect due to GHG emissions is therefore accepted. It has also been proved that peat dams strongly diminish the overall climate mitigation potential of rewetting in the first years of the project (**Hypothesis H2c**).

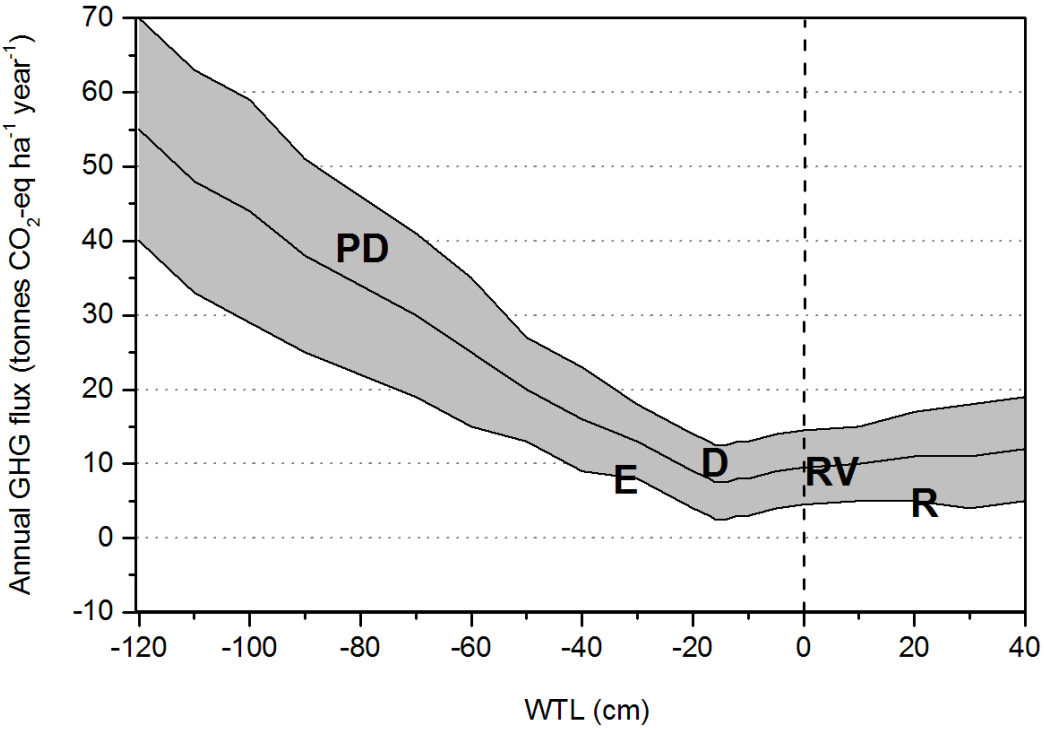


Figure 54. The relation between mean annual water table level (cm) and annual GHG emissions converted with GWP (tonnes CO_2 equivalents over a 100 year period, t CO_2-eq ha^{-2} y^{-1}) for all study sites. The total mean GHG emissions, which combine the CO_2, N_2O and CH_4 fluxes, vary from 4.7 to 38.1 t CO_2-eq ha^{-1} y^{-1} for the rewetted R and dry PD sites respectively. Abbreviations in black show mark the location of the study sites: PD – peat dam; D – ditch; RV – an area with vegetation, rewetted in 2004; E – extraction area; R – rewetted in 2009. The grey area indicates averaged GHG emissions reported for different drained, managed and rewetted peatlands (Couwenberg et al., 2008; Hooijer et al., 2012; Barthelmes et al., 2015; Jauhiainen et al., 2016).

6 Conclusions and recommendations

This study presents the results of field measurements from the Himmelmoor peatland and laboratory incubations, with the aim of examining overall GHG fluxes at the site and comparing fluxes from drained and “rewetted” peat areas. New findings on GHG emissions from dry peat dams are presented for different typical land use types in degraded peatlands under restoration, the properties and GHG fluxes, which are not well covered by the published literature. It was found that all five study sites (peat dam, ditch refilled with peat, extraction area, area rewetted in 2009 and in 2004 with vegetation) differed considerably in their soil-physical and -chemical properties, SOC and total nitrogen, water table level, and vegetation cover. These differences reflect the former management types at the various sites and show that after the first ten years of flooding, peat soils remain strongly affected by the previous environmental conditions under drainage and extraction.

All study sites were found to emit significant amounts of greenhouse gases to the atmosphere on an annual basis, and in terms of radiative forcing, it was found that the flooded areas have the weakest warming effect, whereas drained and dry peat dam areas showed the strongest warming effect on the climate. It is suggested that during restoration activities, especially those involving peat dams, as it was the case in the Himmelmoor, the negative effect of GHG fluxes on climate can be mitigated by a number of cost-effective actions such as reducing the height of dams, or regulating a permanent water table level close to the soil surface to prevent peat oxidation. The hypotheses that were formulated at the beginning of the study and discussed above can now be answered as follows:

Hypothesis 1. Five years of re-wetting has resulted in 53% lower CO_2 and 16% lower N_2O emissions from formerly extracted bare peat soils, whereas CH_4 emissions were found to have doubled. In fact, the specific “rewetting” measures implemented in the Himmelmoor serve as a method for reducing GHG emissions relatively quickly, and led to a 36% reduction GHG emissions, although the overall warming effect remains positive.

Hypothesis 2. Peat dams are hotspots of CO_2 and N_2O emissions, and strongly diminish the overall climate mitigation potential of rewetting actions during the first years of restoration. Dry peat areas, including the top of the dams, should be reduced in height to prevent erosion and oxidation of the peat material as well as to reduce GHG emissions.

Hypothesis 3. Ditches re-filled with peat are hotspots of N_2O at bare sites, and hotspots of CH_4 at sites vegetated by vascular plants. However, overall the total GHG flux from both sites decreased between the 2014 and 2015 study years.

Hypothesis 4-5. Both aerobic and anaerobic CO_2 und CH_4 production potentials of recently abandoned but still drained peat extraction sites are low compared to restored and re-vegetated peatlands under rewetting. This observation is due to the limited fresh and easily decomposable soil organic matter supply and low soil microbial activity at the extraction site.

This study has found that the creation of peat dams as part of a rewetting project can cause a significant increase in the overall GHG fluxes of rewetted peatlands, which can affect the GHG balance of a site for several years. For this reason, it is recommended that current rewetting practices be adjusted in the form of improved water saturation and the reduced aeration of peat dams. In order to prevent the loss of organic matter from the peatland, any remaining drainage ditches should be closed. As mentioned before, mitigation of GHG emissions can be achieved by the recolonization of natural bog vegetation and the reduction of water losses. Natural re-colonization of *Eriophorum spec.* has been occurring since the 1980s in refilled ditches in the Himmelmoor, however this process is complex and slow as described previously (Stewart and Lance, 1991; Van Seters and Price, 2001). Furthermore, without water management practices such as flooding and decreasing of water fluctuations, Sphagnum may never begin to re-colonize. In light of this information, it is recommended that the dry bare peat in the Himmelmoor should be rewetted and covered using bryophyte vegetation in order to prevent further oxidization of the peat material. The so-called Sphagnum farming or peat moss cultivation, which has been used for last two decades in Germany is a promising method offering the fast establishment of Sphagnum on degraded peatlands (Gaudig et al., 2014; Muster et al., 2015). Once *Sphagnum*, the most important species for the biological and structural restoration of peatlands, has established in the peatland, the moss community starts to develop and water can be retained in the soil.

Overall, this study concludes that, although rewetting reduces GHG emissions, the full restoration of soil conditions can take at least decades to occur. It is recommended, however, that restoration managers who employ peatland rewetting to limit N_2O and CO_2 losses should also consider the increased CH_4 emissions associated with this practice. Future assessments of the climate mitigation potential of bog rewetting must consider all GHG gases, as well as their high spatial and temporal variability. The current knowledge-gap in understanding of the soil-microbial processes, microbial biota and community in degraded and restored peatlands provides reason to investigate the effect of peat conditions and former human activity on the spatial distribution of C and N losses and the shift in structure of the microbial community.

The raw datasets generated and analyzed during this study are available at doi:10.1594/PANGAEA.872182 (Vybornova, 2017).

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