Impact of summer drought on greenhouse gas fluxes and nitrogen availability in a restored bog ecosystem with differing plant communities

Dissertation zur Erlangung des Doktorgrades
der Naturwissenschaften im Fachbereich Geowissenschaften
der Universität Hamburg

vorgelegt von
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Hamburg 2014

Als Dissertation angenommen vom Fachbereich Geowissenschaften der Universität Hamburg auf Grund der Gutachten von Prof. Dr. Eva-Maria Pfeiffer
und Prof. Dr. Lars Kutzbach

Hamburg, den 29.01.2014

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Summary

Natural peatlands are important carbon sinks and potential sources of methane (CH_4). In contrast drained peatlands turn from a carbon sink to a carbon source and can emit nitrous oxide (N_2O). Therefore restoration of peatlands implies climate change mitigation. In addition, climate change is expected to have a significant impact on the peatland carbon store and changes in precipitation amount and frequency are potentially damaging for peatlands. Thus, the carbon dioxide (CO_2) and greenhouse gas balance (GHG) balance of three different plant communities at a restored bog site was investigated and the impact of summer drought was estimated by a manipulative field experiment using rainout shelters.

As study site, the Himmelmoor in the metropolitan region of Hamburg was chosen which is one of the largest ombrotrophic bogs in Schleswig-Holstein and is still used for peat mining. At the peat mining area and in the restored boundary fluxes of CO_2, CH_4 and N_2O were monitored with closed chambers, for a one year period, with and without drought treatment. This study thus presents the first annual estimates of the CO_2 and the GHG balance for parts of the Himmelmoor. In addition, nitrogen availability and meteorological and hydrological measurements were collected as well.

All three vegetation communities established in the restored study site, as well as the industrial peat mining area turned out to be sources of CO_2 ranging between 0.60 ± 1.43 t CO_2 ha^{-1} $year^{-1}$ (*Sphagnum* dominated vegetation) and 3.09 ± 3.86 t CO_2 ha^{-1} $year^{-1}$ (vegetation dominated by heath plants) in the restored area. The industrial peat mining area had with 7.30 ± 0.67 t CO_2 ha^{-1} $year^{-1}$ the greatest CO_2 emissions which were approximately 15 times higher, when the C-content of the extracted peat was included in the calculation (114.02 ± 6.70 t CO_2 ha^{-1} $year^{-1}$).

While accounting for the different global worming potentials (GWP) of the three greenhouse gases, the annual GHG emissions at the restored area ranged between 19 and 40 t CO_2-eq ha^{-1} $year^{-1}$ and were dominated by huge emissions of CH_4 (15 to 37 t CO_2-eq ha^{-1} $year^{-1}$) while N_2O emissions did not play a significant role. These CH_4 emissions are the highest emissions so far reported for bog ecosystems in temperate Europe. The highest annual emission rates were found at the site that was dominated by dense stands of purple moor grass *(Molinia caerulea)*. As the restored area was subject to large fluctuations in water table it can be expected that these high CH_4 emissions were caused by a combination of both the plant mediated transport through *M. caerulea* and the temporal inundation of the easy decomposable plant litter of this grass species.

The annual GHG balance of the peat mining area (8.90 ± 1.06 t CO_2-eq ha^{-1} $year^{-1}$) mainly derived from CO_2 emissions. While CH_4 emissions did not contribute to the GHG budget of this site, significant amounts of N_2O were detected in summer month resulting in annual fluxes of 1.54 ± 0.80 t CO_2-eq ha^{-1} $year^{-1}$. Again including the C-content of the extracted peat the GHG balance of the peat mining area is with 122.92 ± 6.78 t CO_2-eq ha^{-1} $year^{-1}$ 3 to 6 times higher than the GHG emissions of the restored area.

Summer drought had unequal effects on the vegetation communities of the restored area. The nitrogen availability in the soil pore water significantly increased due to rainout treatment indicating an increase in peat mineralisation. The GHG emissions of the *Sphagnum* and the heath plant dominated subsite slightly increased due to the rain exclosure, while it decreased at the site dominated by *M. caerulea.* Here, a considerable reduction of CH_4 emissions outweighs the increase in CO_2 emissions. *M. caerulea* additionally benefited from drought, as it significantly increased its biomass within one year. It is thus of concern that *M. caerulea* will spread out due to climate change and outcompetes typical bog plants.

With regards to the extraordinary high CH_4 emissions, and the situation that none of the plant communities which established at the restored site acts as CO_2 sink suggestions for the improvement of the restoration and for adaptation to climate change were developed, based on the obtained results. As these suggestions are transferable, this study provides valuable implications not only for the Himmelmoor but also for other bog ecosystems in temperate Europe facing the challenges of climate change.

Zusammenfassung

Während ursprüngliche Moore signifikante Senken für Kohlendioxid (CO_2) und natürliche Quellen von Methan (CH_4) darstellen, wandeln sich antropogen genutzte Moore von Kohlenstoffsenken zu Kohlenstoffquellen und emittieren Lachgas (N_2O). Daher ist die Renaturierung von Mooren ein wichtiger Beitrag zum Klimaschutz. Darüber hinaus wird erwartet, dass der Klimawandel erhebliche Auswirkungen auf die CO_2-Senkenfunktion von Mooren hat und sie durch Veränderungen in Niederschlagsmenge und -frequenz negativ beeinflusst werden können. Aus diesem Grund wurde die CO_2- und Treibhausgasbilanz (THG-Bilanz) von drei verschiedenen Pflanzengemeinschaften einer restaurierten Hochmoorfläche ermittelt und die Auswirkungen der für die Region Norddeutschland prognostizierten Sommertrockenheit durch ein manipulatives Feldexperiment abgeschätzt.

Innerhalb der Metropolregion Hamburg wurde das Himmelmoor als Untersuchungsgebiet ausgewählt. Es handelt sich hierbei um eines der größten Hochmoore Schleswig-Holsteins und wird noch immer für die Torfgewinnung genutzt. Auf der Torfabbaufläche, sowie in einem bereits renaturierten Randbereich des Moores wurden CO_2-, CH_4- und N_2O-Flüsse mit und ohne Einfluss von simulierter Sommertrockenheit mit Hilfe der Hauben-Messtechnik gemessen. Darüber hinaus wurden die Stickstoffverfügbarkeit sowie meteorologische und hydrologischen Daten erhoben.

Alle drei Vegetationstypen der renaturierten Fläche sowie die industriell genutzte Torfabbauflache stellen CO_2-Quellen dar. Die jährlichen CO_2-Emissionen der renaturierten Fläche liegen zwischen 0,60 ± 1,43 t CO_2 ha^{-1} $Jahr^{-1}$ (von Sphagnen dominierte Vegetation) und 3,09 ± 3,86 t CO_2 ha^{-1} $Jahr^{-1}$ (von Heidepflanzen dominiert). Die höchsten jährlichen Emissionen wurden mit 7,30 ± 0,67 t CO_2 ha^{-1} $Jahr^{-1}$ auf der industriellen Abbaufläche gemessen. Schließt man die Menge an Kohlenstoff mit ein, die durch den Abbau an Torf entnommen wird, ist die Emission mit 114,02 ± 6,70 t CO_2 ha^{-1} $Jahr^{-1}$ ca. 15-mal höher.

Unter Berücksichtigung der unterschiedlichen Klimawirksamkeit von CO_2, CH_4 und N_2O liegten die jährlichen THG-Emisionen des renaturierten Bereichs zwischen 19 und 40 t CO_2 Äquivalent ha^{-1} $Jahr^{-1}$ hauptsächlich verursacht durch große Methanemissionen von 15 bis 37 t CO_2-eq ha^{-1} $Jahr^{-1}$. Diese CH_4-Emissionensraten sind die höchsten die im Europa der gemäßigten Breiten bisher in einem Hochmoor dokumentiert wurden. Am meisten emittierte dabei die Fläche, die von dichten Beständen an Pfeifengras (*Molinia caerulea*) dominiert wurden.

Da die renaturierte Fläche starken Wasserstandsschwankungen unterlag, ist zu vermuten, dass eine Kombination aus pflanzenvermitteltem Methantransport durch *M. caerulea* und des zeitweiligen Überstaus der Bodenoberfläche und damit der leicht zersetzbaren Streu diese Grases, die Ursache für diese extrem hohen Methanemissionen ist.

Die jährliche THG-Bilanz der Torfabbaufläche (8,90 ± 1,06 t CO_2-eq ha^{-1} $Jahr^{-1}$) setzt sich vor allem aus CO_2-Emissionen zusammen. Während CH_4-Emissionen nicht zur Treibhausgasemission dieser Fläche beitragen, konnten N_2O-Emissionen während der Sommermonate nachgewiesen werden und tragen signifikant zur Jahresbilanz bei (1,54 ± 0,80 t CO_2-eq ha^{-1} $Jahr^{-1}$). Schließt man wiederum die abgebaute Menge an Torf in die Berechnung mit ein, ist die THG-Bilanz der Abbaufläche mit 122,92 ± 6,78 t CO_2-eq ha^{-1} $Jahr^{-1}$ drei- bis sechsmalmal höher als die der renaturierten Fläche.

Die simulierte Sommertrockenheit wirkte sich in unterschiedlicher Weise auf die verschiedenen Vegetationsgemeinschaften des renaturierten Bereichs aus. Die Stickstoffverfügbarkeit im Porenwasser hat sich durch die Trockenheit erhöht was als Zunahme der Torf-Mineralisierung gedeutet werden kann. Die THG-Emissionen der von Torfmoosen und der von Heide dominierten Untersuchungsflächen nahmen aufgrund des Regenausschluss-Experiments zu, während es sich bei der von *M. caerulea* dominiert Fläche verringerte. Obwohl hier die CO_2-Emissionen anstiegen, wurde dieser Effekt durch die deutliche Reduzierung der CH_4-Emissionen überwogen. Es konnte außerdem gezeigt werden, dass *M. caerulea* von der induzierten Trockenheit, vermutlich aufgrund der erhöhten Nährstoffverfügbarkeit, profitiert hat. Dies ist durch einen signifikanten Biomasseanstieg innerhalb eines Jahres ableitbar. Es ist daher zu befürchten, dass sich *M. caerulea* in Zukunft durch die Folgen des Klimawandels ausbreiten und andere hochmoortypische Arten verdrängen kann.

Im Hinblick auf die außergewöhnlich hohen CH_4-Emissionen der renaturierten Fläche, und der Tatsache, dass sich keine der drei untersuchten Pflanzengesellschaften zu einer CO_2-Senke entwickelt hat, wurden Verbesserungsvorschläge für die Renaturierungsmaßnamen und zur Anpassung an den Klimawandel, auf Basis der vorliegenden Daten, erarbeitet. Aufgrund der Übertragbarkeit, liefert diese Studie wertvolle Schlussfolgerung nicht nur für das Himmelmoor, sondern auch für andere ehemals genutzte Hochmoore in den gemäßigten Breiten.

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Abbreviations

C/N	Carbon nitrogen ratio
DIN	Dissolved inorganic nitrogen
DON	Dissolved organic nitrogen
E	Experimental subsite at the industrial extraction site
GHG	Greenhouse gas
GPP	Gross primary production
Gt	Giga tons
GWP	Global warming potential
H	Experimental subsite dominated by heath plant species
HPLC	High performance liquid chromatography
IRGA	Infrared gas analyser
NEE	Net ecosystem exchange
P	Experimental subsite dominated by purple moor grass
PAR	Photosynthetic active radiation
Pmax	Maximum photosynthetic rate (light saturation point)
ppb	Parts per billion (10^{-9})
ppm	Parts per million (10^{-6})
Reco	Ecosystem respiration
rpm	Rounds per minute
S	Experimental subsite dominated by *Sphagnum* plant species
TDN	Total dissolved nitrogen

1. Introduction

Covering only 3 % of the Earth´s land surface, peatlands store as much carbon as all terrestrial biomass and twice as much as all global forest biomass (Parish et al., 2008).
Today, merely 1 % of the former extent of European peatlands has still living mire vegetation and accumulates peat (Koster, 2005). Worldwide an area of 80 million ha of peatlands have been destroyed due to peat mining for fuel and horticulture but mostly (50 %) by drainage for agricultural use (Joosten, 2006). While growing mires have a cooling effect on the climate by acting as a carbon sink (despite natural methane emissions), degraded peatlands are a major and growing source of the greenhouse gases (GHG) carbon dioxide (CO_2) and nitrous oxide (N_2O). As peat-based GHG emissions are substantial, they are reported in national GHG inventories under the United Nations Framework Convention on Climate Change (UNFCCC).

CO_2, CH_4 and N_2O are considered to be the principal contributors to the anthropogenic positive atmospheric radiative forcing. The global atmospheric concentration of these GHG has significantly increased from a pre-industrial level: CO_2 from about 280 ppm to 379 ppm, CH_4 from about 715 ppb to 1774 ppb and N_2O from approx. 270 ppb to 319 ppb until 2005. The global increase in the CO_2 concentration is primarily attributed to the combustion of fossil fuels and land use change, while those of CH_4 and N_2O are mainly due to agriculture (IPCC, 2007).

CO_2 emissions from degenerated peatlands are estimated to be equivalent to more than 10 % of the global fossil fuel emissions (Parish et al., 2008). Even in an industrialized country, like Germany, the estimated peatland GHG exchange accounts for 2.3-4.5 % of the anthropogenic emissions (Drösler et al., 2008). Therefore restoration of peatlands implies not only a recovery of ecosystem functions and biodiversity, but also climate protection (Drösler et al., 2009). Rewetting of drained peat soils as climate change mitigation measure presents a new challenge (Erwin, 2009; Couwenberg, 2009a): In Germany a potential reduction of 35 million tons carbon dioxide equivalents per year is possible by peatland restoration, which is a cost-effective mitigation strategy (Joosten, 2006; Drösler et al., 2009; Freibauer et al., 2009). However, data about the C-balance of restored and abandoned peatlands is scarce on a national and global scale and is urgently needed (Yli-Petäys et al., 2007; Drösler et al., 2008; Artz et al., 2013).

In the future, restoration and management of peatlands will be more complex and challenging due to climate change (Erwin, 2009; Grand-Clement et al., 2013). Climate change scenarios suggest e.g. changes in temperature and precipitation amount and frequency. The regional climate Model (REMO) predicts a reduction in summer precipitation by a maximum of 30 % for northern Germany (scenario A1B, (Umweltbundesamt, 2006)). This climatic change is expected to have significant impacts on the peatland carbon store and GHG-fluxes (Parish et al., 2008) and there is considerable uncertainty about the fate of peatland carbon in the future (Reay et al., 2008; Charman et al., 2013). Thus a profound knowledge of peat soil processes will help adaptation of management practices and in turn in stabilisation of ecosystem functions (e.g. carbon sink function) of peatlands against new environmental threats.

This study is part of the research topic “sustainable and cultivated environment” within the climate research project KLIMZUG-Nord. It provides annual flux estimates of the three main greenhouse gases CO_2, CH_4 and N_2O from a restored cutaway peatland (Himmelmoor, Hamburg metropolitan region) in 2011, supplemented with a manipulative field experiment, pore water analyses and an incubation experiment. Three restored sites differing in plant communities (dominating species: 1.*Molinia caerulea*, 2.*Sphagnum* spec. and 3.Ericaceous shrubs) were studied. The predicted summer drought due to a decrease in precipitation was simulated by a manipulative field experiment using rainout shelters.

This approach is of outstanding relevance for the Himmelmoor, since peat mining will be ceased until 2020 at an area of 70 ha in the central part of the peatland and will be successively restored. In conclusion the results will offer recommendations for an integrated peatland restoration facing the challenges of climate change.

The following main hypotheses, deduced from the scientific background presented in the following chapter, were addressed:

GHG emissions and evaluation of restoration success and potentials

Hypothesis 1) The restored cutaway peat site acts as a CO_2 sink, while the industrial peat extraction area is a large source for CO_2.

Hypothesis 2) CO_2 fluxes depend on the plant community and thus vary between the three subsites.

Hypothesis 3) CH_4 fluxes differ between experimental sites due to different plant communities and show no significant fluxes on the industrial peat extraction area.

Hypothesis 4) N_2O fluxes appear on the industrial peat extraction area only, no N_2O fluxes are found at the vegetated sites.

Increased summer drought is supposed to:

Hypothesis 5) Alter the CO_2 balance of the three vegetated sites, while the impact of the drought is different depending on plant community.

Hypothesis 6) Reduce CH_4 emissions but increase N_2O emissions.

Hypothesis 7) Enhance peat mineralisation detectable by increased concentrations of organic and inorganic dissolved nitrogen in the peat pore water and increased ecosystem respiration.

2. Background

2.1 Peatland ecosystems

Peatlands are wetlands characterised by the accumulation of incomplete decomposed plant litter called peat, leading to huge amounts of carbon stored in the peat soils (approx. 550 Gt carbon worldwide (Parish et al., 2008)). Decay of plant litter is limited due to water saturated, anoxic conditions and optionally due to low temperatures, as peatlands are mainly distributed in boreal and subarctic regions (Figure 1).

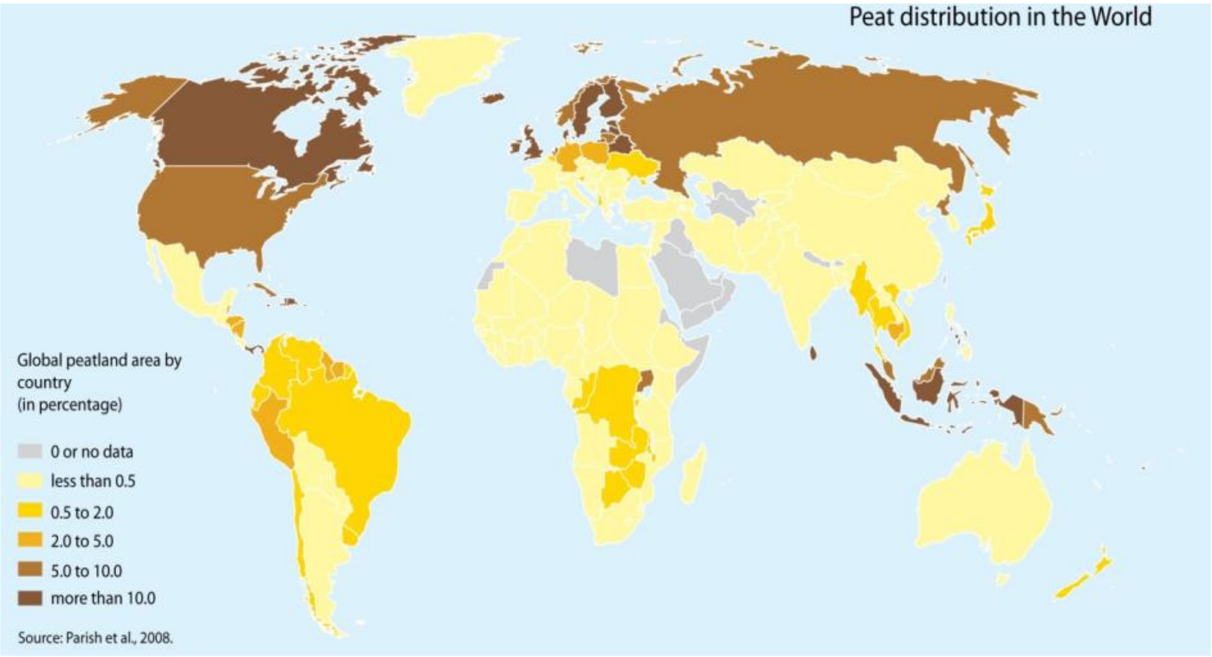


Figure 1: Percentage of the area covered with peatland per country (Parish et al., 2008).

The surface of peatlands rises continuously upwards since plants grow further on the sedentary deposits. The long term peat accumulation rate is about 0.5-1 mm per year which is equivalent to 10-40 t C per km^2 (Parish et al., 2008).

According to the source of water surplus, peatlands can be classified in ombrotrophic (rainwater-fed) bogs and minerotrophic (ground or surface water fed) fens. As this study focuses on a raised bog, this type of peatland is described more detailed in the following.

Raised bogs are distributed in areas where the amount of rainfall increases the losses of water due to evaporation and runoff. They are typically dome shaped because the bog continues to accumulate peat so that the surface elevation increases, especially in the centre forming a typically treeless landscape with a hummock-hollow surface structure. Due to rainwater supply and their dome shape, bogs are hydrologically isolated from the surrounding landscape and are generally nutrient poor (N-limited) and acidic. As peatlands in the present are faced high atmospherically nitrogen depositions there is evidence for the N-limitation shifting to a P-limitation (Lund et al., 2009). Typical pH values in bogs range between 3.3 and 4.5 (Sjörs, 1950; Clymo, 1964). Plant species growing in this special ecosystem are adapted to the waterlogged, acidic, nutrient poor and exposed conditions. These are for example ericaceous shrubs (e.g. *Andromeda polifolia*, *Calluna vulgaris* or *Vaccinium oxycoccos)* and cyperaceous graminoids *(e.g. Eriophorum vaginatum or s*edges) while the dominating and main peat forming species are mosses of the genus *Sphagnum*. These specialised mosses acidify their surrounding by donating H^+ Ions for taking up nutrients out of the soil water (Clymo, 1964; Brehm, 1968). With their high cation exchange capacity *Sphagnum* thus fosters unfavourable conditions for competing plant species. Additionally *Sphagnum* has a high nutrient use efficiency resulting in a competitive advance over vascular plants (Malmer et al., 1994). Peat formation takes place within the acrotelm, which is the surface layer of two distinct layers of a growing peat bog. It is characterised by high hydraulic conductivity and microbial activity, is rich in aerobic bacteria and contains the oscillating water table. In contrast, the lower layer called catotelm contains only dead plant material and has permanently anoxic conditions, possesses a small hydraulic conductivity and is poor in microbes (Ingram, 1978; Moore, 1989; Morris et al., 2011).

2.2 Greenhouse gas emissions from peat soils

Three of the six greenhouse gases (GHG) listed in the Kyoto Protocol play an important role in the nutrient cycles of peatlands: carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O). Natural peatlands act as carbon sinks and emit methane. When peatlands are drained, CH_4 emissions are significantly reduced, but CO_2 emissions increase and additionally N_2O can be emitted (Figure 2). The processes leading to theses emissions in natural and disturbed peatlands are described in the following.

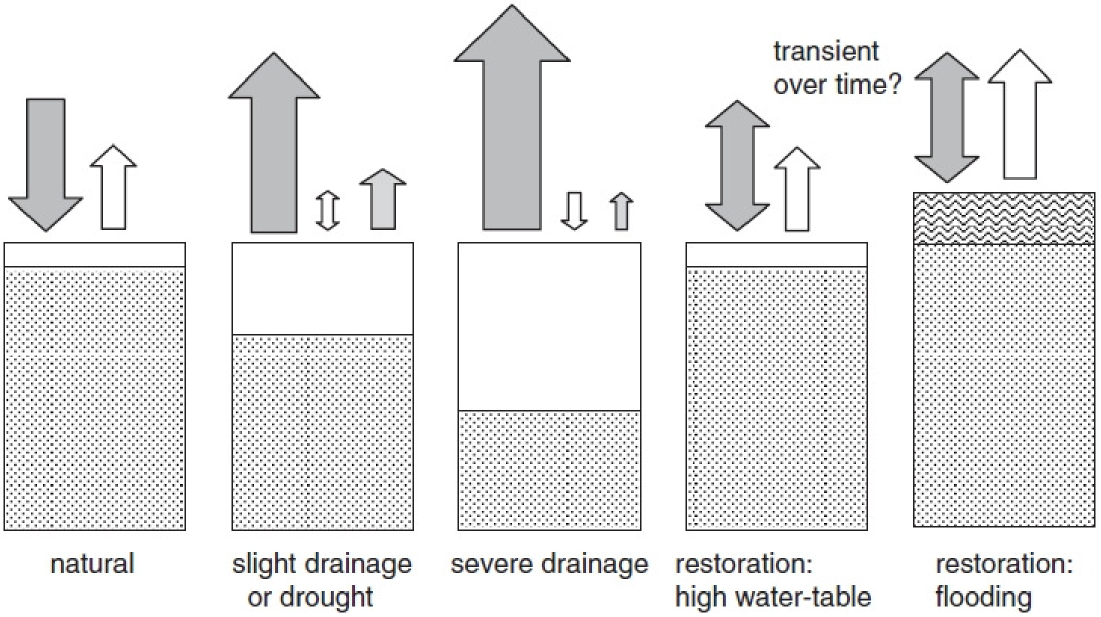


Figure 2: Schematic comparison of flux directions and relative magnitudes of Net-CO_2 (dark grey), CH_4 (white) and N_2O (light grey) fluxes in peatlands. Dotted areas symbolize the water saturated part of the profile. Waves in last panel illustrate a flooded soil

2.2.1 CO_2

Between 1850 and 2005 CO_2 increases by human activities have caused the largest radiative forcing among all greenhouse gases. The average rate of increase in atmospheric CO_2 over the period 1960 to 2005 is 1.4 ppm yr^{-1}, while the highest average growth rate was recorded in the 10 years from 1995 to 2005 by about 19 ppm. These increases are mainly due the combustion of fossil fuels, cement production and land use changes such as deforestation (IPCC, 2007).

Natural peatlands represent a CO_2 sink. The current annual carbon storage rate in the world's peatlands is approximately 100 million tonnes (Mt), which is equivalent to approx. 370 Mt CO_2 yr^{-} (Strack, 2008). It was observed by (Turunen et al., 2002) that the long term carbon accumulation rate (LORCA) in boreal and subarctic regions shows great variations between mire types and was higher in bogs (20,8 g m^{-2} yr^{-1}) then in fens (16,9 g m^{-2} yr^{-1}).
Carbon dioxide is fixed by plants via the process of photosynthesis and incorporated into plant biomass (Gross primary production, GPP). Autotrophic respiration of plants returns some CO_2 to the atmosphere. CO_2 is also respired heterotrophically by soil biota such as microbes decomposing organic matter. Together, heterotrophic and autotrophic respiration is called ecosystem respiration (Reco). Thus, the total net ecosystem exchange for CO_2 (NEE) is the balance between CO_2 uptake by GPP and CO_2 losses by Reco. In contrast, the net primary productivity (NPP) is GPP minus the autotrophic respiration only.

Net ecosystem exchange (NEE) can undergo large annual variations and is very sensitive to variations in environmental factors e.g. water level or temperature (Alm et al., 1999; Tuittila et al., 1999). Differences in NEE between wet and dry years can be attributed primarily to increased respiration (Bubier et al., 2003). Frolking et al. (1998) considered data from peatlands in North America and Europe and determined that at high light levels NEE was less at bogs than at rich and poor fens. Some reported NEE fluxes of CO_2 in natural peatlands are listed in Table 1.

Table 1: Reported NEE-CO_2 fluxes in natural peatlands, negative fluxes represent CO_2 uptake by the ecosystem, (Strack, 2008), modified.

Peatland type and location	NEE (g CO_2 m^{-2} yr^{-1})	Reference
Boreal raised bog		
Canada	-7 to -411	(Roulet et al., 2007)
Sweden	-7 to -37	(Waddington and Roulet, 2000)
Siberia	-79 to 132	(Arneth et al., 2002)
Temperate blanket bog (Ireland)	-179 to -223	(Sottocornola and Kiely, 2005)
Boreal oligotrophic pine fen (Finland)	-359	(Alm et al., 1997)
Subarctic palsa mire (Finland)	-135 to -509	(Nykänen et al., 2003)

Gross primary production (GPP) is related to the vegetation species and life forms present at that location, which is driven by the nutrient status and hydrology of the site. Weltzin et al. (2000) reported that rates of plant production of shrubs were higher under drier conditions while production of sedges and bryophytes was higher under wetter conditions. This is in correspondence to the findings of Bubier et al. (2003) who observed nearly no decline in photosynthesis of ericaceous shrubs in a dry year, while sedge dominated communities had lower rates over the whole growing season showing early senescence under drought conditions.

Decomposition and thus ecosystem respiration (Reco), is directly affected by temperature through its influence on microbial activity (Rowson et al., 2013). Frolking et al. (1998) reported that bogs have lower ecosystem respiration than fens. Reco is additionally dependent on the amount of labile or easily decomposable material. Root respiration accounts for up to 45 % of Reco and follow the phenology of the vegetation through the turnover of fine root litter and root exudates (Silvola et al., 1996; Rowson et al., 2013). The second main source of easily decomposable carbon is leaf litter (Schaefer et al., 2009).

In degenerated peatlands the CO_2 balance is altered. Peat extraction in bogs results in CO_2 losses via combustion (fuel) or decomposition (horticulture), a decrease in CO_2 uptake by the removal of vegetation and an increase in CO_2 emission by lowering water table (peat oxidation) as Reco is positively correlated with the depth of the oxic acrotelm (Moore and Dalva, 1993; Rowson et al., 2013). Carbon dioxide emissions from degenerated peatlands are estimated to be equivalent to more than 10 % of the global emissions from fossil fuels (Parish et al., 2008). Rewetting recreates anoxic conditions and decreases peat oxidation, thus it was shown in a cutaway peatland that rewetting reduces total respiration, increases gross photosynthesis and thus results in recreating a CO_2 sink a few years after rewetting (Tuittila et al., 1999).

2.2.2 CH_4

The global atmospheric CH_4 concentration has more than doubled since pre-industrial times. As its global warming potential is 25 times higher compared to CO_2 over a 100-year period, it is an important greenhouse gas. In total, human activities contribute to methane emissions about 60 to 70 %, while the greatest source of natural methane emissions are wetlands being therefore of great interest in a global change perspective (IPCC, 2007).

Methane is produced by anaerobic respiration in the waterlogged, anoxic condition of the catotelm by methanogen microorganisms from the domain Archaea. Methanogenesis contributes to the degradation of organic matter using CO_2 or small organic compounds such as acetic acid as electron acceptor (Lai, 2009). Substantial amounts of methane are only produced when labile carbon substrates are available (Couwenberg, 2009a). The produced CH_4 is liberated (Figure 3) to the atmosphere via diffusion along the concentration gradient, ebullition or plant mediated transport through plant arenchyma (Kutzbach et al., 2004; Lai, 2009). Ebullition of CH_4 containing gas bubbles is recorded especially in air pressure declining periods and can be the main transport mechanism (50-64 %) during these phases (Tokida et al., 2005; Tokida et al., 2007). The plant mediated transport, called “chimney effect”, is responsible for 30 up to 100 % of the total methane flux, allowing methane to bypass the oxic acrotelm of the peat soil (Bhullar et al., 2013), while other gases e.g. CO_2 or N_2O are transported as well (Conrad, 1996). This relationship between CH_4 emissions and vascular plants leads to a seasonal variation of CH_4 emission related to the vegetative cycles and to variations between plant communities (Le Mer and Roger, 2001; Yli-Petäys et al., 2007; Drösler et al., 2008). However, 60 to more than 90 % of the CH_4 that is produced in the anoxic layer is reoxidised by methanotrophic bacteria in the acrotelm (Le Mer and Roger, 2001). Raghoebarsing et al. (2005) revealed that methane can also be oxidised by methanotrophic symbionts present in the hyaline cells of *Shagnum* providing a carbon source (10-15 %) to its host species which is instantly incorporated.

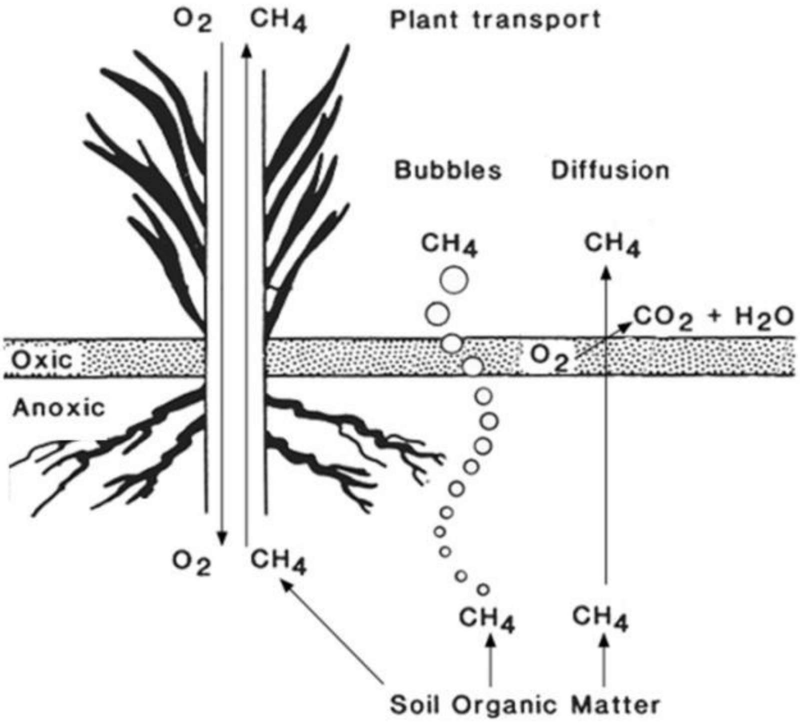


Figure 3: Production, re-oxidation and emission of methane from a vegetated peatland (Cowenberg 2009b, modified).

CH_4 fluxes are positively correlated with temperature, as increasing temperatures generally accelerates microbial processes and diffusion velocity through the soil. Therefore, these emissions follow generally a seasonal curve (Drösler et al., 2008). It was shown that methanotrophy is less sensitive to temperature changes than methanogenesis and that it has a broader temperature tolerance (Dunfield et al., 1993; Moore and Dalva, 1993).

As the water table determines the thickness of the aerobic layer, methane emissions are negatively correlated to the water table depth (Moore and Dalva, 1993). Water table dependency results also in differing fluxes between microforms: the efflux from hummocks is usually about a third of that from hollows (Clymo and Pearce, 1995; Dorodnikov et al., 2011). Additionally a different abundance of vascular plant species at these microform types can contribute to differing fluxes, too, due to the chimney effect.

Significant methane emissions occur only at mean annual water levels above -20 cm (Couwenberg, 2009a). Methane emissions from drained peat soils are therefore negligible small or even negative implying a small net-uptake (Freeman et al., 1993; Couwenberg, 2009b). Rewetting of peatlands leads initially to increased methane emissions by reducing methane oxidation (Vasander et al., 2003; Wilson et al., 2009; Couwenberg, 2009a). Flooding of peat soils for restoration should be avoided as inundation leads to huge CH_4 emissions (up to 205 g-C m^{-2} yr^{-1}) especially if fresh plant litter (originating from plants killed by inundation) is available (Drösler et al., 2008; Hahn-Schöfl et al., 2011). Despite high initial CH_4 pulses, rewetting leads to huge benefits in the C-balance compared to the drained situation as they can be compensated by CO_2 emission savings (Augustin and Joosten, 2007).

2.2.3 N_2O

Nitrous oxide (N_2O) is a greenhouse gas which has a global warming potential of 296 relative to CO_2 (over a 100-yr time horizon). The concentration of N_2O is increasing in the atmosphere while more than one third of the emissions derives from human activities mainly by agricultural soils through increased fertilizer use (IPCC, 2001).

Nitrous oxide is produced in soils through nitrification and denitrification. Nitrification is the aerobic microbial oxidation of ammonia to nitrate, while denitrification is the anaerobic microbial reduction of nitrate to nitrogen gas (N_2). In nitrification N_2O is a by-product that leaks from microbial cells into the soil and ultimately into the atmosphere. It is usually produced by autotrophic bacteria, but under conditions characterised by low pH, high oxygen amounts and the availability of organic material, heterotrophic nitrification can produce significant amounts of N_2O, too (Wrage et al., 2001). However the most efficient N_2O producer in the soil are denitrifiers: N_2O is an intermediate in the sequential reduction pathway of nitrate to N_2 and is produced and consumed by this microbial group as well (Conrad, 1996). The portion of the intermediate N_2O is higher if the pH is low, because the enzyme reducing N_2O to N_2 is inhibited at low pH values (Knowles, 1982). N_2 becomes the main end product if the water filled pore space is greater than 80 % due to oxygen limitation (Bouwman, 1998).

Additionally nitrifier denitrification should be mentioned, which is a production pathway of N_2O by which nitrification and denitrification are performed subsequently by only one group of microorganisms, namely autotrophic NH_3-oxidizer (Wrage et al., 2001).

One of the main controlling factors in the production of N_2O is the availability of inorganic N in the soil (Wrage et al., 2001; IPCC, 2006). Therefore N_2O emissions depend on the availability of nitrate, whereas plants and bacteria compete for this substrate (Silvan et al., 2005; Repo et al., 2009). While in ecosystems in dry temperate or subtropical regions micro organisms seem to be stronger competitors for inorganic N than plants it was shown in peatlands that *Eriophorum vaginatum* is an effective competitor for nitrate and hence controls its availability for denitrification and moderates the N_2O emissions (Silvan et al., 2005).

The highest terrestrial nitrous oxide emissions were observed in agricultural used and tropical soils. Emissions from cultivated organic soils in Europe are one order of magnitude greater than those from mineral soils. The default emission factor for agricultural used organic soils under temperate climate used by the IPCC (2006) is 8 kg N_2O-N ha^{-1} $year^{-1}$.
N_2O emissions from wet virgin peat soils are generally low due to nutrient poor conditions and low nitrification activity and can even show a small uptake of N_2O (Martikainen et al., 1993b; Dinsmore et al., 2009). However large emissions of N_2O (0.9 - 1.4 g N_2O per m^2) were observed from Repo et al. (2009) and Marushchak (2011) in tundra peat soils, when cryoturbation formed bare peat surfaces (peat circles). These unvegetated surfaces display a lack in competition for nitrogen between plants and microbes, thus produced NO_3^- is available for denitrification and can be emitted as N_2O.

It was shown that nitrous oxide emissions from peat soils strongly depend on the C/N ratio, which can thus be used to predict N_2O emissions, while soils with a C/N ratio higher than 25 showed only negligible N_2O emissions. However, at low C/N ratios (i.e. below 15–20) other parameters such as climate, pH and groundwater tables increase in importance to explain N_2O emissions (Klemedtsson et al., 2005).

Lowering the water table of peat soils e.g. for agricultural use can significantly increase N_2O by accelerated decomposition of organic matter (Regina et al., 1999; Couwenberg, 2009b; Rochette et al., 2010). Emission strength in drained peatlands depends on the nutrient status of the peat soil. While nutrient poor drained bog peat displays near to negligible N_2O fluxes, emissions increased in minerotrophic nutrient rich fens (Martikainen et al., 1993; Aerts and Ludwig, 1997).

2.3 Peatland use and restoration

While living peatlands have a cooling effect on the climate by acting as a carbon sink, degraded peatlands are a major and growing source of greenhouse gases. Worldwide huge areas of peatlands are destroyed, due to peat mining for fuel and horticulture and by drainage for agricultural use (Joosten, 2006), while Germany and Canada account for over half of horticultural peat extraction (Strack, 2008). The exploitation of north-western Europe's peatlands started in the Late Neolithic (4000 before present) and Europe is the continent with the largest peatland losses (Joosten and Clarke, 2002; Koster, 2005). Today, a mere 1 % of the former extent of European peatlands has still living mire vegetation and accumulates peat (Koster, 2005).

Restoration of peatlands implies not only a recovery of landscapes and biodiversity but also of ecosystem services such as carbon storage, water regulation or climate change mitigation/climate protection as shown in Figure 4 (Drösler et al., 2009; Grand-Clement et al., 2013). Carbon dioxide emissions from degenerated peatlands are estimated to be equivalent to more than 10 % of the global emissions from fossil fuels (Parish et al., 2008). In Germany, the estimated peatland GHG exchange accounts for 2.3-4.5 % of the anthropogenic emissions (Drösler et al., 2008). Restoration can recover the carbon sink function within some years after rewetting (Tuittila et al., 1999; Vasander et al., 2003; Bortoluzzi et al., 2006; Yli-Petäys et al., 2007). Consequently, conservation projects are aiming to re-establish active peat-forming mires, which practically means in the first instance blocking drainage to increase water table to a predrained level (Grand-Clement et al., 2013). In Germany a potential reduction of 35 million tons carbon dioxide equivalents per year is possible by peatland-restoration, which is the most cost-effective mitigation strategy (Joosten, 2006; Drösler et al., 2009; Freibauer et al., 2009). Due to the economic and tradeable value of carbon offset, it could be possible that restoration itself funds restoration cost (Worrall et al., 2009). However in the future conservation and restoration of wetlands in general will be more challenging due to climate change (Erwin, 2009; Grand-Clement et al., 2013).

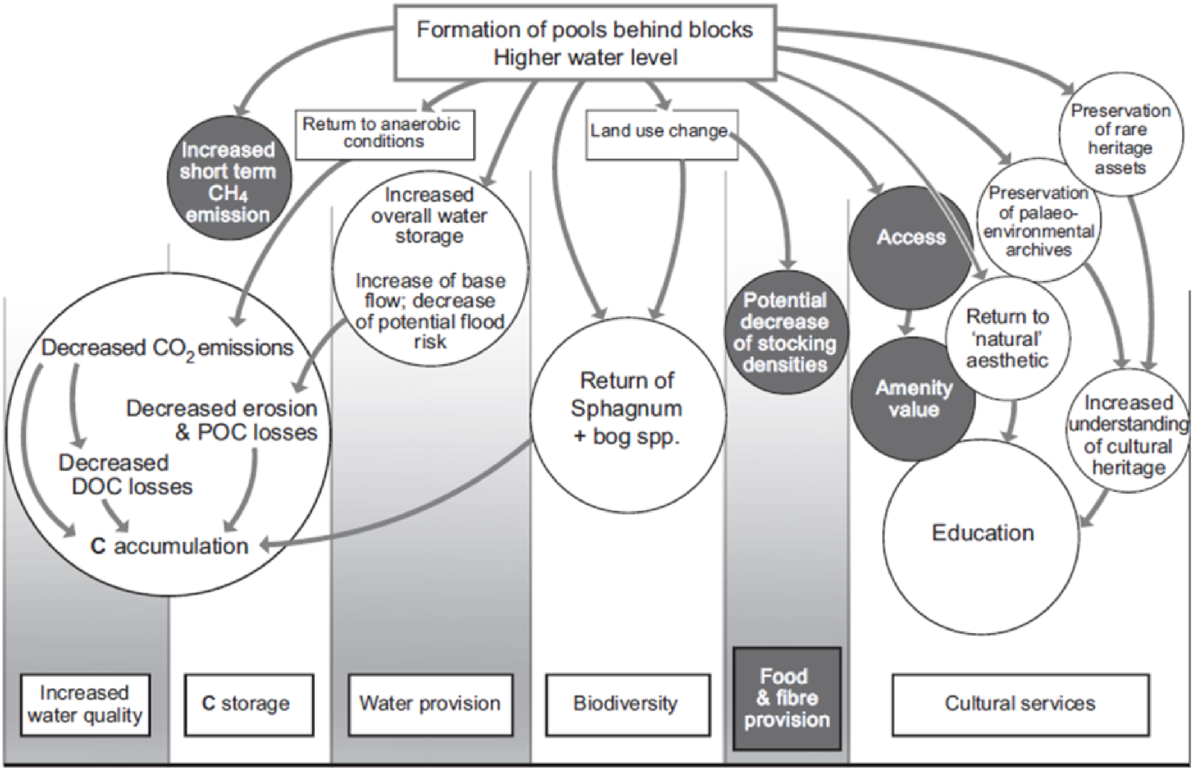


Figure 4: Conceptual model of the impact of drain blocking on ecosystem services provided by peatlands. Circle sizes value the expected impact. Light and dark circles indicate overall positively and negatively affected ecosystem services respectively (Grand-Clement et al., 2013).

2.4 Peatlands and climate change

Potentially damaging for peatlands are long-term environmental changes like global warming as well as changes in precipitation amount and frequency or atmospheric nitrogen deposition (Dise, 2009). As wetlands in general, peatlands are vulnerable to changes in quantity of water supply (Erwin, 2009). Particular susceptible are mid-latitude peatlands like in Western Europe, especially if summer precipitation decreases like predicted (Christensen et al., 2007; Charman et al., 2013) and increased winter precipitation does not provide compensation for summer drought due to runoff of excess water (Parish et al., 2008). In particular this applies for ombrotrophic bogs, while groundwater flow of fens can buffer temporary drought periods (Parish et al., 2008). Drier surfaces will in consequence emit less CH_4 but more CO_2 and N_2O (Gorham, 1991; Freeman et al., 1993; Parish et al., 2008; Wilson et al., 2009). In contrast, the predicted warming and atmospheric CO_2 increase could increase net ecosystem productivity which is positively correlated with CH_4 emissions by increasing substrate availability to methanogens and increased plant mediated transport (Whiting and Chanton, 1992; Drösler et al., 2008; Dorodnikov et al., 2011). Poll et al. (2013) concluded from their manipulation experiment in an agricultural ecosystem that the moisture regime of a soil will determine the effect of elevated temperatures on soil respiration. They found that water limitation reduced microbial biomass and thereby the stimulatory effect of elevated temperature on soil respiration was suppressed.

The challenge is to forecast the future environmental conditions and the possible changes and feedbacks of peatlands that may be provoked by these modified conditions. To handle this challenge it is important to continue and expand research in this vulnerable ecosystem for instance by manipulation experiments in the field (Dise, 2009; Poll et al., 2013). Jassey et al. (2013) performed a manipulation experiment simulating global warming by open top chambers showing that vascular plant cover benefits from warming while *Sphagnum* cover decreases by altered competition. Increased air temperature and changes in precipitation could convert the carbon sink into a carbon source (Worrall et al., 2009) by accelerated peat decomposition (Dise and Phoenix, 2011; Charman et al., 2013) or negatively affected CO_2 uptake of *Sphagnum* species (McNeil and Waddington, 2003; Robroek et al., 2009). The latter stresses the strong impact of drought on the carbon budget of raised bogs (Robroek et al., 2009). The effect of droughts on GPP, Reco and NEE will vary depending on the species composition and hydrology of each particular peatland ecosystem (Bubier et al., 2003).

On a global scale the response of peatlands to climate change can be contrary. It is suggested that the reduction of peatland extend in the mid-latitudes might be compensated by increased carbon sequestration in high-latitude peatlands and their expansion into areas being currently to cold and dry for peat formation (Parish et al., 2008; Charman et al., 2013; Loisel and Yu, 2013).

3. Material and Methods

3.1 Investigation area

The Himmelmoor is located approximately 20 km north-western of Hamburg close to Quickborn (Figure 5 and Figure 6). It is bordered by the beck Bilsbek (north-west) and the river Pinnau (east) which drains into the river Elbe and is with an extent of about 6 km² one of the largest raised bogs in Schleswig-Holstein. The mean precipitation sum in Quickborn is 838.0 mm per year and the mean air temperature is 9.0 ℃ (long-term average from 1981 to 2010, DWD 2013).

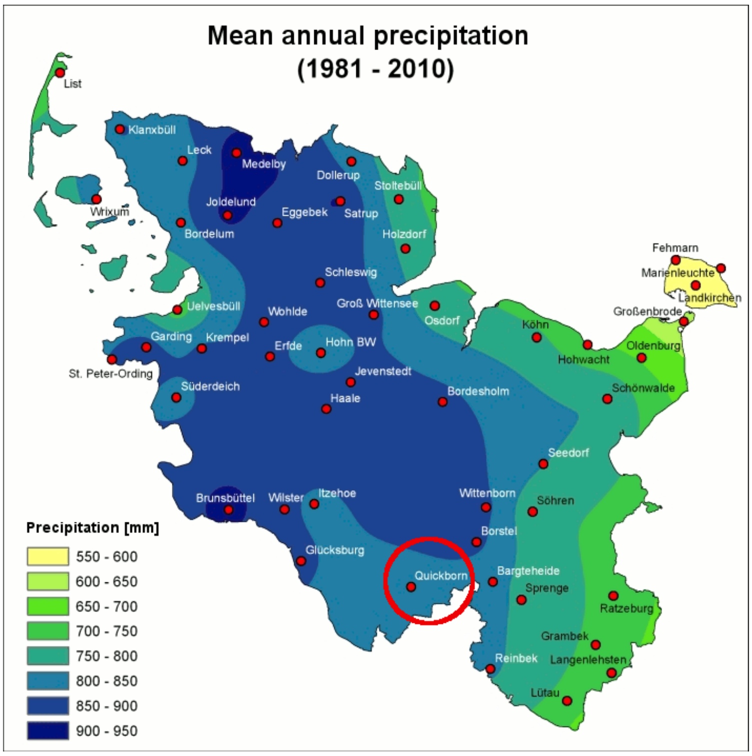


Figure 5: Location of the investigation area (red circle) and mean anual precipitation (1981 2010) of Schleswig-Holstein (modified, www.schleswig-holstein.de).

Peat formation in the Himmelmoor started after the last ice age 10.020 ± 100 years before present by infilling bodies of water in a depressional area (characterized by underlying mud) and locally by paludification on poorly drained soil (saalian age till). The different states of low-mire peat found in the Himmelmoor are: sedge peat, reed peat and alder/birch carr peat. After precipitation increases within the atlantic period the formation of the raised bog started with sphagnum-peat formation characterized by hummock-hollow complexes. Bog and fen peat together can reach an overall thickness of 10 m (Pfeiffer, 1998; Grube et al., 2010).

Since the 18th century peat mining started in the boundary areas by block trench cutting for fuel until the 1960th. Industrial peat mining started in 1920 and persists until now. In 2004 approximately 50.000 m³ peat per year were harvested at an area of 130 ha for horticulture using the milled peat technique (Struppek, 2004). As peat mining contracts are limited until 2020 and legislation requires restoration, the area will be rewetted successively. The rewetting process has started in 2002 in the boundary areas by building dams to re-establish an increased water table. In 2008 a first part of the industrial peat mining area was ceased, other parts followed 2009 and 2011. Thus the volume of harvested peat in 2011 was reduced in comparison to 2004: according to information provided by the peat company 38 000 m³ peat were harvested at an area of 70 ha. Peat mining will be completely ceased in 2016 due to exhaustion of usable peat resources (personal communication 22.10.2013).

As all raised bogs the Himmelmoor (expect the industrial peat mining area) is protected under the Habitats Directive of the European Union (Evans, 2006).

3.2 Measurement site

The measurement site is located north-western of the industrial used area (Figure 6). It is separated from it by a local elevation (called “Knust”) whose stratigraphy never was changed by peat extraction and thus represents the former land surface of the raised bog. However, it’s depth decreased by subsidence due to increased mineralisation as a result of drainage. The measurement site was used for block trench cutting by which the stratigraphy of the peat layers has altered: after removing the upper layer (white peat and mire vegetation) black peat was cut for fuel and trenches were subsequently filled up with the put aside white peat. In the 1960th block cutting was ceased and the site was abandoned. Restoration began in the 1980th with blocking drainage and cutting birches in order to raise the water table. Birch stems were piled or left lying around. The last birch cutting event was in 2008.

Today this former block cutting area is populated by typical high moor flora, but the areas between former drainage ditches (distance 30-45 m) show differences in vegetation communities. Thus three subsites with differing dominant vegetation were identified and one reference site for analysing possible benefits from restoration at the industrial used area (approx. 200 m apart, Table 2 and Figure 6).

Table 2: Description of experimental subsites in the Himmelmoor.

Subsite		Dominating species
Heath	H	*Erika tetralix* and *Calluna vulgaris.*
Sphagnum	S	*Sphagnum spec.* and *Eriophorum angustifolium.*
Purple Moor Gras	P	Dense tussocks of *Molinia caerulea,* nearly no other species.
Extraction Site	E	Industrial peat extraction site, bare peat and no visible plant growth, but a greenish surface indicating algae growth.

Figure 6: Above: Overview over the location of the Himmelmoor close to the City Quickborn. Bottom: Detailed view on the measurement sites and restoration status in the Himmelmoor. Subsite abbreviations: H= heath, S= Sphagnum, P= purple moor grass and E= extract site (Satellite images by Google Earth 2013).

3.3 Rainfall exclusion

To simulate a summer drought which is likely to occur in Central Europe due to climate change (Pal et al., 2004), a rainfall exclusion experiment was set up.
At each subsite eight plots with 3 by 3 m were selected. Subsite E got only four plots because no rainfall exclusion was performed here. Four replicates of each subsite were chosen randomly as drought plots while the remaining plots are the control replicates (Figure 7). The number of replicates and the two different treatments allows all analysis to be evaluated statistically with an analysis of variance (ANOVA) using SigmaPlot (Systat Inc.).

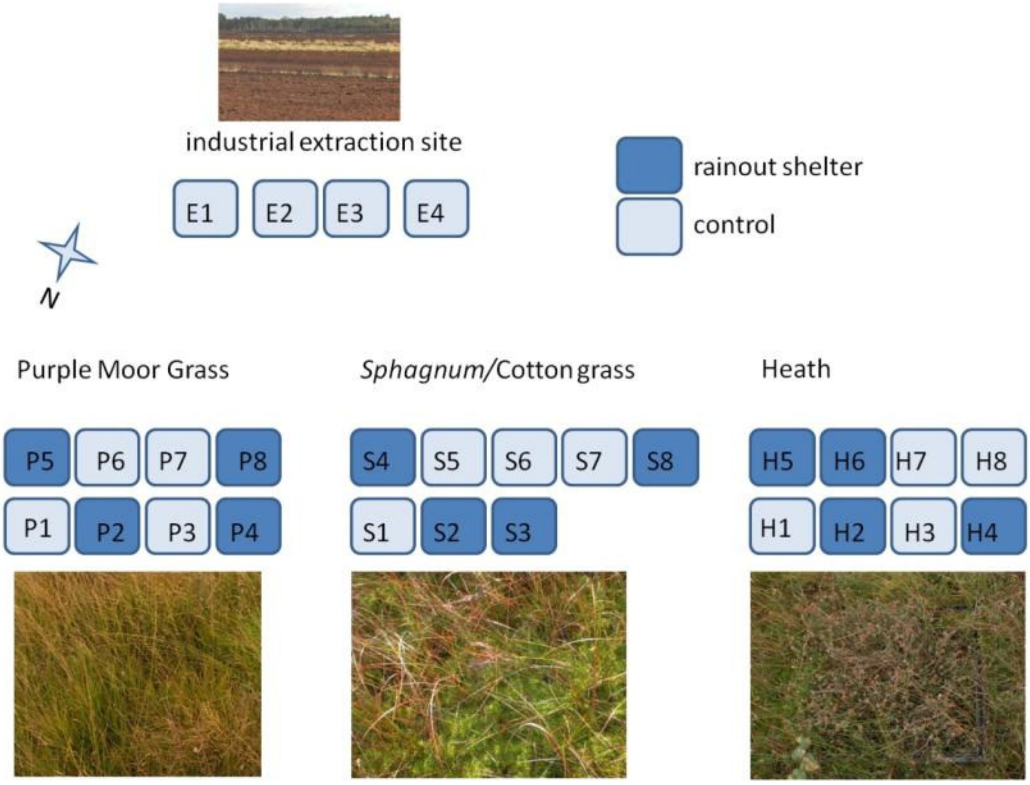


Figure 7: Overview about plots and distribution of the treatments in the Himmelmoor (picture source: *Sphagnum* subsite: M.Greeven, Purple moor grass and heath subsite: D.Holl).

Drought plots were equipped with metal rainout shelter racks (Figure 8) covering an area of 3 by 3 m. Wood piles were plunged into the soil to anchor the shelter legs approx. 30 cm above the soil surface. Total shelter height was then 1 m at the lowest site and 1.5 m at the tallest site, thus having a 17 % inclination. This heights minimizes greenhouse effects by allowing near-surface air exchange (Glaser et al., 2013) and additionally taking into account the growth of small birches at the study site. Shelter racks were covered with a transparent greenhouse plastic sheet (trade name: “UV-Window”, by folitec GmbH, Germany), from the beginning of May till the end of September of each year (2010 and 2011).

The sheet is UV-B permeable and the UV-transmittance ranges between 70-90 % depending on wavelength. The transmittance in the PAR wavelength range is between 85-90 %. According to Yahdjian and Sala (2002) shelters were oriented in north-south direction, with the tallest side to the south tending that most light can directly shine on the plot area without passing through the roof (Figure 8).

Figure 8: Rainout shelters at the heath subsite in the Himmelmoor.

As photosynthetically active radiation (PAR) and air temperature are important input values for modelling CO_2 fluxes, the shelter effect on these two parameters was quantified. One PAR sensor (SKP 215, Skye instruments) was place under a shelter and a control one was used at the meteorological station (Chapter 3.5) 2 m aside. Measuring interval was 1 s while a half-hourly mean was logged. Additionally at each subsite one control and one sheltered plot were equipped with a temperature logger (EBI 20T, ± 0.2 °C, EBRO) during the sheltered period, to estimate shelter effect on air temperature. Temperature logger were shaded in small, ventilated white boxes made out of wood ribs and positioned approx 1 m above the surface. Measuring interval was 15 minutes. Temperature loggers were tested for compliance in a cooled incubator (RUMED® 3051, Rubarth Apparate GmbH) for three days with changing temperatures between 0 and 40 °C. The data measured in the field was corrected by the default determined in the test before comparing the sheltered values with the unsheltered ones.

All performed analyses were done within the inner 2 m² of the sheltered area, assuming that the outer 50 cm at each side receive some amounts of rainwater if rain does not fall vertically and from shelter runoff (edge effect).

The use of these rainout shelters was a project requirement, however, they were adapted to the wet bog ecosystem by keeping away 100 % of precipitation instead of 25 % as used in other ecosystems with well drained soils.

3.4 Vegetation analysis

Vegetation analyses at the three vegetated subsites were gratefully performed by the KLIMZUG-Nord project partner Sebastian Roman Schmidt from the group of Applied Plant Ecology at the Biocenter Klein Flottbek (University of Hamburg). He provided vegetation coverage and abundance of 2010 and 2011 according to the decimal scale of Londo (1976).

3.5 Environmental parameter

At the three vegetated subsites soil and climate measurement stations powered by solar energy were permanently installed for the continuous monitoring of environmental variables (Table 3). Soil temperature, soil water potential, volumetric water content and water table were measured at each subsite in two replicates per treatment to compare between sheltered and control plots.

Table 3: Monitoring of environmental parameter at the soil and climate measurement stations.

Parameter	Used equipments	Intervals/remarks
Wind speed/direction	Wind monitor, 05103-5 (Young)	
PAR	Quantum sensor, SKP215 (Skye instruments)	half hourly means from 1 s intervals
Air temperature/humidity	Temperature probe HMP45 (Campbell scientific)	
Soil temperature	107 thermistor (Campbell scientific)	
Soil water potential	Soil matric potential Sensor, 257-L (Campbell scientific)	half-hourly readings Depth: 10 cm
Precipitation	Tipping bucket rain gauge, 52202 (Young)	half-hourly sums
Air pressure	Barometric pressure sensor, Setra278 (Campbell scientific)	half hourly means from 30 s intervals
Water table	Ground water data logger, “mini diver” (Schlumberger)	15 min readings

3.6 Soil analyses

To characterise the soils of the four subsites and to find potential differences in e.g. growth conditions for plants or nutrient availabilities (which may explain differences in GHG fluxes) various soil analyses were performed as described below.

3.6.1 Soil sampling

Assessment on summer drought on soil parameter was done monthly during the sheltered period (Mai-Oktober) of 2011. Soil samples from the upper 10 cm were taken and analysed for water content, C/N ratio, nitrate/nitrite, ammonium and total dissolved nitrogen (TDN). For this purpose the peat was shredded with a knife mill (Grindomix GM200, Retsch) until the sample was homogenised. Nitrate/nitrite, ammonium and TDN were analysed in a $CaCl_2$-extract of the fresh soil (5 g wet soil was extracted for 1 hour with 20 ml $CaCl_2$ (0,0125 $mol*l^{-1}$) and centrifuged at 8500 rpm for 20 minutes). Nitrate and nitrite were measured using HPLC (Agilent 1200) while ammonium was determined photometrically as described in Sanders (2011). TDN was estimated by alkaline persulfate digestion converting all nitrogen to nitrate (Hagedorn et al., 2000; Sanders, 2011) which was subsequently measured using HPLC. Dissolved organic nitrogen (DON) was calculated by subtracting DIN (nitrate, nitrite and ammonium) from TDN.

Water contend was determined gravimetrically and C/N ratio by high temperature combustion of oven dried and milled samples at 900 °C using an elemental analyser (VarioMax, Elementar). According to Chambers (2011) the commonly assumption is made that in non-calcareous, acidic peat soils total C equals organic C content.

3.6.2 Soil pore water sampling

To analyse nitrate/nitrite, ammonium and total dissolved nitrogen (TDN) in the soil pore water, samples were taken monthly from May 2010 until December 2011 from all vegetated plots. Samples were taken by inserting “Rhizon Soil Moisture Samplers” (pore size 0,15 µm, Eijkelkamp) approx. 5 cm deep into the soil. Rhizons were connected to a 10 ml vacuum glas tube to obtain a sample (max. 7 ml). Samples were transported and stored cooled. Nitrate/nitrite, ammonium and TDN were determined as described in chapter 3.5.1 (soil sampling). Sample volumes were not always sufficient for determining TDN.

3.6.3 Soil profiles

For the characterisation of the peat soils of the measurement site, four soil profiles were dug in 2011(one at each subsite) approx. 1 m deep. Profiles were described pedologically on site according to the German Soil Classification System (Ad-hoc-AG Boden, 2005) and classified according to the US Soil Taxonomy (Soil Survey Staff, 2010) and World Reference Base for Soil Resources (IUSS Working Group WRB, 2007). Degree of decomposition was assessed in the field following the von Post measure (von Post and Granlund, 1926). Soil samples were taken from every identified pedogenic horizon and homogenized subsequently in the lab with a knife mill (Chapter 3.6.1). Samples were stored cooled and analysed for water content, C/N ratio, nitrate/nitrite, ammonium and total dissolved nitrogen (TDN) as described in chapter 3.6.1. Additionally pH, ash content (non combustible material at 550 °C), and bulk density were measured following operation procedures for peat soils (Hoffmann, 1997).

3.6.4 Net nitrogen mineralisation

The possible effect of a drought on the nitrogen availability and net nitrogen mineralisation (ammonification and nitrification) in the peat soil was assessed within an aerobic incubation experiment under controlled conditions during 6 weeks.

From each subsite soil samples of the upper 10 cm were taken from three replicate control plots. Samples were homogenized subsequently in the lab with a knife mill (Chapter 3.6.1). 10 mg of wet soil was incubated in duplicates for 0, 2, 4 or 6 weeks and extracted with $CaCl_2$ as described by Sanders (2011). DIN was measured within the extract as described in chapter 3.6.1 (soil sampling). To assess the possible effect of a drought this experiment was performed at the same time with identical soil samples, but reduced water contents. Water content was determined gravimetrically and soil samples were air dried subsequently until their water content reached -15 % and -30 % respectively of the original water contend. These amounts were chosen as a regional climate model (REMO, scenario A1B) predicts a reduction in summer precipitation by a maximum of 30 % (Umweltbundesamt, 2006). For analysing seasonal effects this experiment was executed once a season in 2011 (January, March, August and October).

3.7 Chamber measurements

At each plot (in total 28) plastic collars (60 x 60 cm) were permanently installed by inserting them approximately 50 cm deep into the peat soil to assure that they reach under the water table also in dry periods (especially at the drier extraction subsite). Installation took place two month before the beginning of the chamber measurements to allow recovery of disturbance. The position of the collars was chosen within the inner 2 m^2 of each measurement plot at a place where vegetation represents the vegetation composition of the whole subsite. These collars were used for CO_2 flux measurements as well as for CH_4 and N_2O flux measurements. Collars were equipped with a groove around the top, to be filled with water during measurement to avoid gas exchange.

All plots at the vegetated sites were equipped with board walks which do not rest on the peat (Figure 8). This procedure avoids damage of the vegetation and vibrations during measurements, which may provoke ebullition. Board walks were oriented northwards of the plots so that no shading through a person will occur during measurements with the transparent chamber.

3.7.1 CO_2 flux measurements

To measure CO_2 fluxes a close, transparent, climate-controlled chamber system with an infrared gas analyser (IRGA, LI-840, LI-COR inc.) was used as described by Schneider et al. (2011). Additionally the chamber was equipped with a PAR-sensor inside the chamber (Figure 9). Measurements were done for three minutes with recording CO_2, water vapour, PAR and temperature every second. To increase the number of net ecosystem exchange (NEE) measurements and to gain a wide spectrum of different light conditions for modelling, the transparent chamber was shaded in two intensities with black gauze (Burrows et al., 2005; Elsgaard et al., 2012). After the first measurement with the transparent chamber a second measurement was performed while shading the chamber with one layer of gauze (PAR approx. 50%) and a third one with two layers (PAR approx 30%). Subsequently the chamber was darkened in a fourth measurement with an opaque cover (PAR = 0) to estimate ecosystem respiration (Reco). Between each measurement the chamber was removed and ventilated to obtain ambient CO_2 concentrations within the camber. If vegetation exceeds chamber height, a transparent polycarbonate elongation of 60 cm was used, which was shaded and darkened correspondently (Figure 9). Measurements were generally conducted between 10:00 am and 02:00 pm when PAR reached the maximum and measuring order of the subsites was randomised.

CO_2 flux measurement was performed from August 2010 until January 2012. Measurement intervals depend on vegetation growth with higher frequency in summer than in winter (at least biweekly up to twice a week). Between December 2010 and February 2011 only one CO_2 flux measurement took place because of a thick snow cover. To compare day and night respiratory fluxes night time measurements were done at one day in August 2011. In total a considerable quantity of 6079 CO_2 flux measurements was done (NEE: 4303, Reco: 1776).

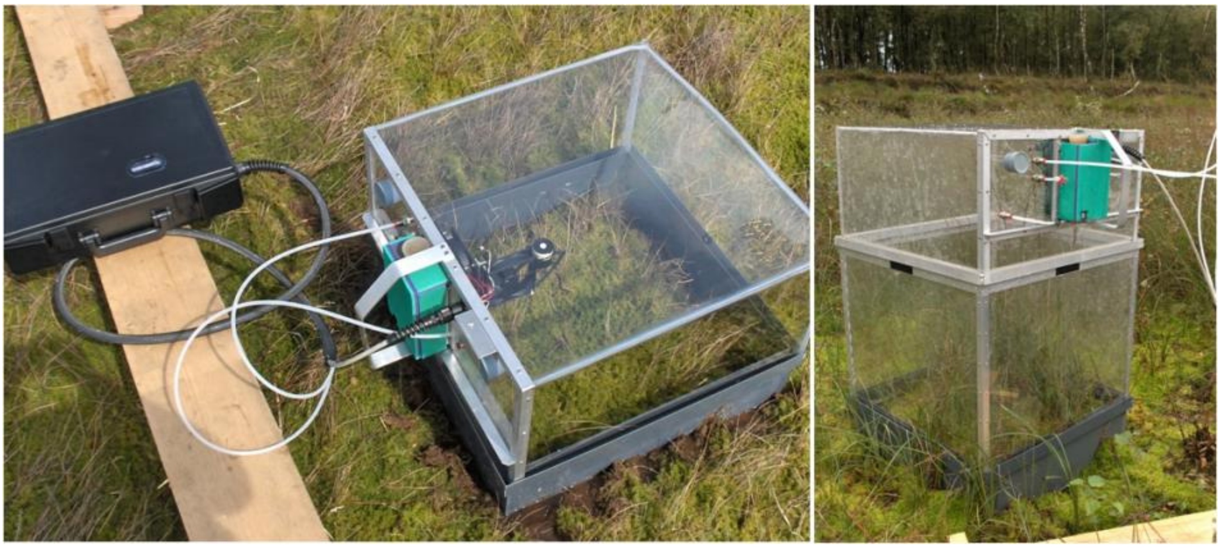


Figure 9: Close chamber system with measurement unit (left) and with optional elongation (right).

3.7.2 CH_4 and N_2O flux measurements

Measurements were done with closed aluminium chambers (60 x 60 x 32 cm, an elongation of 60 cm was used if vegetation height exceeds chamber height). They were equipped with a fan, a pressure vent, a thermometer for chamber air temperature and a sampling port. Two circular opening (4 cm diameter) at the front site were open while setting the chamber on the collar and closed afterwards to reduce initial pressure shocks. Within the closing time of 20 minutes six samples were taken of the chamber headspace with 60 ml plastic syringes connected to the sampling port via three-way stopcocks. Sampling started directly after closing and the remaining five samples followed in intervals of four minutes. Gas samples were analysed subsequently in the lab using a gas chromatograph (GC) as described below.

CH_4 and N_2O flux measurements were performed over a one-year period from April 2011 until March 2012. Measurement intervals were biweekly (except December 2011: once per month) for methane and monthly for nitrous oxide as first results showed no significant fluxes. CH_4 was measured at all four replicates per subsite and treatment while N_2O only at three replicates. To compare day and night fluxes, night time measurements were done at one day in August 2011. In total a quantity of 607 CH_4 and 249 N_2O flux measurements was done.

Methane

Gas samples were analysed using a GC equipped with a flame ionization detector (HP 5890 Packard Series II). Syringes were directly connected (HP 5890 Packard Series II, FID) via a loop. Analyses were done within four days after sampling and each sample was analysed twice. Two standard gases were used for calibration (1.7 ppm and 200 ppm CH_4) being injected triply before and after samples of three plots. Methane concentrations of the samples were calculated based on the means of the two calibration events being temporally closest to the measurement of the samples.

Nitrous oxide

N_2O was measured at a GC provided with an electron capture detector (Agilent Technologies 7890A). As connecting the syringes to the GC was not possible, 20 ml of the gas sample was injected in the field into an air-filled septum-vial, from where it was then taken with a microliter syringe and injected into the GC. The GC was calibrated daily with three standard gas concentrations (0,3 ppm, 0,9 ppm and 1,5 ppm N_2O) being injected triply before measurement. Since the samples were diluted by injecting them into the air-filled vials standards were treated similar to have the appropriate concentration. As injecting the gas sample into the septum-vial and from there again into the GC might cause mistakes and a decrease in accuracy the procedure was tested with a standard gas. There was no decrease in reproducibility in comparison to a standard gas that was injected directly into the GC (the coefficient of variation was 0,01 in both cases).

3.8 Flux calculation and modelling

Fluxes of CO_2, CH_4 and N_2O were calculated from concentration changes over time. According to the micrometeorological sign convention negative flux values represent a net loss of CO_2 from the atmosphere to the vegetation (CO_2 uptake through photosynthesis) and positive values represent emissions of CO_2 through both autotrophic and heterotrophic respiration (Lund et al., 2009). The same applies for CH_4 and N_2O fluxes.

3.8.1 CO_2 flux calculation

Flux calculation of each single CO_2 chamber measurement was done with an updated version of the MATLAB® routine of Kutzbach et al. (2007) using linear and non-linear regression. As no linear increase or decrease of GHGs within the chamber headspace can be expected (Kutzbach et al., 2007) an exponential fitting was normally used to determine initial concentration changes. The first and the last 10 seconds of the 180 data points of each measurement were discarded and the flux rate was calculated at $t = 10$ s out of the remaining 160 data points with water vapour correction of CO_2 concentrations.

Each single flux curve was reviewed for abnormalities such as abrupt changes in slope due to e.g. changes in PAR derived from cloud movement. If possible, the flux was recalculated by using only a part of the 160 s interval with constant conditions (minimum 40 s). Secondly dataset was checked by plotting the standard deviation of the residuals against the flux to see if the point cloud is evenly distributed. Flux calculations with outlying residuals were checked again for mistakes e.g. in data preparation and were discarded from the dataset if the mistake could not be eliminated. Standard deviation of the residuals of most of the data (98%) was lower than 0.55 ppm and had a mean of 0.42 ± 0.06 ppm, which is remarkably low as the noise of the IRGA is specified to be <1 ppm. Finally was screened if the non-linear fitted curves are concordant with the theoretical model of Kutzbach et al. (2007). If CO_2 flux curvatures show an upward concavity, the slope of the linear regression was used instead for estimating the flux as executed by Schneider et al. (2011). It was shown that for this type of flux curves a linear flux calculation achieves more robust flux estimates (Schäfer, 2012).

3.8.2 CO_2 flux modelling

Net ecosystem exchange (NEE), ecosystem respiration (Reco) and gross primary production (GPP) fluxes were modelled over a complete year (2011). Modelling was based on ambient PAR and air temperature measured at the meteorological station in half-hour steps.

To analyse the differences between and within vegetation types and treatments, each single plot (28) was modelled separately. All modelled fluxes (Reco, GPP and NEE) were summed up for the whole year 2011 and the values of the 4 plots from one treatment and vegetation type were averaged and compared statistically using ANOVA.

Gross primary production

GPP fluxes were calculated as the difference of the measured NEE and Reco fluxes. Depending on measurement interval and season a rectangular hyperbolic light response curve (GPP model 1 (Schäfer, 2012),Table 4) was fitted to the GPP data of one week up to one month using OriginPro (OriginLab Corp.). With this model, light saturation points (Pmax) for a certain time of the year were determined for each plot (using PAR values from inside of the chamber). In cases where the hyperbolic curve could not obtain an acceptable curve (standard error exceeds the value of Pmax) a linear model was used (GPP Model 2). When the linear model did not explain the data appropriate (standard error exceeds the value of intersection and slope) the mean of the GPP flux values was used to model the time series for this period. This appeared only in winter time when GPP ranged around zero.

Table 4: Two model approaches for Gross primary production (GPP) as a function of photosynthetically active radiance (PAR).

GPP Model	Model formula	Remarks
1	$GPP = \frac{Pmax \ \ \alpha \ PAR}{Pmax + \alpha \ PAR}$	Rectangular hyperbolic function (Schäfer, 2012). Pmax is the asymptotic value of GPP at high PAR and *α* is the slope at t = 0.
2	$GPP = a + b \ \ PAR$	Linear model, a and b are fitting parameter.

Model parameters were interpolated linearly between the certain time periods. When two different models temporally succeed each other, the model parameter could not be interpolated, but rather the results for the modelled GPP. In this case, and also when the mean value of the measured GPP fluxes was used, the modelled GPP values for this timeframe were not calculated as a function of PAR. This would result in GPP fluxes even in nighttimes, so that GPP was set to zero when PAR was below 5 µmol m^{-2} s^{-1} to have no photosynthetic uptake in these cases.

Ecosystem respiration (Reco)

Measured Reco-fluxes were plotted as a function of air temperature and soil temperature. Three different models were tested to explain the temperature dependency of Reco respectively (Table 5).

Table 5: Temperature driven ecosystem respiration (Reco) models, where t = air /soil temperature respectively and a,b and c are fitting parameter.

Reco Model	Model formula	Remarks
1	$\text{Reco} = \frac{a}{1 + be^{-kt}}$	Simple logistic function (Richards, 1959; Rodeghiero and Cescatti, 2005; Schäfer, 2012)
2	$\text{Reco} = ae^{bt}$	Two parameter exponential function (Schneider et al., 2009; Schäfer, 2012)
3	$\text{Reco} = a + be^{ct}$	Exponential growth model, three parameter exponential function.

The best suited model (Reco Model 1-3 and air or soil temperature) was identified by comparing the qualifying parameter R^2 (adjusted) and reduced chi-square. It was thus chosen as model for all plots, also if another model had better qualifying parameter for a single plot. This procedure ensures the comparability between the modelled results of all plots.

Net ecosystem exchange

NEE was calculated as the sum of modelled half hourly GPP and Reco fluxes showing negative values for a net CO_2 uptake and positive values for a net CO_2 emission.

3.8.3 CH_4 and N_2O

Flux calculation

Flux calculation of each CH_4 and N_2O chamber measurement was done with an updated version of the MATLAB® routine of Forbrich et al. (2010) using linear and non-linear regression. Each single flux curve was reviewed for abnormalities such as abrupt changes in slope due to e.g. ebullition. As only six concentration measurements were available for flux calculation, the application of the non-linear regression model is checked by the Akaike information criterion with small sample correction (AICc), as proposed by Forbrich et al. (2010).

As the majority of the N_2O flux measurements ranged around zero, the slope of the flux was checked for significance with a t-test. Significant fluxes are highlighted in the result graphs, however, all fluxes were included in N_2O calculations. This procedure is justified because most of the flux measurement over the year ranged around zero and were not significant. Balancing only the higher, significant fluxes would artificially increase N_2O flux estimates. With the very limited number of significant fluxes no correlation with water table depth or soil temperature can be made. Thus, calculated N_2O fluxes were averaged to estimate annual N_2O emissions or uptakes.

CH_4 Flux modelling

Measured CH_4 fluxes were plotted as a function of soil temperature and water table depth. Modified after Saarnio et al. (1997) an exponential multiple regression model was used to explain CH_4 fluxes by water table depth and soil temperature (Table 6). Secondly the model was used to correlate CH_4 fluxes with soil temperature and water tension, as the rainout treatment may not change the water table depth, but the water tension at the soil surface. Additionally light saturation points (Pmax) calculated for the plots (Chapter 3.8.2) were used as input variable as a proxy for plant growth status. As none of these model approaches explained CH_4 flux variation properly to gap-fill the time series of measured CH_4 ecosystem fluxes, the mean of the measured CH_4 fluxes was used instead.

Table 6: CH_4 model, where t10 = soil temperature at 10 cm depth, WT = water table, wten10 = water tension at 10 cm depth, Pmax = light saturation point and "α", "β" and "γ" are fitting parameter.

CH_4 Model	Model formula	Remarks
1	$CH_4 = \alpha \ e^{\beta \, t10} \ e^{\gamma \, WT}$	
2	$CH_4 = \alpha \ e^{\beta \, t} \ e^{\gamma \, wten10}$	Exponential multiple regression model (Saarnio et al., 1997)
3	$CH_4 = \alpha \ e^{\beta \, t} \ e^{\gamma \, Pmax}$	

GHG budget

For the calculation of the greenhouse gas budget the fluxes of CH_4 and CO_2 were converted in CO_2 equivalents according to their global warming potentials (chapter 2). The C losses due to peat mining were estimated by the amount of harvested peat in 2011 and the bulk density and C-contents measured at the upper three soil horizons of subsite E. Although not all of the harvested peat decomposes and emits its entire C as CO_2 within one year, it is completely added to the balance of one year. This procedure is justified as the whole C-loss is effectively apparent on site. Additionally, it would be challenging to determine the proportional amount of CO_2 emitted by the harvest of the current year by which the proportional amounts of CO_2 emissions of the harvests years before must be added.

4. Results

4.1 Environmental conditions

4.1.1 Mean air and soil temperature

The mean annual air temperature in 2010 was 7.7 °C measured at the meteorological station in Quickborn (DWD) which is located approximately 3 km away from the field site and 9.4 °C in 2011 measured at the field site in the Himmelmoor. In comparison to 2011, the year 2010 was colder from January until June and in December, while it was warmer from July until November with a steep maximum in July (Figure 10). The beginning of 2012 was marked with an exceptionally warm January switching over in an exceptionally cold begin of February (Figure 11). The mean annual soil temperature at 10 cm depth at the field site in 2011 was 9.0 °C and had less fluctuation than air temperature and never fell below zero.

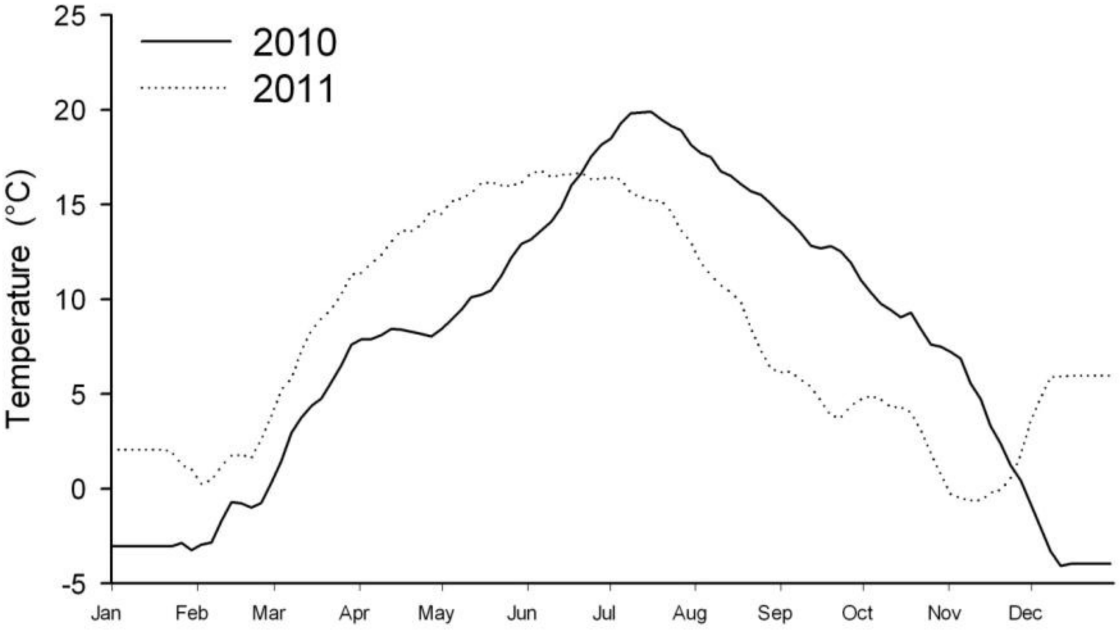


Figure 10: Smoothed mean daily air temperature of 2010 and 2011. Data source 2010: DWD meteorological station in Quickborn and 2011: own data from the field site.

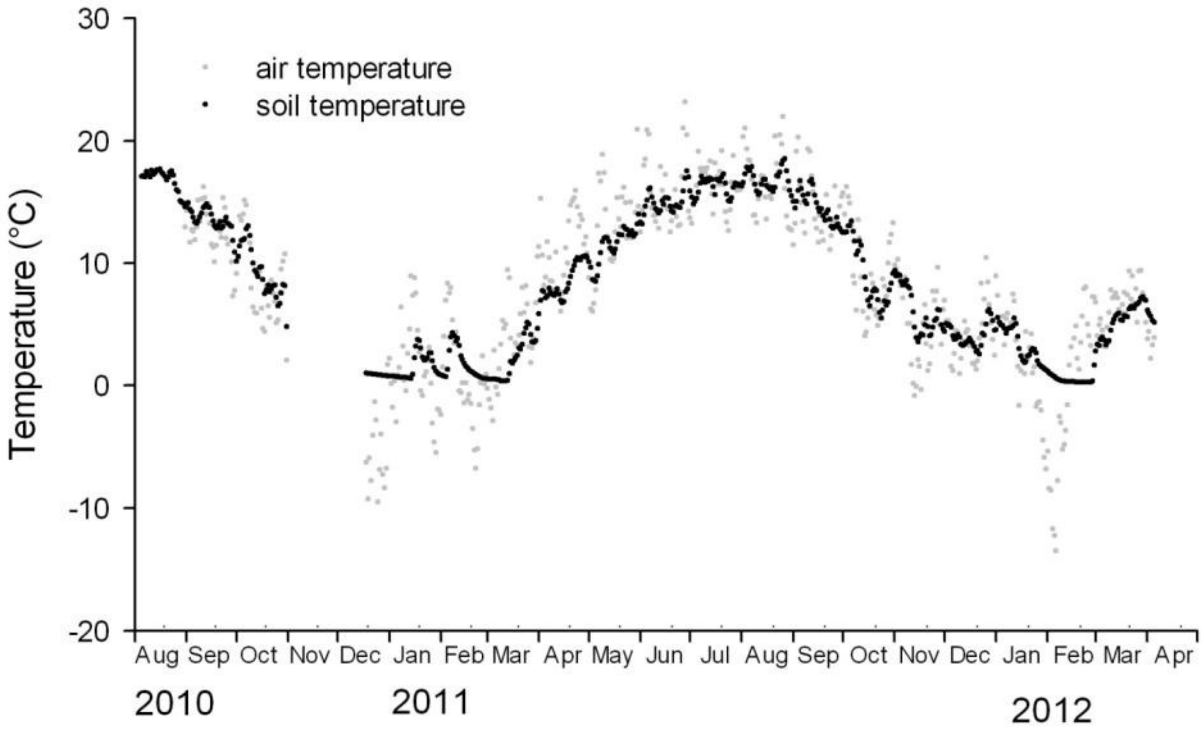


Figure 11: Comparison of mean daily air and soil temperature over the whole measurement period from August 2010 until April 2012. Soil temperature was measured at 10 cm depth. Gap: No data available.

4.1.2 Precipitation

The annual precipitation sum at the DWD-station in Quickborn in 2010 was 797 mm. The measured amount of precipitation at the investigation site in 2011 was 712 mm. In 2010, August, September and November were the months with the highest amounts of rainfall ranging between 104 mm and 123 mm. In 2011 December and August had the highest precipitation sums with 127 mm and 146 mm respectively. The driest months in 2010 were April, June and July ranging between 22 and 31 mm while the driest months in 2011 had much less precipitation: March: 10 mm, April: 8 mm and November: 8 mm (Figure 12).

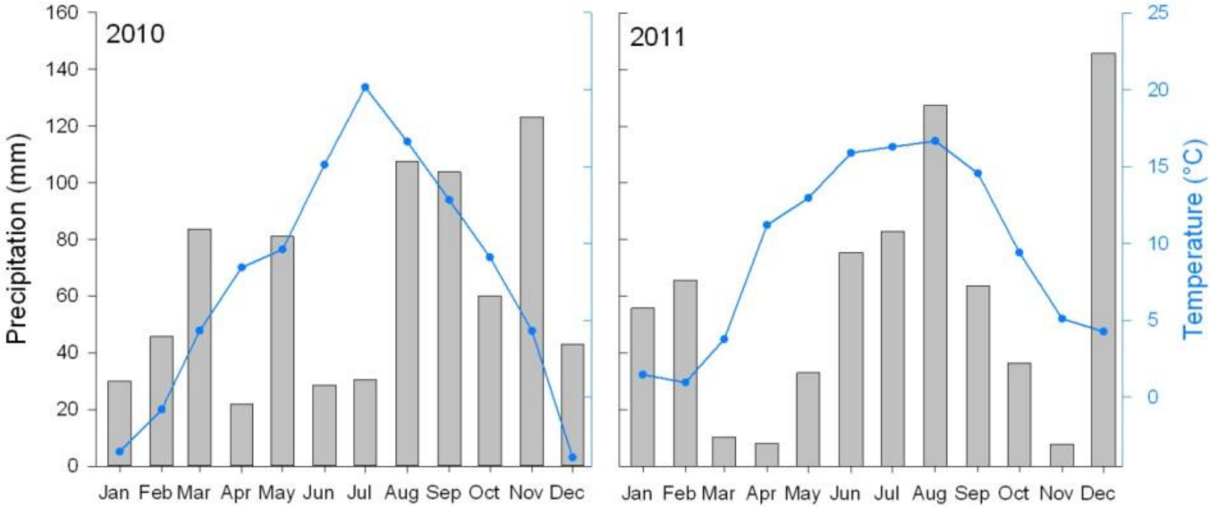


Figure 12: Monthly precipitation sums and mean air temperature of 2010 and 2011. Data from 2010 derives from the DWD-weather station in Quickborn at a distance of 3 km from the field site while 2011 displays own data from the field site.

4.1.3 Wind speed and wind direction (2011)

The main wind directions in 2011 were west and northwest and in a huge frequency additionally east, as the treeless investigation site expands in an east-west direction boarded by a wooded elevation in the south ("Knust") and a dense birch stand in the north (Figure 13). The mean wind speed in 2011 was 1.5 m s^{-1} with most of the data (95 %) ranging from 0.1-3.9 m s^{-1}, reaching a maximum value of 8.1 m s^{-1}.

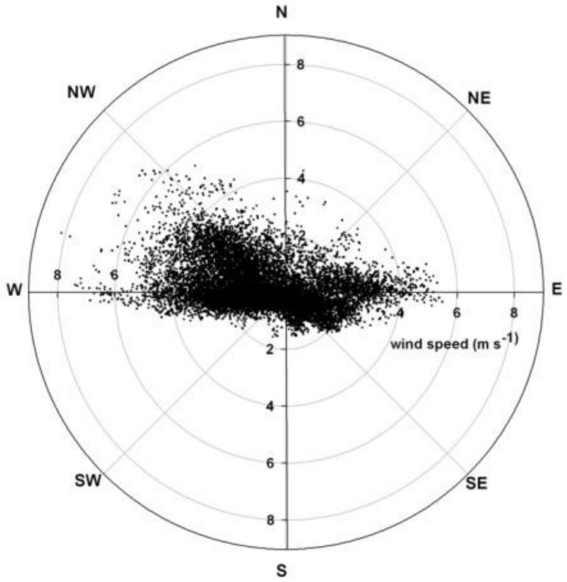


Figure 13: Wind speed and wind direction at the investigation site in the year 2011.

4.1.4 Water table

The four subsites displayed differences in water table depth (Figure 14). The *Sphagnum* subsite had the highest water table followed by heath and the purple moor grass site. The lowest water table was recorded at the drained industrial extraction site with a minimum of 55 cm under the soil surface showing a great width of water table amplitude. Contrary, the variation in the water table depth of the three vegetated sites was much less. Inundation of the soil surface appeared in the winter half year at all subsites and in summer after strong rain events at the vegetated sites.

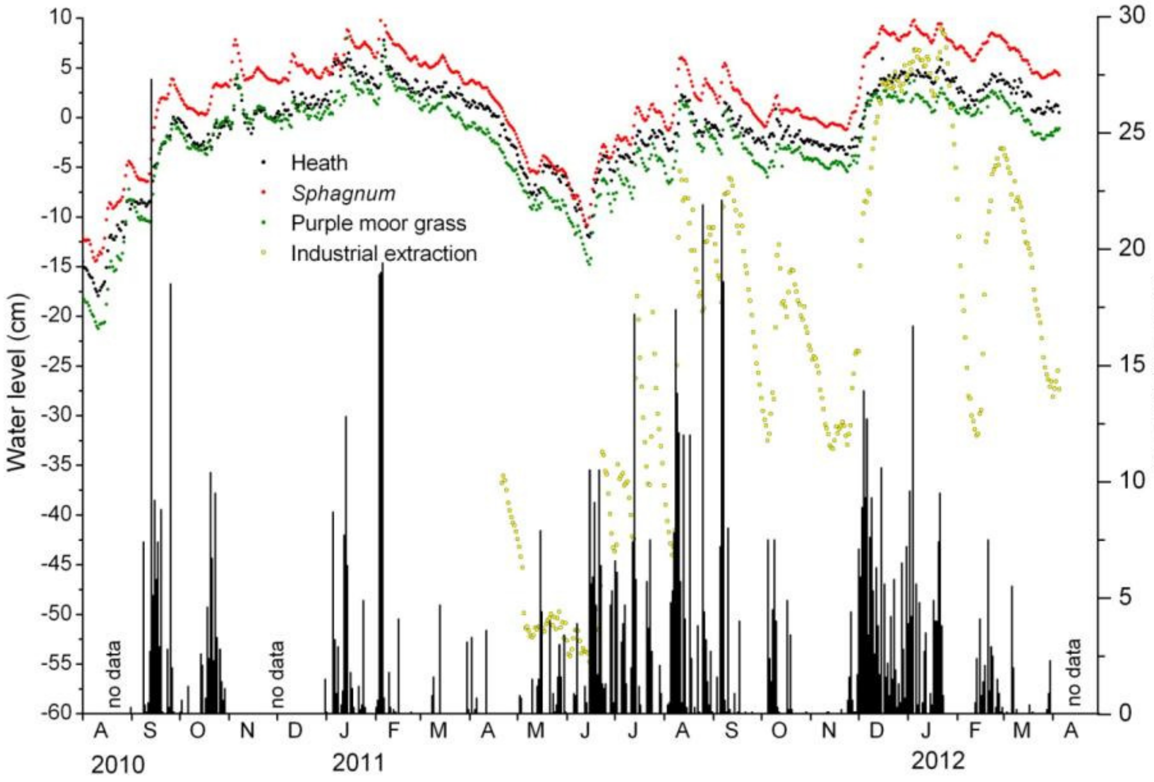


Figure 14: Mean daily water table of four different subsites in the Himmelmoor and daily precipitation sums over the whole measurement period from August 2010 until April 2012. No data available for the industrial extraction site before April 2011. Negative values indicate water levels below soil surface while positive values indicate inundation.

4.2 Evaluating side effects of rainout shelters

Precipitation

Rainout shelters kept away the whole precipitation from the drought treatment plots between Mai and September 2010 and 2011. During these 22 weeks, a total amount of 392.3 mm precipitation was retained in 2011 (Figure 15), while no data is available for 2010.

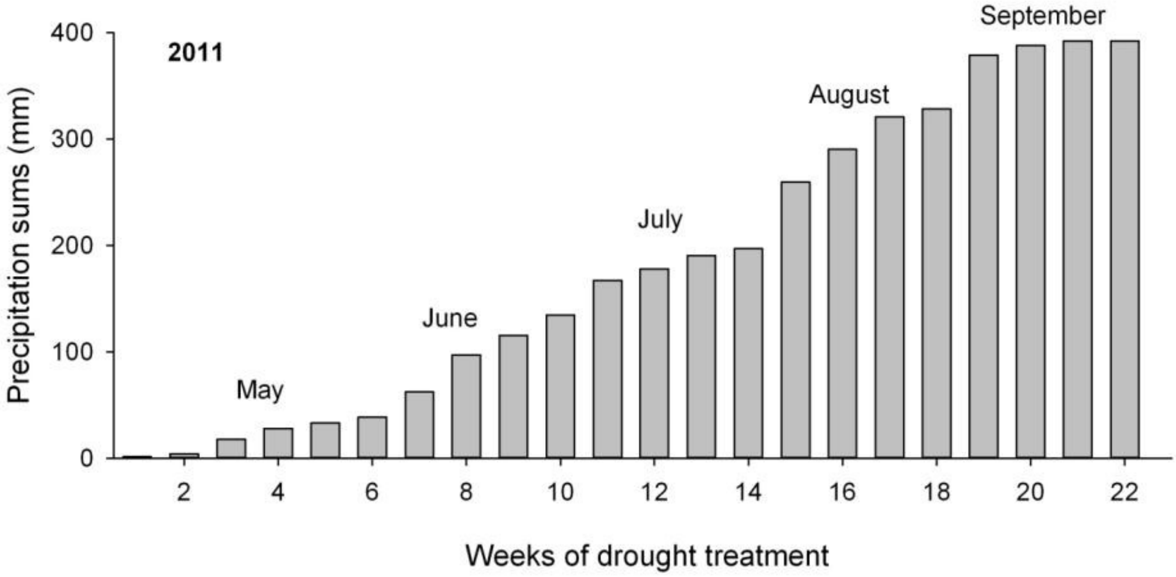


Figure 15: Cumulative, weekly precipitation sums kept away from drought treatment plots in 2011.

Water table

At one sheltered and one unsheltered plot at each subsite water table depth was measured. The drought treatment had not a clear effect on water table depth, as the water table from the drought plot is varying 1-2 cm between a higher or lower position in comparison to the control plot at each subsite (Figure 16). However, it can be seen that the decrease in water table position is sometimes continuing for one or two days longer at the drought plot while it started to increase again due to rainfall events at the control treatment. This increased the span between the two water level positions until they are nearly equalised again. This effect can especially be seen at the two rainfall events with daily sums greater than 20 mm at the end of August and the beginning of September, respectively.

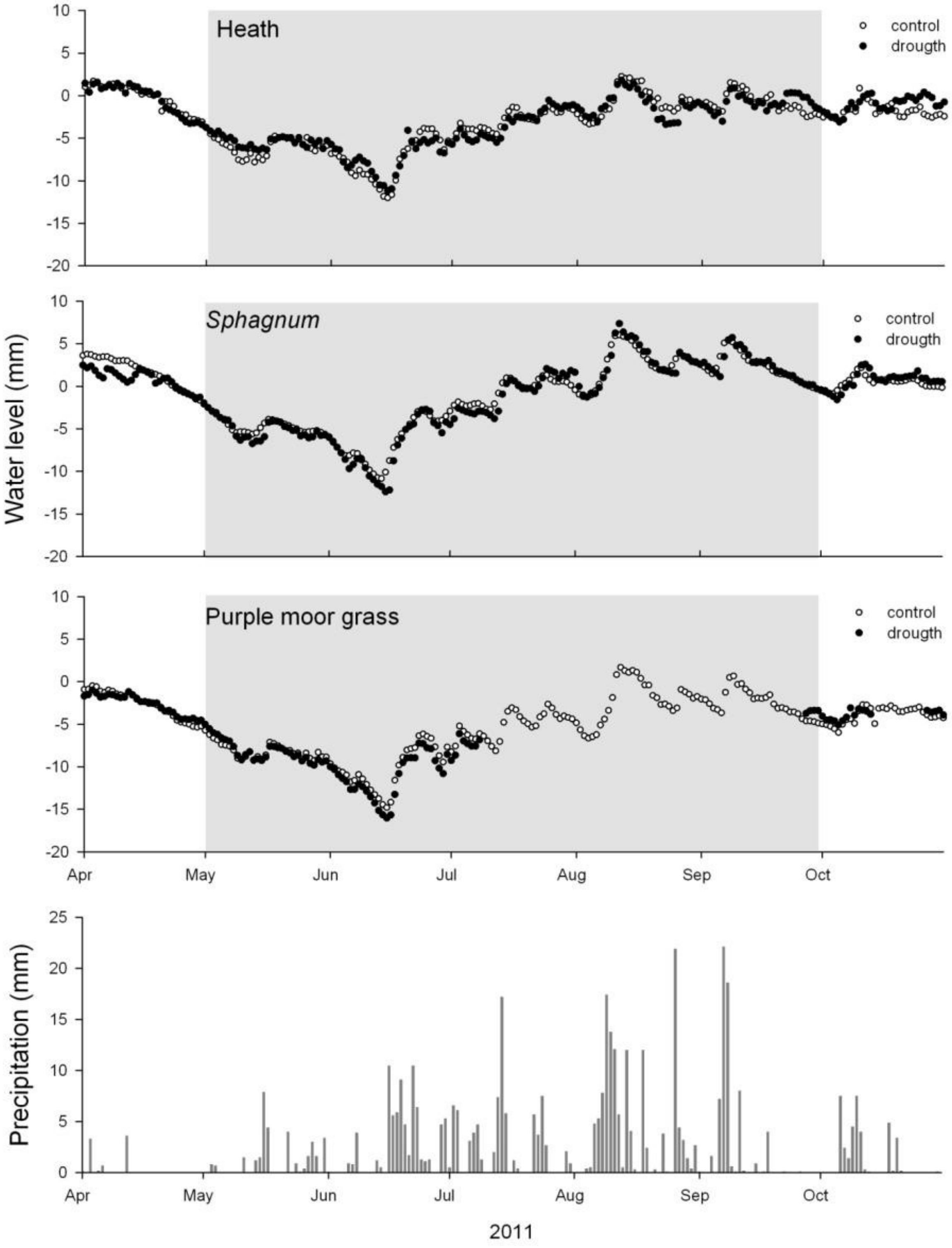


Figure 16: Impact of drought treatment on water table position at three different subsites in the Himmelmoor with differing plant communities. Grey background indicates period when shelter racks were covered with greenhouse sheets for rain exclusion. Temporally no data available at the purple moor grass site for the drought treatment. Precipitation is displayed as daily sums.

Shading

Beside inducing a summer drought, rainout shelters had a shading effect on the experimental plots, reducing the photosynthetically active radiation (PAR) under the shelter (Figure 17). The intensity of the shading effect increased with increasing insolation. In summer time the PAR under the shelters was 30 % less than the ambient PAR, while it was only 10 % less in winter time. Thus, shading appeared not only during the rain exclusion period, when shelter racks were covered with greenhouse sheets, but also during the non-covered time span.

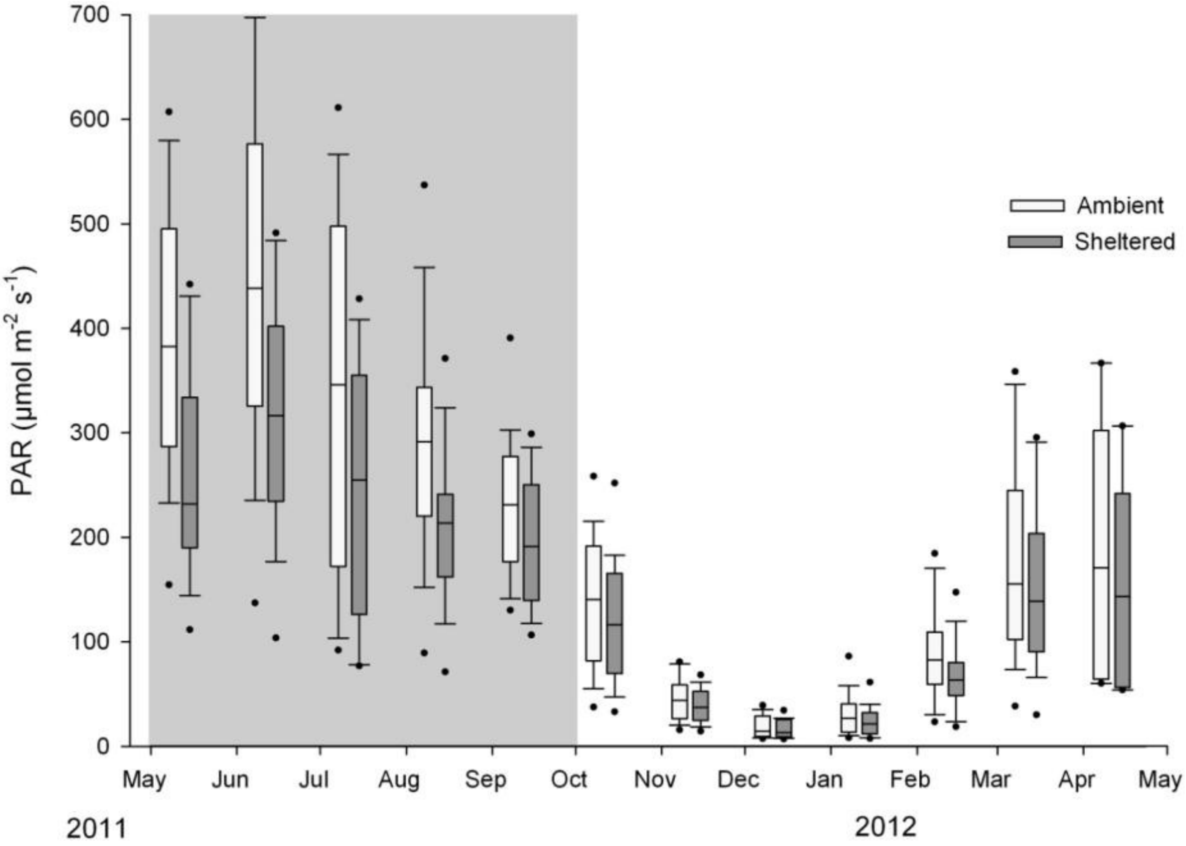


Figure 17: Effect of rainout shelter on monthly photosynthetically active radiation (PAR). Grey background indicates period when shelter racks were covered with greenhouse sheets for drought treatment. Boxes represent the interquartile range and the line inside represents the median. Whiskers denote the 10th and 90th percentile respectively. Points represent outliers.

Temperature

The mean air temperature under the shelters during the covered time span is not different from the temperature at the control plots (Figure 18). However, an interesting tendency can be seen when different temperature ranges are considered. Between 0 and 10 °C the temperature under the shelters was higher than the ambient temperature. From 10 to 20 °C this warming was reduced and between 20 and 40 °C the sheltered plot air temperatures were cooler in comparision to the control plots. This effect displays the greatest magnitude at the purple moor grass subsite.

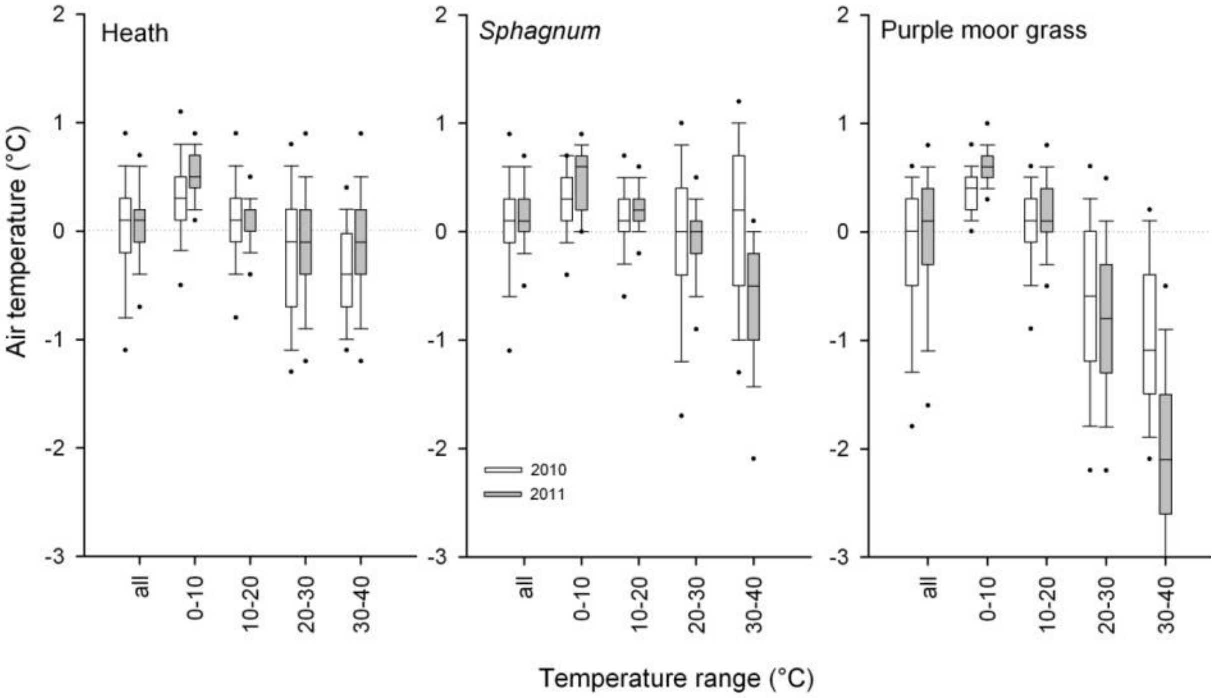


Figure 18: Deviation of sheltered air temperature from ambient air temperature for different temperature ranges in 2010 (white) and 2011(grey). A negative value indicates cooling under the shelter and a positive value indicates warming. Boxes represent the interquartile range and the line inside represents the median. Whiskers denote the 10th and 90th percentile respectively. Points represent outliers.

4.3 Vegetation analysis

The three different subsites in the Himmelmoor were chosen due to visual differences in plant communities and were named after the most prominent plant species or groups as heath, *Sphagnum* and purple moor grass subsite. The vegetation analysis of Sebastian R. Schmidt (Research group: Applied Plant Ecology, University of Hamburg) in 2010 and 2011 confirmed this approach, as differences in species cover between the three defined subsites were found (Table 7). The heath subsite is dominated by two heath species with a cover 45 %, the *Sphagnum* subsite has 99 % coverage of *Sphagnum* species and the Purple moor grass site is dominated by *Molinia caerulea* with 67 % coverage.

Subsites additionally differ in the number of species decreasing in the following order: heath, *Sphagnum* and purple moor grass with nine, eight and seven species respectively. The purple moor grass site is considerably dominated by only one vascular plant species (*Molinia caerulea)*.

No significant influence of drought treatment on species coverage and biomass was found at the heath and the *Sphagnum* site, while *M. caerulea* at the puple moor grass site had a significant increase in plant biomass between 2010 and 2011 due to drought (ANOVA, $p < 0.05$).

Table 7: Average coverage of plant species according to Londo (1976) in 2011 at three different subsites in the Himmelmoor with standard deviation of 7-8 replicates. Numbers in bold print indicate eponymous species for the subsite denotation. Data provided by Sebastian R. Schmidt (Applied Plant Ecology, University of Hamburg). * Sphagnum species were mainly S. *cuspidatum* und *S. fimbriatum.*

Plant species coverage (%)	Heath	*Sphagnum*	Purple moor grass
Sphagnum spec. *	22.6 ± 33.6	**98.9 ± 2.7**	24.2 ± 16.0
Eriophorum angustifolium	16.9 ± 7.9	45.7 ± 27	3.4 ± 3.7
Molinia caerulea	1.3 ± 1.6	18.4 ± 19.6	**66.9 ± 14.6**
Erica tetralix	**30.6 ± 17.8**	5.7 ± 7.8	not found
Calluna vulgaris	**14.4 ± 16.9**	0.9 ± 1.7	1.4 ± 3.3
Eriophorum vaginatum	5.1 ± 4.7	2.4 ± 3.5	0.1 ± 0.3
Vaccinium oxycoccus	1.6 ± 1.6	0.7 ± 1.7	not found
Andromeda polifolia	2.6 ± 3.5	not found	not found
Betula pubescens	2.0 ± 3.1	1.4 ± 2.8	1.4 ± 1.5
Drosera rotundifolia	0.1 ± 0.3	0.3 ± 0.5	0.1 ± 0.3
Juncus spec.	not found	not found	0.3 ± 0.4

4.4 Soil profiles: classification and characteristics

The soil profiles of the four subsites in the Himmelmoor were largely similar to each other. This applies especially for the soil of the three vegetated subsites, as they are located very close to each other and have the same history of utilisation. As a result of the former peat cutting, the peat stratigraphy of the heath, *Sphagnum* and purple moor grass site is mixed. This disturbed sequence of soil horizons can be seen by lens-shaped zones of peat with different attributes e.g. colour and humification (Table 8-11). Solely the stratigraphy of the soil at the industrial extraction site follows the natural order. However, the soils of all subsites can, according to the German classification system, concordantly be classified as “**Normhochmoor**”. This soil type is applied as the soil contains more than 30 % organic material, the peat layer has a depth greater than 30 cm and the ecosystem is fed by rainwater and (Ad-hoc-AG Boden, 2005).

According to the World Reference Base for Soil Resources (IUSS Working Group WRB, 2007) the soils of the four subsites in the Himmelmoor can be described as Histosols (soils with thick organic layers) applying the prefix qualifiers Fibric (two thirds ore more recognisable plant tissue) and Ombric (saturated predominantly by rainwater). Thus, the complete classification is **Fibric Ombric Histosol**. In addition, the soil of the industrial extraction site is intensively drained and in consequence the suffix qualifier “Drainic” is applied: **Fibric Ombric Histosol (Drainic).**

As more than halve of the upper 80 cm is organic, the soils in this study are classified as Histosols according to the US Soil Taxonomy (Soil Survey Staff, 2010) as well. The soil material contains more than three-fourth of plant fibres and is thus subdivided into the suborder Fibrist and the sub group **Typic Sphagnofibrist,** as fibres derive primarily from *Sphagnum*.

Table 8: Soil characteristics and soil classification of the heath subsite

Heath site (H)

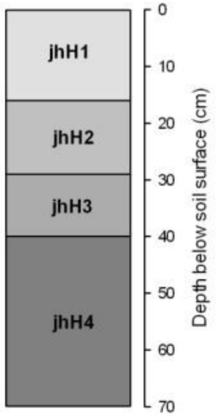
Location: Himmelmoor, Quickborn, Schleswig-Holstein, Germany
GPS coordinates: E: 53443879, N: 9504688
Date of profile acquisition: 04.11.2011
Water level during sampling: -3 cm

Remarks: Mainly moderately decomposed *Sphagnum* peat, inhomogeneous stratification by anthropogenic mixture of the peat soil. Vegetation is dominated by heath plants: *Erica tetralix* and *Calluna vulgaris*

Classification:
German Classification (Ad-hoc-AG Boden, 2005): Normhochmoor aus anthropogen umgelagerten *Sphagnum*-Torf (aus quartär-holozänem Hochmoortorf), HHn: om-Hhs(Hh-qh).

US Soil Taxonomy (Soil Survey Staff, 2010): Typic Sphagnofibrist.

WRB (IUSS Working Group WRB, 2007): Fibric Ombric Histosol.



Horizon No.	Depth (cm)	Horizon denotation*	Soil colour	Grade of Humification **	C/N	pH	Water content (%)	Dry bulk density (g cm^{-3})	**Further characteristics** (decomposition according to von Post**)
1	0-16	jhH1	7.5YR 2.5/3 7.5YR 2.5/1	H 5	42.6	3.1	93.4	0.07	Moderately decomposed S*phagnum* peat, containing many fine roots.
2	16-29	jhH2	7.5YR 2.5/2 7.5yr 2.5/1	H 5	42.1	3.1	90.0	0.10	S*phagnum* peat, partly more decomposed, containing roots.
3	29-40	jhH3	7.5YR 2.5/3 7.5YR 3/4 7.5YR 2.5/1	H 5 (H 8)	43.9	3.2	90.5	0.09	Inhomogeneous stratification of lens-shaped moderately and highly decomposed S*phagnum* peat.
4	40+	jhH4	7.5YR 2.5/3 7.5YR 3/4 7.5YR 2.5/1	H 3 (H 5)	58.8	3.3	93.7	0.06	Inhomogeneous stratification of lowly and moderately decomposed S*phagnum* peat, lens-shaped, containing cotton grass leaf sheaths.

* German classification system   ** von Post scale (von Post and Granlund, 1926), in breaks: peat lenses

Table 9: Soil characteristics and soil classification of the Sphagnum subsite.

***Sphagnum* site (S)**

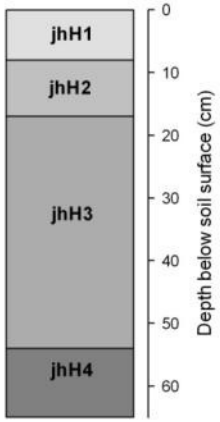
Location: Himmelmoor, Quickborn, Schleswig-Holstein, Germany
GPS coordinates: E: 53443893, N: 9504946
Date of profile acquisition: 04.11.2011
Water level during sampling: 0 cm

Remarks: Lowly up to highly decomposed *Sphagnum* and cotton grass peat with birch twigs, inhomogeneous stratification by anthropogenic mixture of the peat soil. Vegetation is dominated by *Sphagnum* mosses.

Classification:
German Classification (Ad-hoc-AG Boden, 2005): Normhochmoor aus anthropogen umgelagerten *Sphagnum*-Torf (aus quartär-holozänem Hochmoortorf), HHn: om-Hhs(Hh-qh).

US Soil Taxonomy (Soil Survey Staff, 2010): Typic Sphagnofibrist.

WRB (IUSS Working Group WRB, 2007): Fibric Ombric Histosol.



Horizon No.	Depth (cm)	Horizon denotation*	Soil colour	Grade of humification**	C/N	pH	Water content (%)	Dry bulk density (g cm^{-3})	Further characteristics (decomposition according to von Post**)
1	0-8	jhH1	2.5YR 6/4	H 2	32.0	3.3	95.3	0.05	Lowly decomposed, grown *Sphagnum* and cotton grass peat, containing birch twigs.
2	8-17	jhH2	7.5YR 2.5/1	H 8	39.8	3.1	93.4	0.07	*Sphagnum* peat, highly decomposed, containing birch twigs.
3	17-54	jhH3	2.5YR 2.5/3	H 5	57.2	3.1	93.0	0.07	Moderately decomposed *Sphagnum* peat with a small portion of cotton grass.
4	54+	jhH4	2.5YR 2.5/2	H 5	52.8	3.2	94.7	0.05	Moderately decomposed *Sphagnum* and cotton grass peat.

* German classification system   ** von Post scale (von Post and Granlund, 1926), in breaks: peat lenses

Table 10: Soil characteristics and soil classification of the purple moor grass subsite.

Purple moor grass site (P)

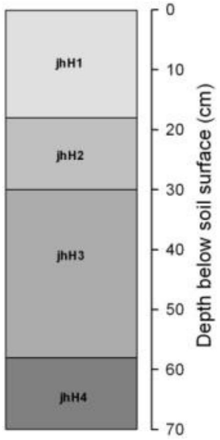
Location: Himmelmoor, Quickborn, Schleswig-Holstein, Germany
GPS coordinates: E: 53443902, N: 9505251
Date of profile acquisition: 04.11.2011
Water level during sampling: 0 cm

Remarks: Densely rooted mainly moderately decomposed *Sphagnum peat*, inhomogeneous stratification by anthropogenic mixture of the peat soil. Vegetation is dominated by *Molinia caerulea*.

Classification:
German Classification (Ad-hoc-AG Boden, 2005): Normhochmoor aus anthropogen umgelagerten *Sphagnum*-Torf (aus quartär-holozänem Hochmoortorf), HHn: om-Hhs(Hh-qh).

US Soil Taxonomy (Soil Survey Staff, 2010): Typic Sphagnofibrist.

WRB (IUSS Working Group WRB, 2007): Fibric Ombric Histosol.



Horizon No.	Depth (cm)	Horizon denotation*	Soil colour	Grade of humification **	C/N	pH	Water content (%)	Dry bulk density (g cm^{-3})	**Further characteristics** (decomposition according to von Post**)
1	0-18	jhH1	7.5YR 2.5/1	H 6	38.0	4.1	89.9	0.11	Moderately decomposed *Sphagnum* peat, containing a huge number of purple moor grass roots.
2	18-30	jhH2	7.5YR 2.5/1	H 5	48.6	3.1	90.9	0.09	Moderately decomposed *Sphagnum* peat, containing roots of purple moor grass.
3	30-58	jhH3	7.5YR 2.5/3 7.5YR 2.5/1	H 5 (H 6)	49.2	3.4	93.1	0.07	Moderately decomposed *Sphagnum* peat, containing some roots of purple moor grass and lens-shaped highly decomposed, blackish peat.
4	58+	jhH4	7.5YR 3/4 7.5YR 2.5/1	H 5 (H 7)	33.0	3.3	90.2	0.10	Moderately decomposed *Sphagnum* peat with lens-shaped highly decomposed, blackish peat.

* German classification system ** von Post scale (von Post and Granlund, 1926), in breaks: peat lenses

Table 11: Soil characteristics and soil classification of the industrial extraction site.

Industrial peat extraction site (E)

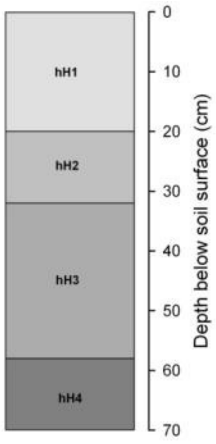
Location: Himmelmoor, Quickborn, Schleswig-Holstein, Germany
GPS coordinates: E: 53443527, N: 9505612
Date of profile acquisition: 04.11.2011
Water level during sampling: -24 cm

Remarks: Deeply drained, moderately decomposed *Sphagnum* and cotton grass peat. No vegetation cover due to active peat mining.

Classification:
US Soil Taxonomy (Soil Survey Staff, 2010): Typic Sphagnofibrist.

WRB (IUSS Working Group WRB, 2007): Fibric Ombric Histosol (Drainic).

German Classification (Ad-hoc-AG Boden, 2005): Normhochmoor aus organogenem *Sphagnum*-Torf (aus quartär-holozänem Hochmoortorf). HHn: og-Hhs(Hh-qh).



Horizon No.	Depth (cm)	Horizon denotation*	Soil colour	Grade of humification**	C/N	pH	Water content (%)	Dry bulk density (g cm^{-3})	Further characteristics (decomposition according to von Post**)
1	0-20	hH1	7.5YR 2.5/3	H 5	56.6	3.0	89.1	0.10	Moderately decomposed *Sphagnum* and cotton grass peat.
2	20-32	hH2	7.5YR 3/3 10YR 2/1	H 5	59.1	3.0	89.5	0.10	Moderately decomposed *Sphagnum* and cotton grass peat, dark colour, fibrous structure.
3	32-58	hH3	7.5YR 2.5/2 7.5YR 2.5/3	H 5	49.5	3.1	87.6	0.12	Moderately decomposed *Sphagnum* and cotton grass peat, with bright and dark horizontal stripes.
4	58+	hH4	7.5YR 2.5/2 7.5YR 2.5/3	H 5	41.7	3.4	91.1	0.09	Moderately decomposed *Sphagnum* and cotton grass peat, smeary.

* German classification system ** von Post scale (von Post and Granlund, 1926), in breaks: peat lenses

The soil profiles of the four subsites were very similar to each other and differed only slightly in physical and chemical properties (Figure 19). In the upper horizon, which is mostly affected by the different vegetation communities and water levels, slightly differences can be seen. Especially the drained and exploited surface of the industrial extraction site differs from the vegetated subsites in having the lowest N content and in consequence the highest C/N ratio. Additionally pH and ash content is low and bulk density is high compared to the vegetated subsites. Within the vegetated subsites the purple moor grass site is notable as pH, ash content and bulk density of the upper horizon were higher than at the *Sphagnum* and the heath subsite.

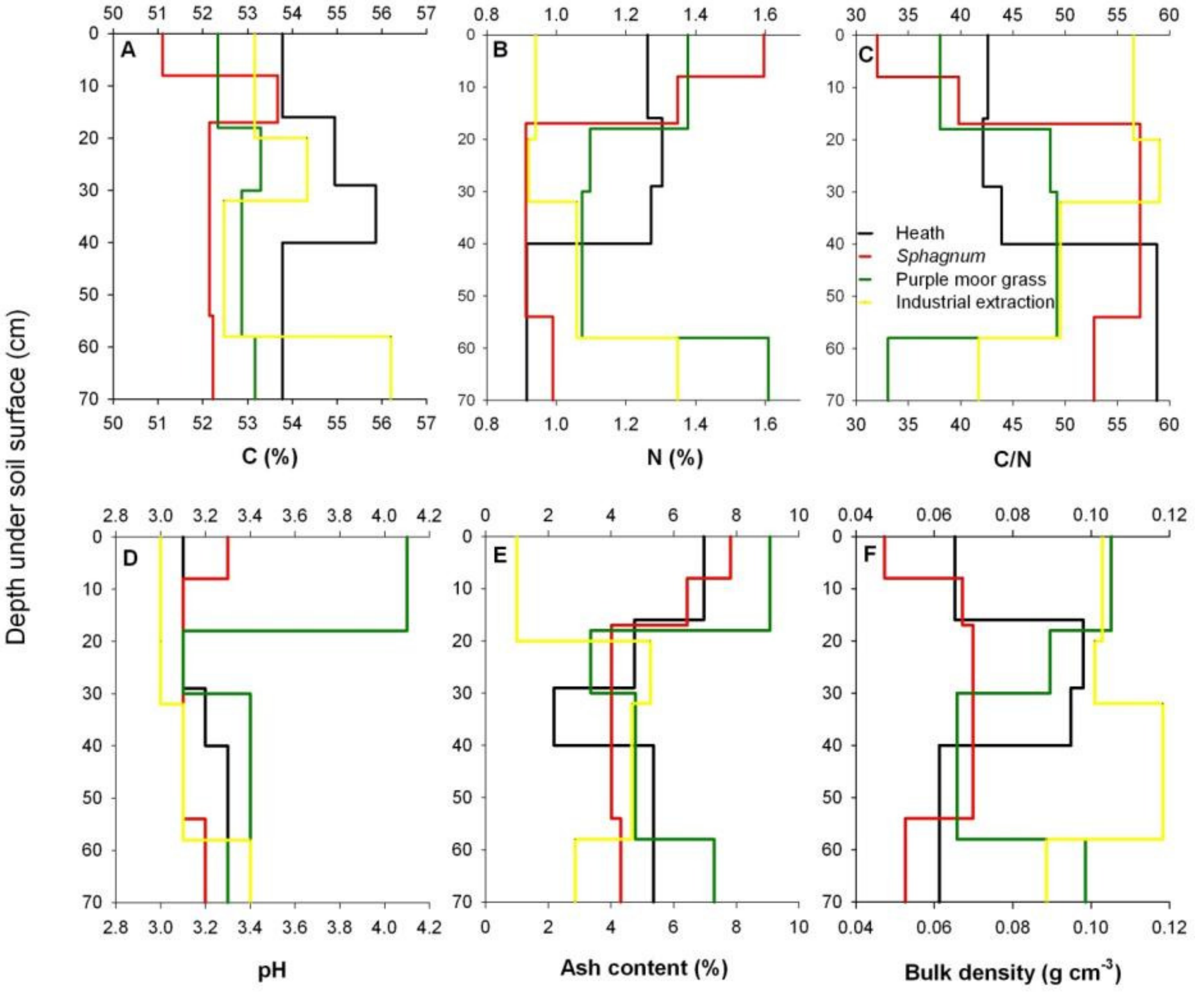


Figure 19: Physical and chemical properties of soil profiles at four different subsites in the Himmelmoor. A: Carbon content, B: nitrogen content, C: C/N ratio, D: pH, E: ash content and F: Bulk density.

4.5 Greenhouse gas fluxes different peat sites in the Himmelmoor with experimental summer drought

4.5.3 CO_2 fluxes and modelling

Measured ecosystem respiration (Reco) fluxes

Ecosystem respiration (Reco) differed highly significantly between the subsites/vegetation types (ANOVA, $p < 0.001$). Highest values were measured at the *Sphagnum* and the purple moor grass site, while Reco of the heath and the industrial extraction site were significantly lower (Figure 20). Drought treatment significantly reduced Reco at the heath ($p = 0.024$) and the *Sphagnum* ($p = 0.011$) subsite, but increased Reco at the purple moor grass site ($p < 0.001$).

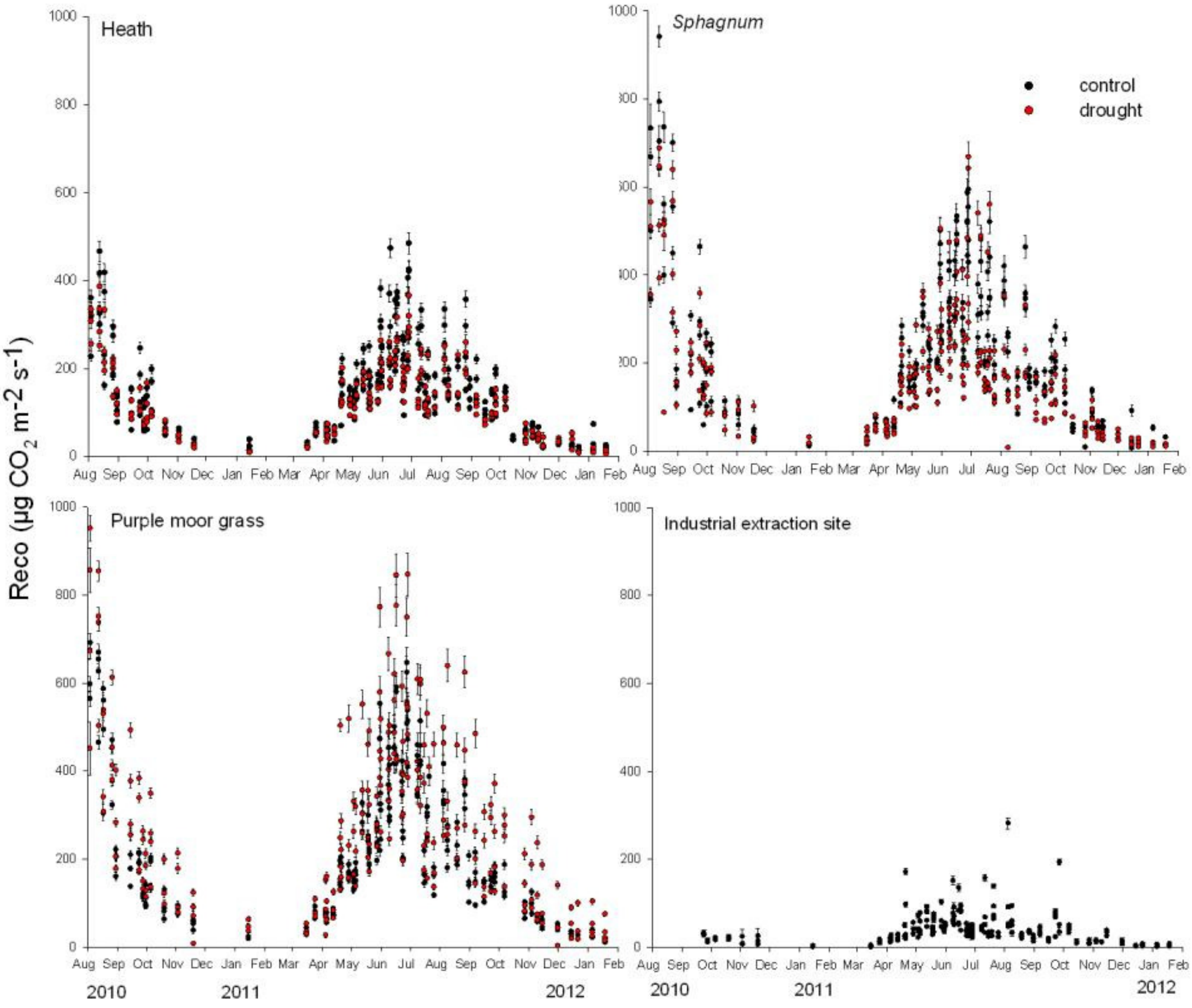


Figure 20: Measured ecosystem respiration (Reco) of 1.5 years (August 2010 until February 2012) at four different subsites in the Himmelmoor, with and without drought treatment. Dots indicate the measured fluxes ± standard error of the flux calculation.

Influence of day and night conditions on Reco

The measured Reco fluxes at day and night time differed significantly between the subsites (Figure 21, ANOVA, $p < 0.001$). Night time fluxes were higher than daytime fluxes and the variation between the replicates was higher at night time. However, day and night time differences at none of the four subsites were statistically significant ($p > 0.05$) and were thus not taken into account for Reco modeling. Air temperature and wind speed were lower in night time than at day time.

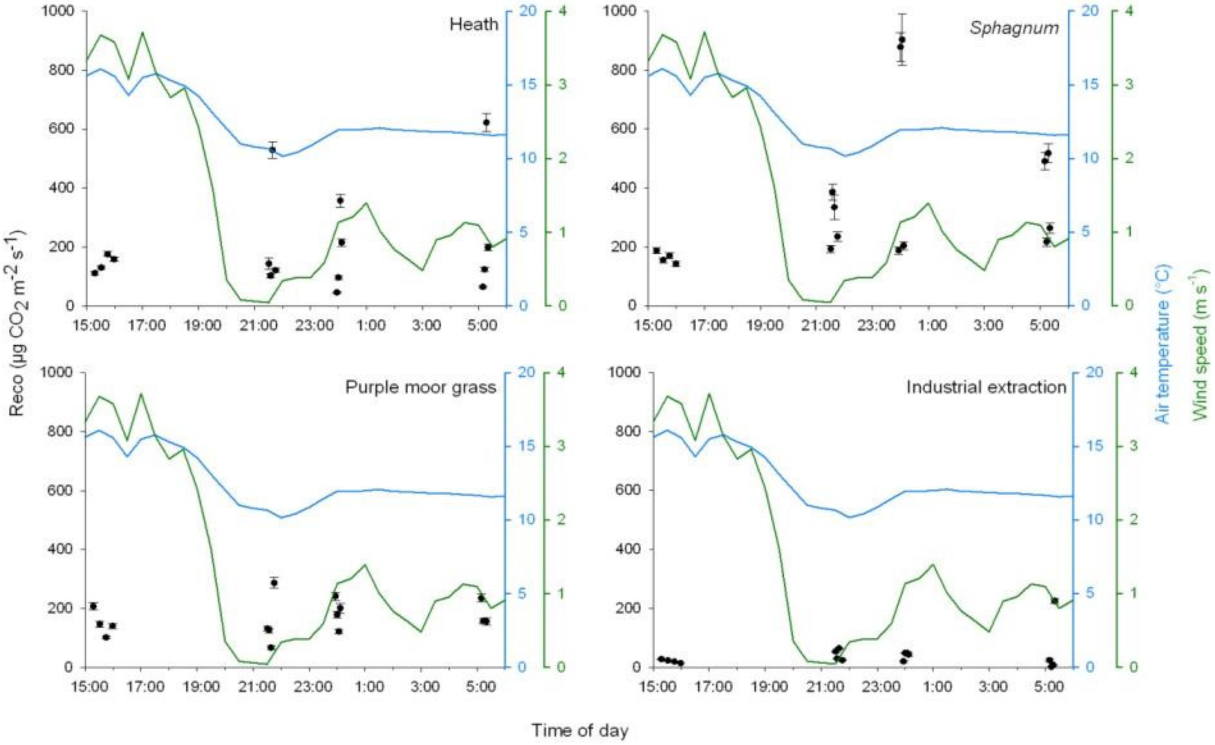


Figure 21: Variations in day and night Reco fluxes at four different subsites in the Himmelmoor measured at 30 to 31 August 2011. Replicates of the subsites were measured directly after each other. Points indicate fluxes of a single replicate ± standard error of. Green line: wind speed, blue line: air temperature.

Influence of air temperature and water table on Reco

Measured respiratory CO_2 fluxes were correlated with air temperature and water table. While Reco increased with increasing temperatures it decreased with higher water table positions (Figure 22). At the industrial extraction site this effect is less pronounced as respiratory fluxes are here generally lower than at the vegetated subsites.

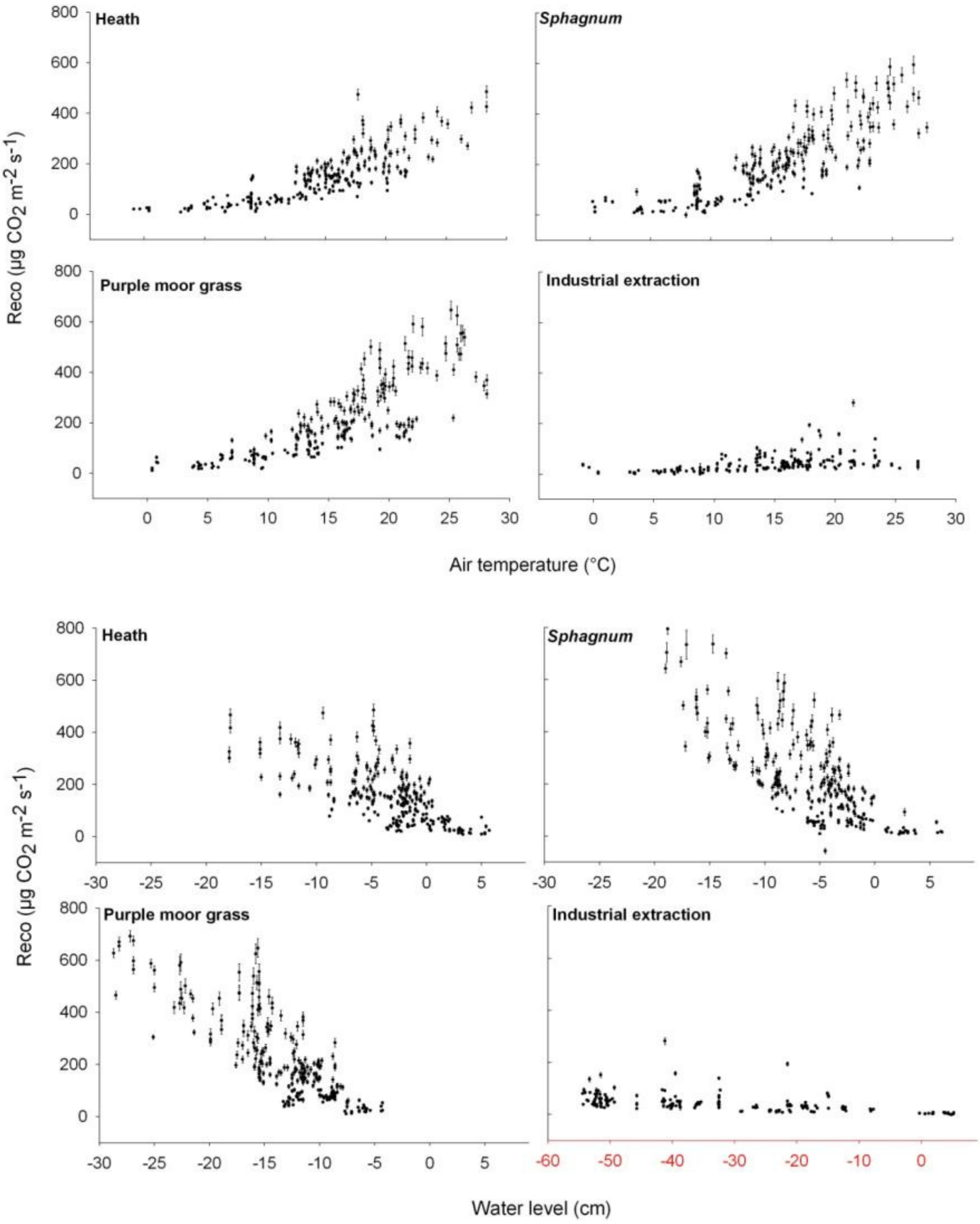


Figure 22: Dependency of measured Reco fluxes from air temperature (upper panel) and water level under soil surface (lower panel) at four different subsites in the Himmelmoor. Points indicate fluxes ± standard error of flux calculation. All four replicates per subsite are shown individually. Remark: Figure of the industrial extraction site (lower panel) has a broader X-Axis scaling.

Ecosystem respiration model (Reco)

Ecosystem respiration is better explained by air temperature than by soil temperature (Table 12 and Table 13) and was thus chosen for modelling Reco fluxes. It was shown that Reco-Model 1 (simple logistic function, chapter 3.8.2) proved to be the best way to explain the air temperature dependency of Reco fluxes in most cases by comparing reduces chi-square and $R^2_{adjusted}$ (Table 12). Consequently it was chosen as model for all plots, also if another model had better qualifying parameter for a single plot. Due to low $R^2_{adjusted}$ values of all models for the data of subsite E, a linear regression was applied for this subsite with adjusted R^2 values ranging from 0.5 to 0.7.

Table 12: Comparison of the qualifying parameter of the three different model approaches for Reco as a function of air temperature at four different subsites in the Himmelmoor. Numbers in bold print show best suited model for the plot. Subsite abbreviations: H: Heath, S: Sphagnum, P: Purple moor grass and E: Industrial extraction site.

Sub site	Treatment	Plot no.	**Reco-model 1** reduced chi-square	$R^2_{adjusted}$	**Reco-model 2** reduced chi-square	$R^2_{adjusted}$	**Reco-model 3** reduced chi-square	$R^2_{adjusted}$	**Best** model
H	control	H1	1006	0.74	1058	0.73	**1000**	**0.74**	3
H	control	H3	1811	0.78	1846	0.77	**1803**	**0.78**	3
H	control	H7	**2306**	**0.81**	2735	0.78	2378	0.81	1
H	control	H8	**3305**	**0.75**	4098	0.69	3576	0.73	1
H	sheltered	H2	**1341**	**0.74**	1403	0.73	1347	0.74	1
H	sheltered	H4	2073	0.63	2120	0.63	**2058**	**0.64**	3
H	sheltered	H5	**1347**	**0.74**	1604	0.69	1438	0.72	1
H	sheltered	H6	**1629**	**0.79**	1890	0.76	1666	0.79	1
S	control	S1	**8261**	**0.67**	8825	0.65	8469	0.66	1
S	control	S5	**4339**	**0.75**	5022	0.71	4638	0.73	1
S	control	S6	**4231**	**0.82**	4652	0.80	4361	0.81	1
S	control	S7	**4674**	**0.73**	5166	0.70	4747	0.72	1
S	sheltered	S2	**2168**	**0.68**	2694	0.60	2454	0.64	1
S	sheltered	S3	**8816**	**0.68**	9810	0.65	9254	0.67	1
S	sheltered	S4	**4186**	**0.66**	4989	0.60	4473	0.64	1
S	sheltered	S8	**3800**	**0.54**	4131	0.50	3907	0.52	1
P	control	P1	**6982**	**0.72**	7835	0.69	7333	0.71	1
P	control	P3	**5223**	**0.68**	6056	0.63	5556	0.66	1
P	control	P6	**5562**	**0.72**	6166	0.69	5929	0.70	1
P	control	P7	**8317**	**0.69**	9285	0.66	8791	0.68	1
P	sheltered	P2	**9979**	**0.71**	12563	0.64	10992	0.68	1
P	sheltered	P4	**15335**	**0.69**	16657	0.66	15585	0.68	1
P	sheltered	P5	**6651**	**0.67**	7093	0.65	6748	0.66	1
P	sheltered	P8	**4564**	**0.68**	5228	0.64	4796	0.67	1
E	control	E1	**721**	**0.33**	794	0.26	764	0.29	linear
E	control	E2	**703**	**0.22**	708	0.22	716	0.21	linear
E	control	E3	**2001**	**0.10**	1993	0.10	2027	0.09	linear
E	control	E4	**2386**	**0.03**	2391	0.03	2398	0.03	linear

Table 13: Comparison of the qualifying parameter of the three different model approaches for Reco as a function of soil temperature at four different subsites in the Himmelmoor. Numbers in bold print show best suited model for the plot. Subsite abbreviations: H: Heath, S: Sphagnum, P: Purple moor grass and E: Industrial extraction site.

Sub site	Treatment	Plot no.	**Reco-model 1**		**Reco-model 2**		**Reco-model 3**		**Best** model
			reduced chi-square	$R^2_{adjusted}$	reduced chi-square	$R^2_{adjusted}$	reduced chi-square	$R^2_{adjusted}$	
H	control	H1	**2234**	**0.51**	2253	0.51	2247	0.51	1
H	control	H3	**4169**	**0.49**	4192	0.49	4167	0.49	1
H	control	H7	392	0.97	1447	0.90	**310**	**0.98**	3
H	control	H8	5133	0.65	5231	0.64	**59**	**0.73**	3
H	sheltered	H2	2842	0.55	**2816**	**0.56**	2827	0.55	2
H	sheltered	H4	3065	0.54	**3016**	**0.55**	3048	0.55	2
H	sheltered	H5	2247	0.62	2236	0.62	**2220**	**0.63**	3
H	sheltered	H6	3876	0.59	3857	0.60	**3841**	**0.60**	3
S	control	S1	13449	0.53	**13282**	**0.53**	13408	0.53	2
S	control	S5	15833	0.48	**15577**	**0.49**	15809	0.48	2
S	control	S6	12900	0.54	**12772**	**0.55**	12786	0.54	2
S	control	S7	26304	0.40	**25880**	**0.41**	26230	0.40	2
S	sheltered	S2	**3715**	**0.45**	4393	0.35	4081	0.40	1
S	sheltered	S3	13092	0.57	13114	0.57	**13012**	**0.57**	3
S	sheltered	S4	**10074**	**0.48**	10307	0.47	10170	0.48	1
S	sheltered	S8	9746	0.35	**9590**	**0.36**	9681	0.36	1
P	control	P1	12183	0.59	**12159**	**0.59**	12190	0.59	2
P	control	P3	9172	0.62	**9024**	**0.63**	9172	0.62	2
P	control	P6	10750	0.54	**10596**	**0.55**	10745	0.54	2
P	control	P7	14522	0.56	**14295**	**0.57**	14462	0.56	2
P	sheltered	P2	**14701**	**0.63**	15525	0.61	14827	0.63	1
P	sheltered	P4	16336	0.69	16599	0.68	**16251**	**0.69**	3
P	sheltered	P5	14296	0.60	**14066**	**0.61**	14290	0.60	2
P	sheltered	P8	5746	0.67	5758	0.66	**5743**	**0.67**	3
E	control	E1	**686**	**0.36**	707	0.34	701	0.35	1
E	control	E2	**775**	**0.14**	778	0.14	782	0.13	1
E	control	E3	2009	0.10	1977	0.11	**1968**	**0.12**	3
E	control	E4	1886	0.23	1855	0.25	**1829**	**0.26**	3

Gross primary production (GPP)

Gross primary production (GPP) was calculated for each single measurement by subtraction of ecosystem respiration from net ecosystem exchange. The curvature of the measured GPP followed growing and senescence of the plants (Figure 23) with highest photosynthesis in summer time and values close to zero in winter time. A highly significant difference was found between the GPP values of all subsites (ANOVA, $p < 0.001$) displaying highest mean photosynthesis at the *Sphagnum* site although absolute values in photosynthetic uptake were higher at the purple moor grass site during summer. Drought treatment reduced photosynthetic uptake of CO_2 at the *Sphagnum* and the heath subsite, while it increased at the purple moor grass site. However, this impact is only significant at the *Sphagnum* site ($p < 0.001$).

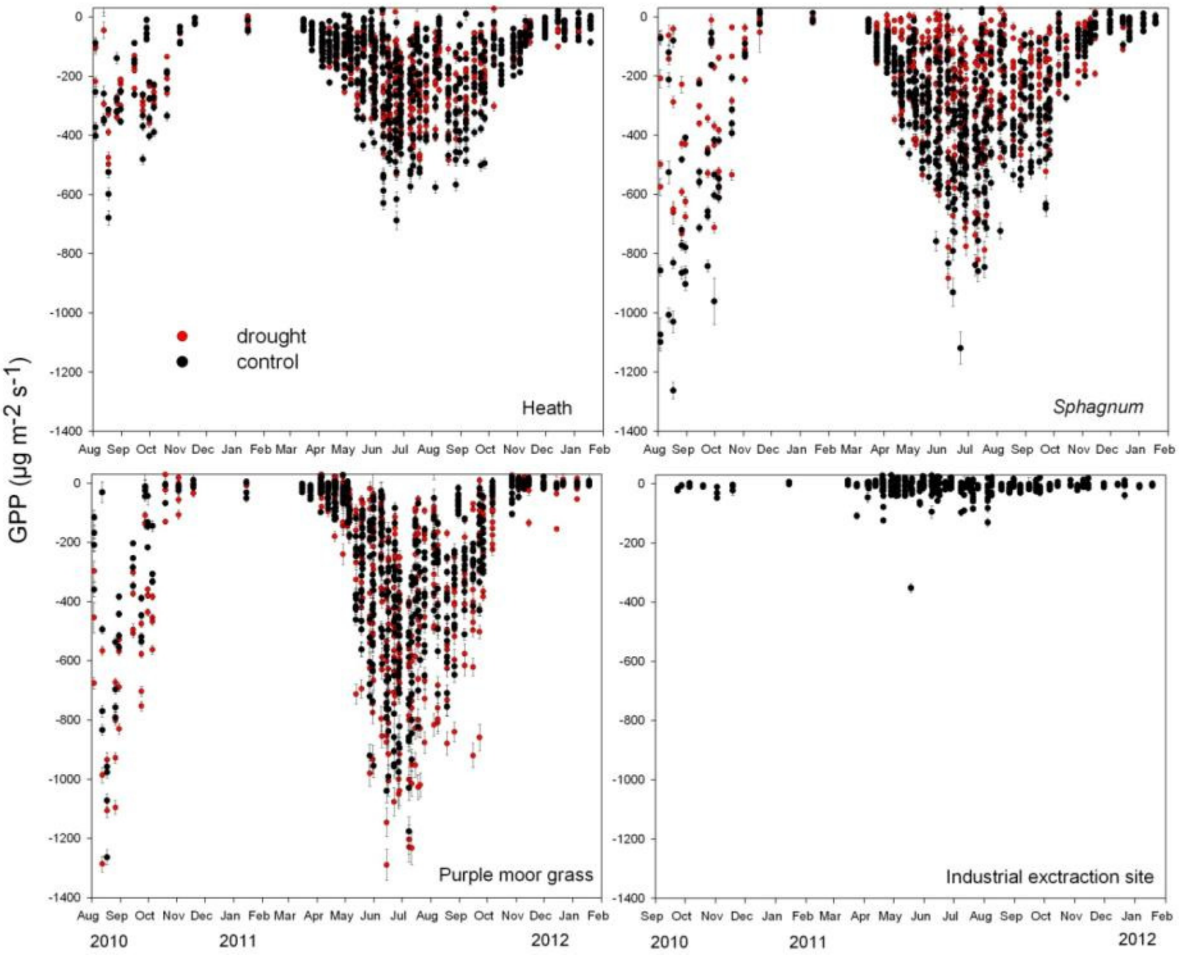


Figure 23: Calculated gross primary production (GPP) of 1.5 years (August 2010 until February 2012) at four different subsites in the Himmelmoor, with and without drought treatment. Dots indicate the measured fluxes ± standard error of the flux calculation.

Maximum photosynthetic rate (Pmax)

On the basis of measured GPP values the maximum photosynthetic rate was calculated. Maximums in potential CO_2 uptake by photosynthesis are reached depending on plant type: heath and *Sphagnum* reached most negative Pmax values at the beginning of June while at the purple moor grass site it was reached after a steep decrease at the end of June (Figure 24). A highly significant difference was found between the Pmax values of all subsites (ANOVA, $p < 0.001$) with the highest values at the purple moor grass site followed by *Sphagnum* and heath. For the heath and the *Sphagnum* subsite fitting of light response curves was possible also in winter time (October 2011 - January 2012) while it was not possible at the purple moor grass due to an abrupt decrease of CO_2 uptake (GPP) in October (Figure 23). From January until March 2011 not enough GPP data was available.

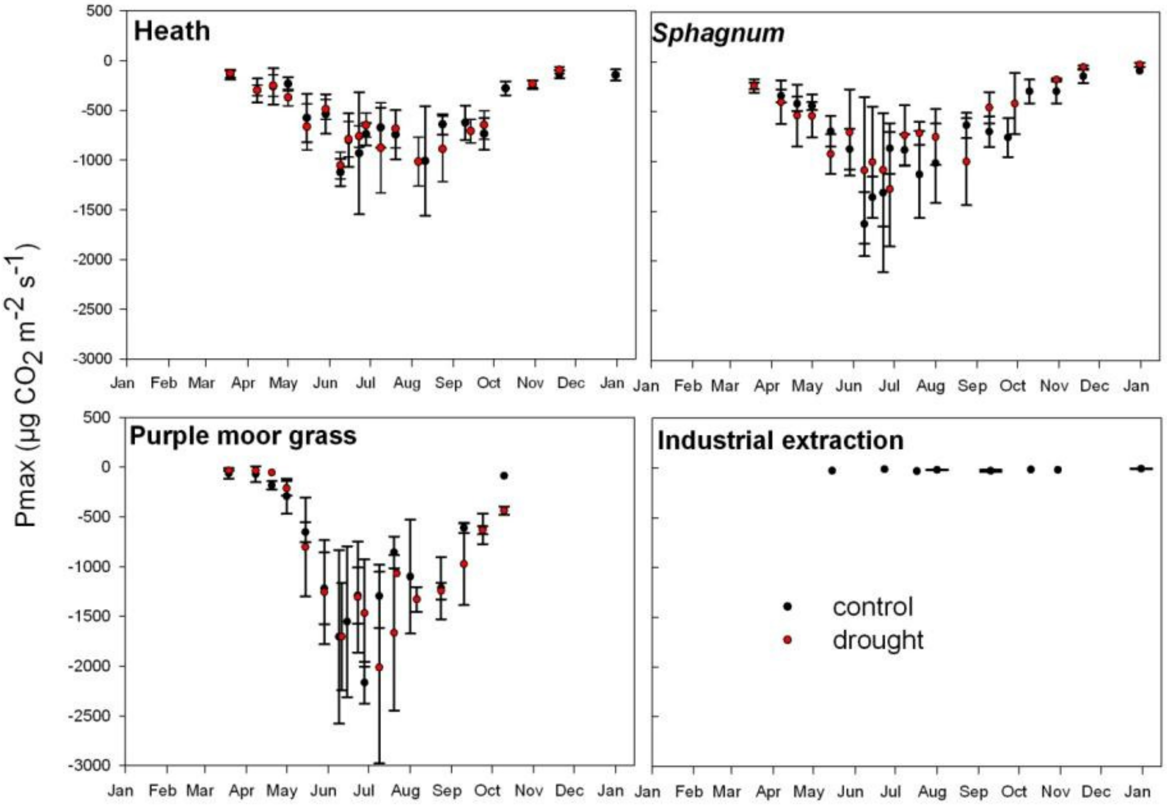


Figure 24: Calculated maximum photosynthetic rate (Pmax) of four different subsites in the Himmelmoor in 2011 with and without drought treatment. Values represent means and standard deviations from up to four replicates of each subsite. When no data points are shown in winter months, the fitting of light response curves was not possible due to low photosynthesis. Negative values indicate a CO_2 uptake.

Net ecosystem exchange (NEE)

The results of the modelled ecosystem respiration (Reco) and the modelled gross primary procuction (GPP) were summed up resulting in modelled net ecosystem exchange (Figure 25). Negative values indicate an overcompensation of Reco (CO_2 release) by photosynthesis, while positive values mean a net CO_2 flux to the atmosphere.

The temporal dynamics between the subsites is quite similar for the heath and the *Sphagnum* site. However, the *Sphagnum* site displays some more pronounced peaks in photosynthesis. Contrary the purple moor grass site is very productive in summertime and has a steep increase in photosynthesis in may, while photosynthesis at the two other vegetated subsites begins earlier and with a slower increase. Bevore the steep increase in photosynthesis respiratory fluxes distinctly predominate and increase from March until May at the purple moor grass site. The industrial extraction site is characterized by low respiratory fluxes.

The NEE dynamics within the same subsite were quite similiar to each other with two conspicuous features: S8 (drought) and P6 (control) which were not as productive as the other plots at the respective subsite. The cover of *Molinia cearulea* at plot P6 is more than on third less in comparission to the mean of the other plots at subsite P. No conspicousness in vegetation cover was present at plot S8.

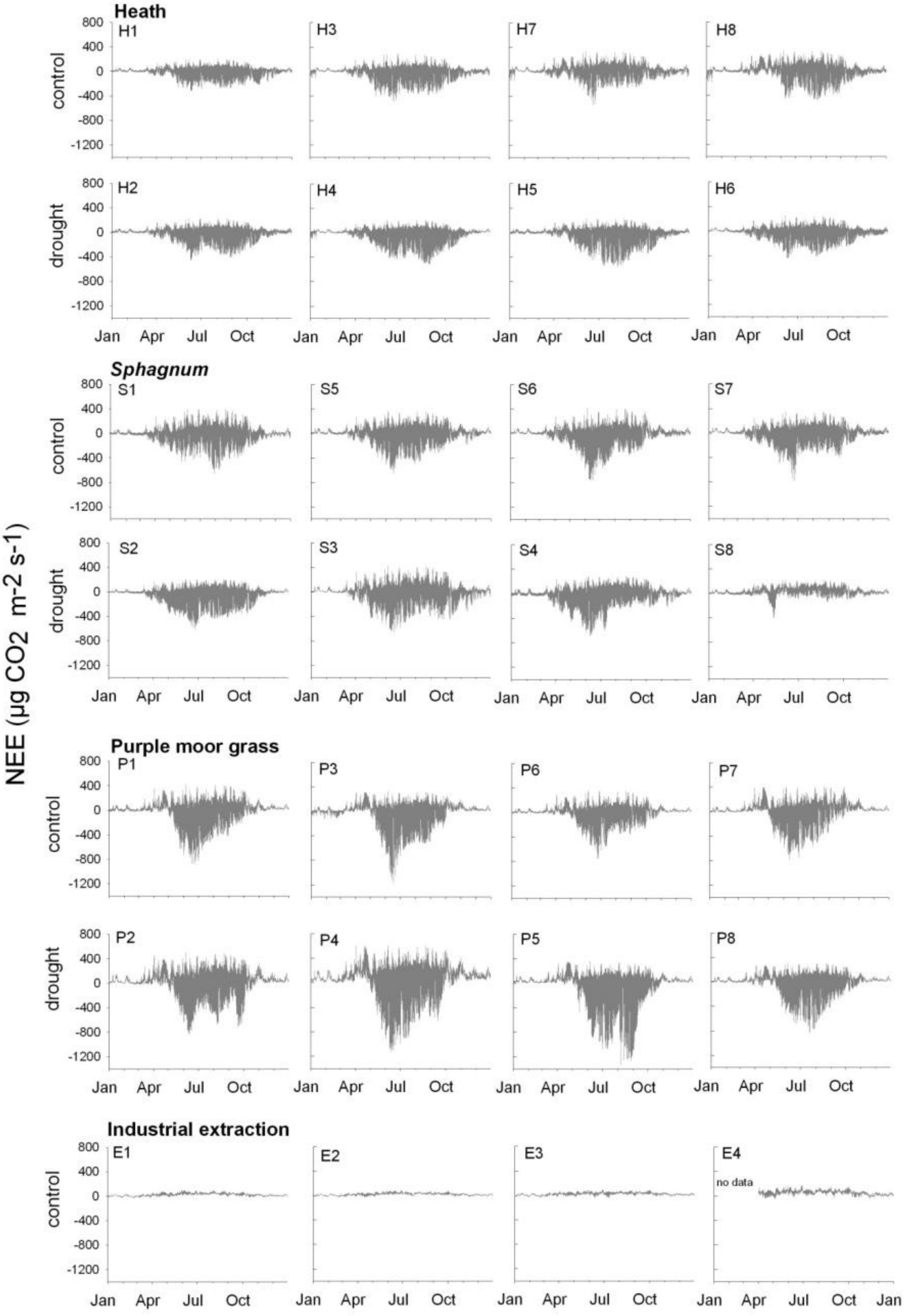


Figure 25: Modelled net ecosystem exchange (NEE) of each plot of four subsites in the Himmelmoor with and without drought treatment. Positive values represent emissions.

CO_2 budget

The annual net ecosystem exchange (NEE) was positive at all vegetated subsites as well as at the industrial extraction site (Figure 26), meaning a net CO_2 release from the ecosystem. The highest values of NEE were determined at the industrial extraction site where ecosystem respiration predominates, followed by heath, purple moor grass and finally the *Sphagnum* subsite. This difference in NEE between the subsites displayed a marginal trend toward significance ($p = 0.055$). The drought treatment had a significant effect on NEE at the heath subsite, resulting in a CO_2 net uptake ($p < 0.05$) while drought increased the amount of released CO_2 from the purple moor grass site. However, this increase was not significant ($p > 0.05$).

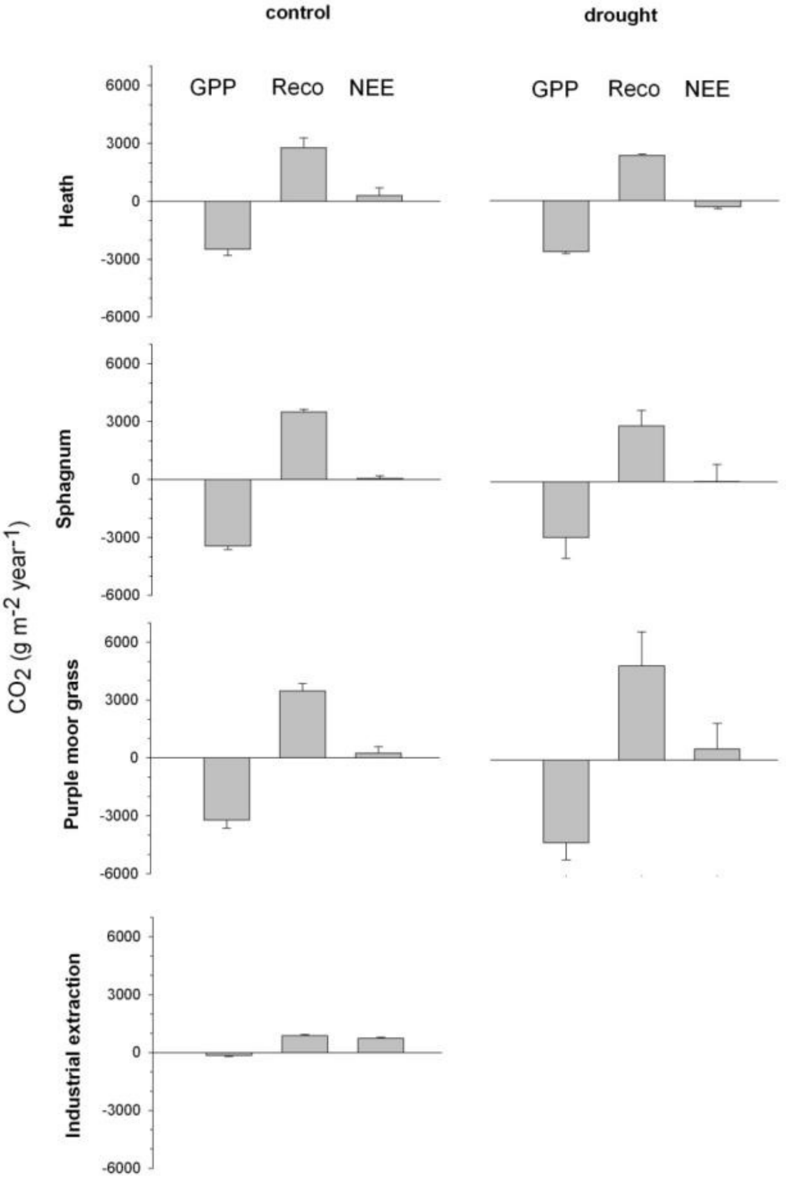


Figure 26: Annual CO_2 budget of four different subsites in the Himmelmoor with and without drought treatment for 2011. Values represent means of the model results of four replicates of each subsite ± standard deviation. Positive values represent emissions. GPP: gross primary production, Reco: ecosystem respiration and NEE: net ecosystem exchange.

4.5.1 Methane (CH_4)

Comparisson between subsites

Methane fluxes from three different vegetation communities and the industrial extraction site were measured over a hole year (April 2011 until March 2012, Figure 27). The three vegetated subsites were sources for CH_4, while fluxes of the industrial extraction site ranged around zero. The highest fluxes and the highest spatial and temporal variability were found at the purple moor grass site with 5.08 ± 11,01 µg CH_4 m^{-2} s^{-1}. The lowest fluxes with low variance were found at the industrial extraction site (0,01 ± 0,02 µg CH_4 m^{-2} s^{-1}). The heath and the *Sphagnum* subsite ranged between these.These differences between the subsites were higly significant (ANOVA, $p < 0,001$). A pairwise comparison showed highly significant differences in the CH_4 fluxes between the purple moor grass site and the heath site ($p < 0,01$), between the purple moor grass site and the industrial extraction site ($p < 0,001$,) and between the *Sphagnum* site and the industrial extraction site ($p < 0,001$, Holm-Sidak test). Other pairwise comparisons of the subsites were not statistically significant ($p > 0.05$).

According to AICc the majority (74 %) of CH_4 flux curves was better fitted with a linear regression than with an exponential model.

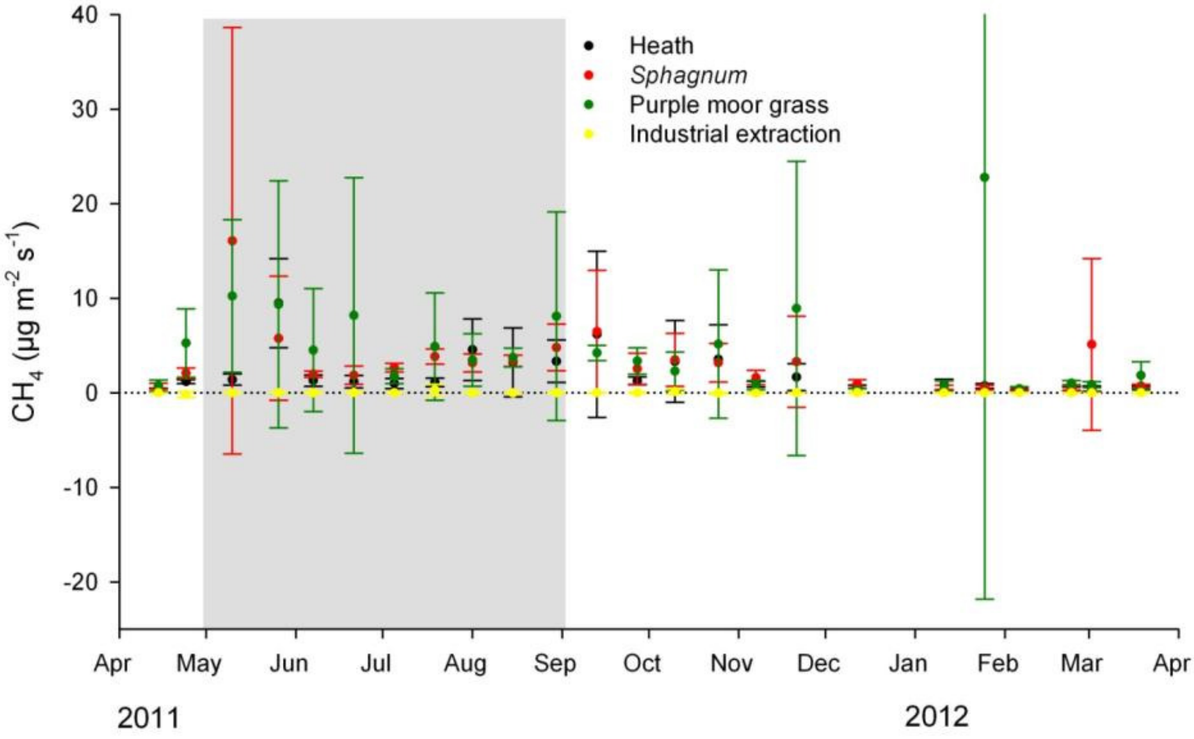


Figure 27: Comparison of CH_4 fluxes between three different vegetation communities and the industrial extraction site in the Himmelmoor over one year (April 2011 to March 2012). Values represent means ± standard deviation of four replicate plots per subsite.

Impact of summer drought on CH_4 fluxes

No significant effect of drought on the CH_4 fluxes at all subsites was detectable (Figure 28, ANOVA). Even though the fluxes of the sheltered plots of the purple moor gras subsite were most of the time less than those of the control plots (Figure 28, lower panel), the difference displayed only a near to significant trend (p = 0.06) due to high spatial variability.

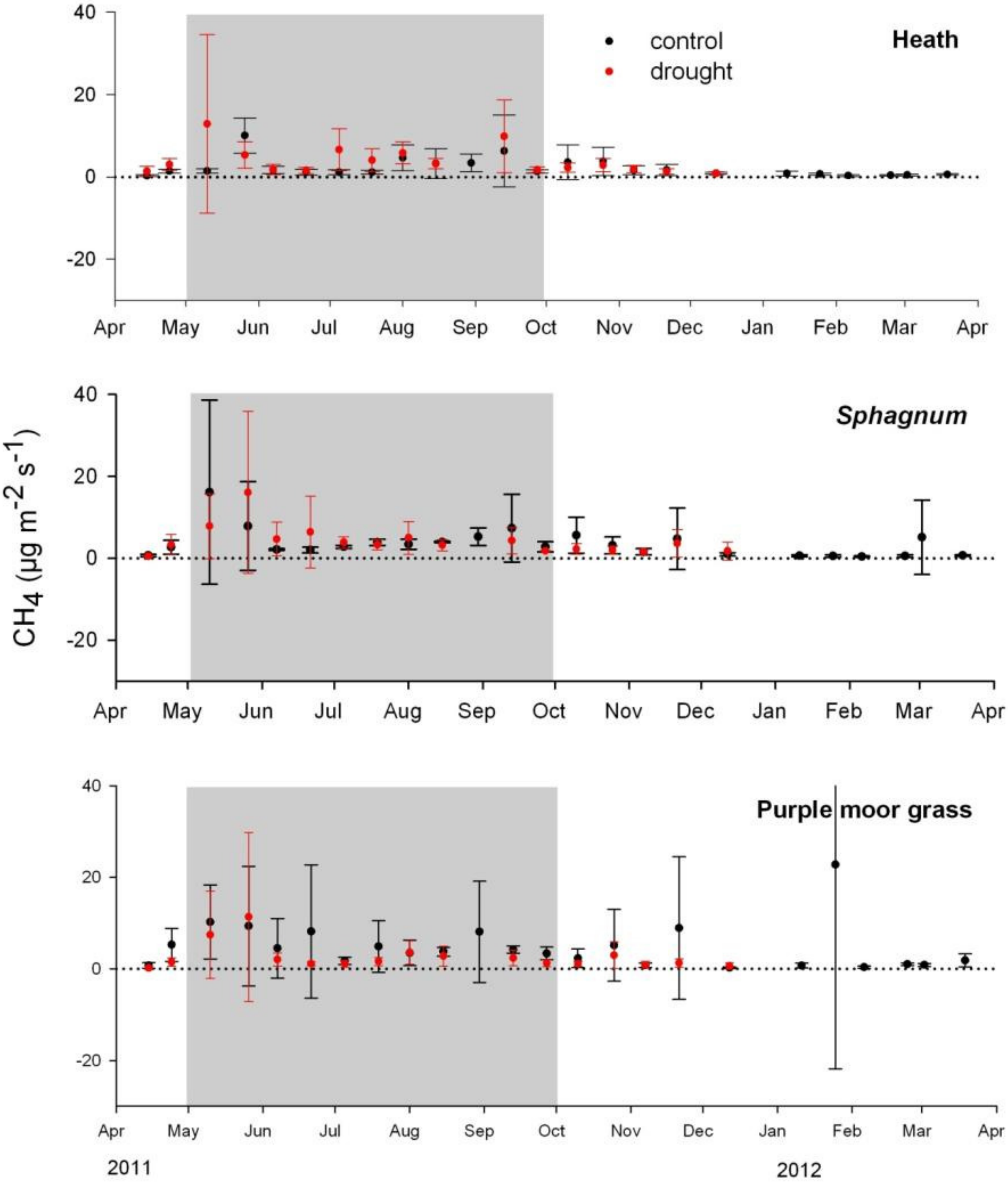


Figure 28: Impact of drought treatment on CH_4 fluxes of three different vegetation communities in the Himmelmoor. Values represent means and standard deviations of four replicate plots per treatment. Grey background indicates period of rainfall exclusion.

Influence of environmental conditions on CH_4 fluxes

Methane fluxes showed an increase with increasing coverage of vascular plants with aerenchyma, called shunt species (Figure 29). No significant difference between day and night at all of the four subsites (Figure 30) was found (ANOVA, $p > 0,05$).

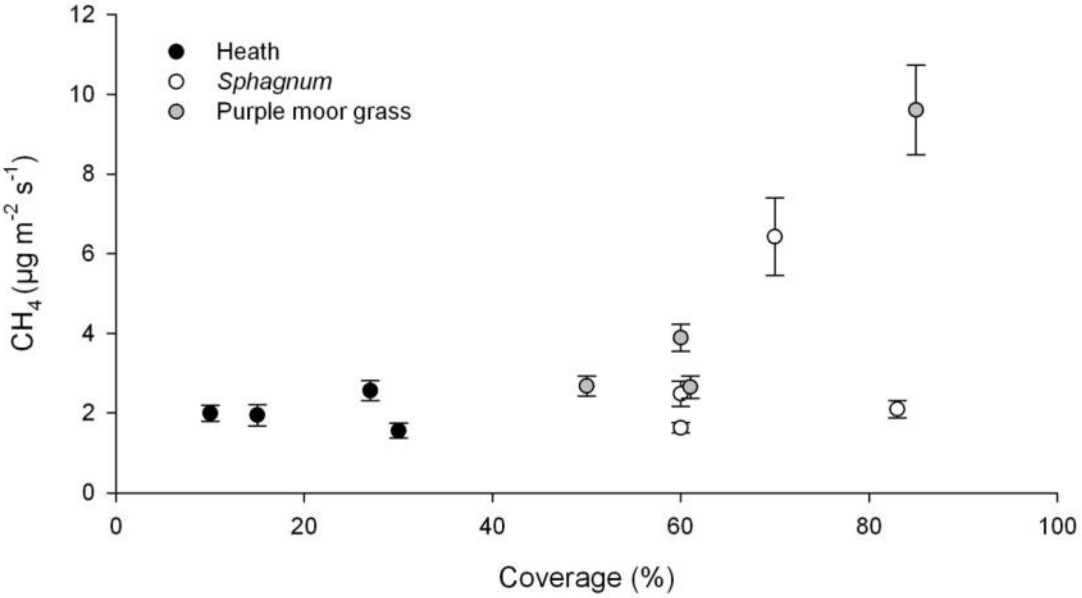


Figure 29: Dependency of CH_4 fluxes from "shunt species" coverage (sum of *M. caerulea*, *E. vaginatum*, *E. angustifolium* and *Juncus spec.*) at three different vegetation communities in the Himmelmoor. Vegetation data provided by Sebastian R. Schmidt (Applied Plant Ecology, University Hamburg).

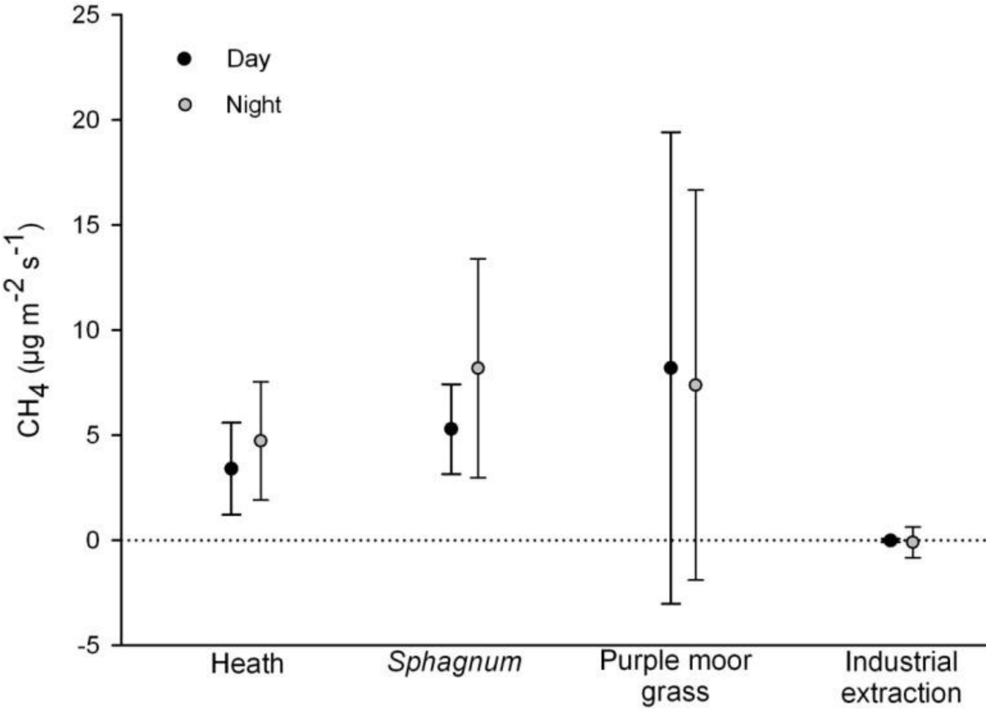


Figure 30: Comparison between day and night CH_4 fluxes at three different vegetation communities and the industrial extraction site in the Himmelmoor measured at 30 to 31 August 2011. Values represent means and standard deviations of four replicate plots per subsite.

Methane fluxes did not show a clear soil temperature or water table depth dependency (Figure 31). Thus, modelling with the classical model of Saarnio et al. (1997) as described in chapter 3.7.3, did not show sufficiently good regression coefficients. Regression coefficients for soil temperature and water table depth ranged from positive to negative values, meaning that an increase in soil temperature or water table depth could either cause an increase or decrease of CH_4 fluxes. The same applies to water tension and Pmax (data not shown). Thus, for the calculation of annual CH_4 emissions the mean was used instead.

Table 14: Mean annual CH_4 fluxes ± standard deviation at four different subsites in the Himmelmoor between April 2011 and March 2012 with and without drought treatment.

Subsite	Treatment	CH_4 (µg m^{-2} s^{-1})
Heath	control	2.02 ± 0.41
	drought	3.17 ± 1.28
Sphagnum	control	3.16 ± 2.20
	drought	3.52 ± 1.82
Purple moor grass	control	4.71 ± 3.32
	drought	2.54 ± 2.26
Industrial extraction	control	0.01 ± 0.02

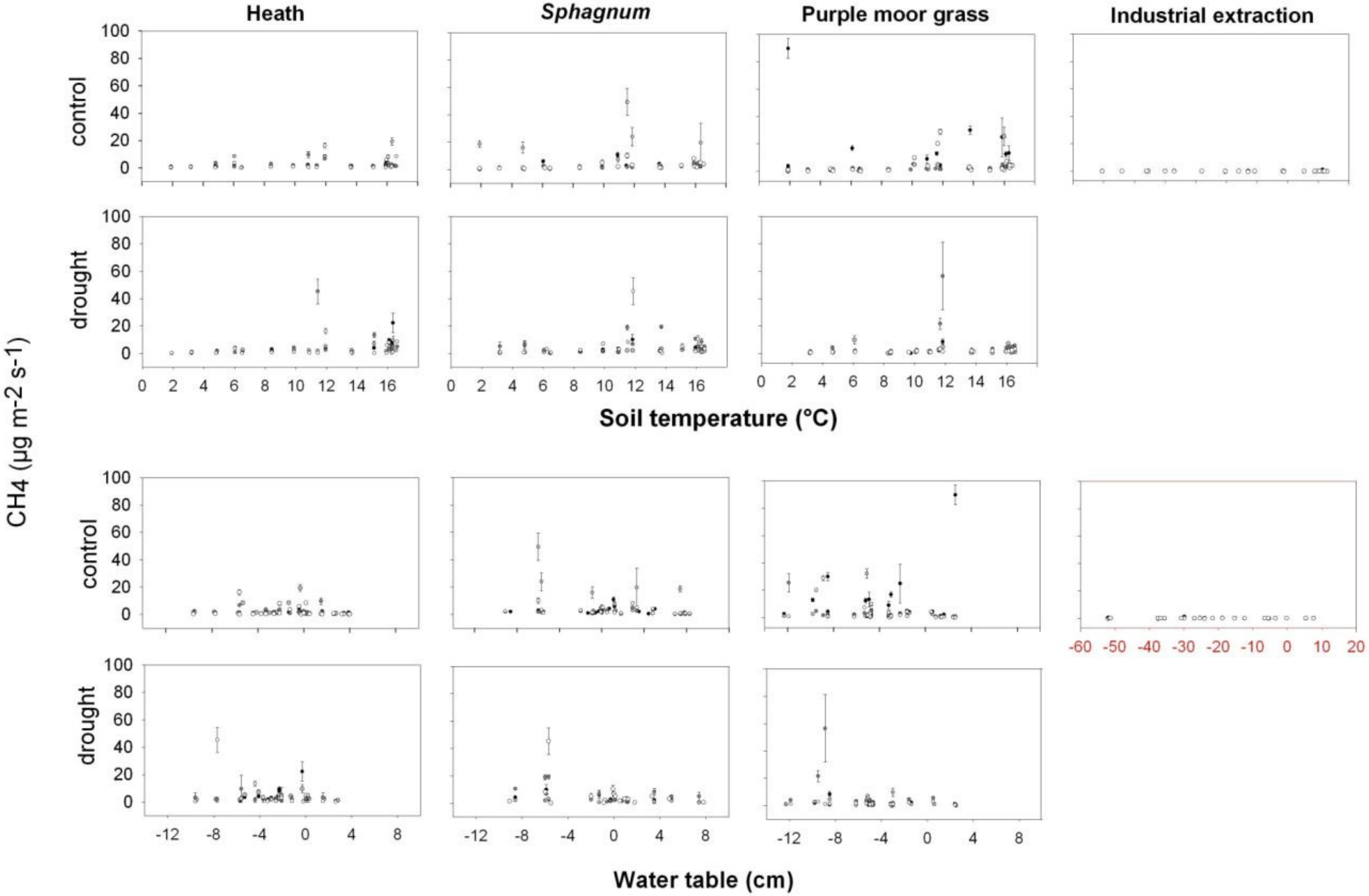


Figure 31: Methane fluxes at four different subsites in the Himmelmoor in dependency of soil temperature (upper panel) and water table under soil surface (lower panel) with and without drought treatment. Data of all replicate plots per subsite are demonstrated together. Remark: Graph of the industrial extraction site (lower panel) has a broader X-axis scaling.

4.5.2 Nitrous oxide (N_2O)

Comparisson between subsites

N_2O fluxes from three different vegetation communities and the industrial extraction site were measured over a hole year (April 2011 until March 2012) .The measured N_2O fluxes were very low, ranging around zero (Figure 32) and were thus varying between small uptakes and emissions over the whole measurement year. Only the industrial extraction site was a significant source of N_2O during summer time (0.07 ± 0.05 µg N_2O m^{-2} s^{-1}), but not over the whole year of measurement (0.02 ± 0.04 µg N_2O m^{-2} s^{-1}). Due to the N_2O emissions of the industrial extraction site in summer time, the N_2O fluxes were highly significantly differend between the subsites (ANOVA, p = 0,002) and the date of measurement (p < 0,001).

As most of the fluxes ranged around zero, only 12 % of the measurements showed a significant increase or decrease of N_2O (p < 0,05) inside the chamber headspace (Figure 33). This percentage was higher within the dataset of the industrial extraction site (26 %). According to AICc the majority (98 %) of N_2O flux curves was better fitted with a linear regression than with an exponential model.

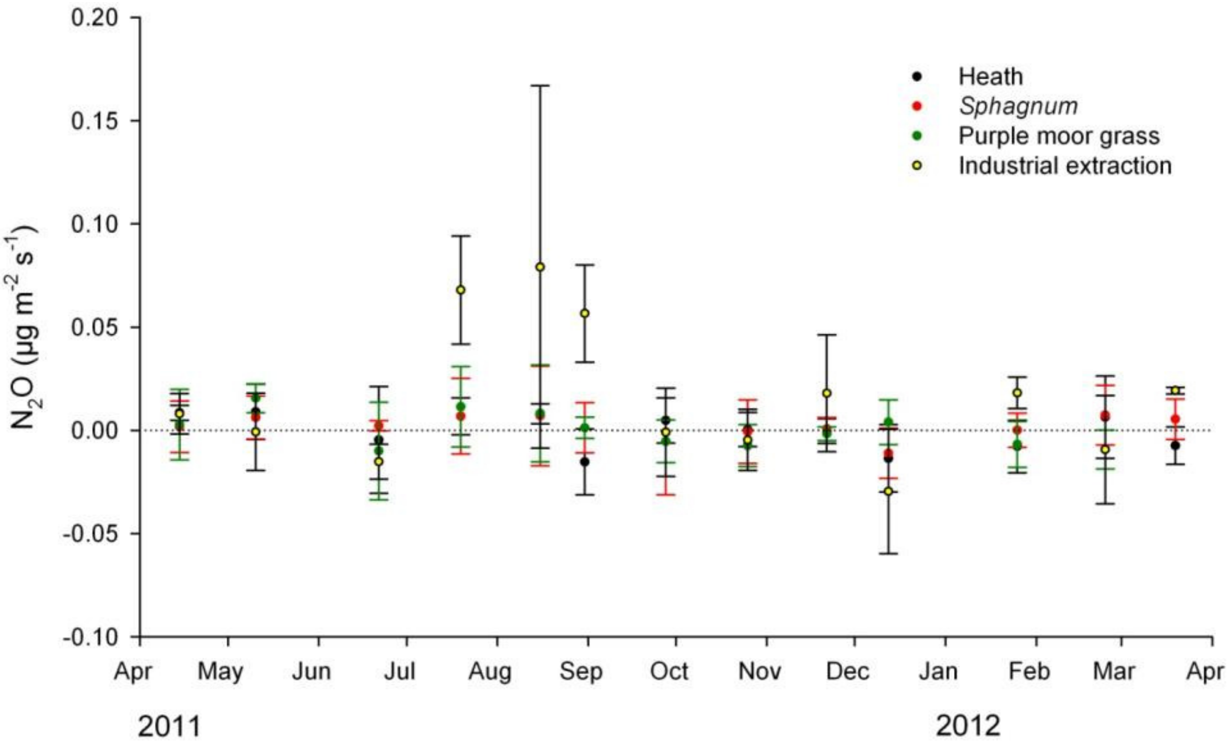


Figure 32: Comparison of the measured N_2O fluxes between four subsites in the Himmelmoor from April 2011 until March 2012. Values represent means and standard deviations of four replicate plots per subsite.

Impact of summer drought on N_2O fluxes

No significant impact of the drought treatment was detectable at all subsites, also if only the sheltered period was considered for the calculation (Figure 32).

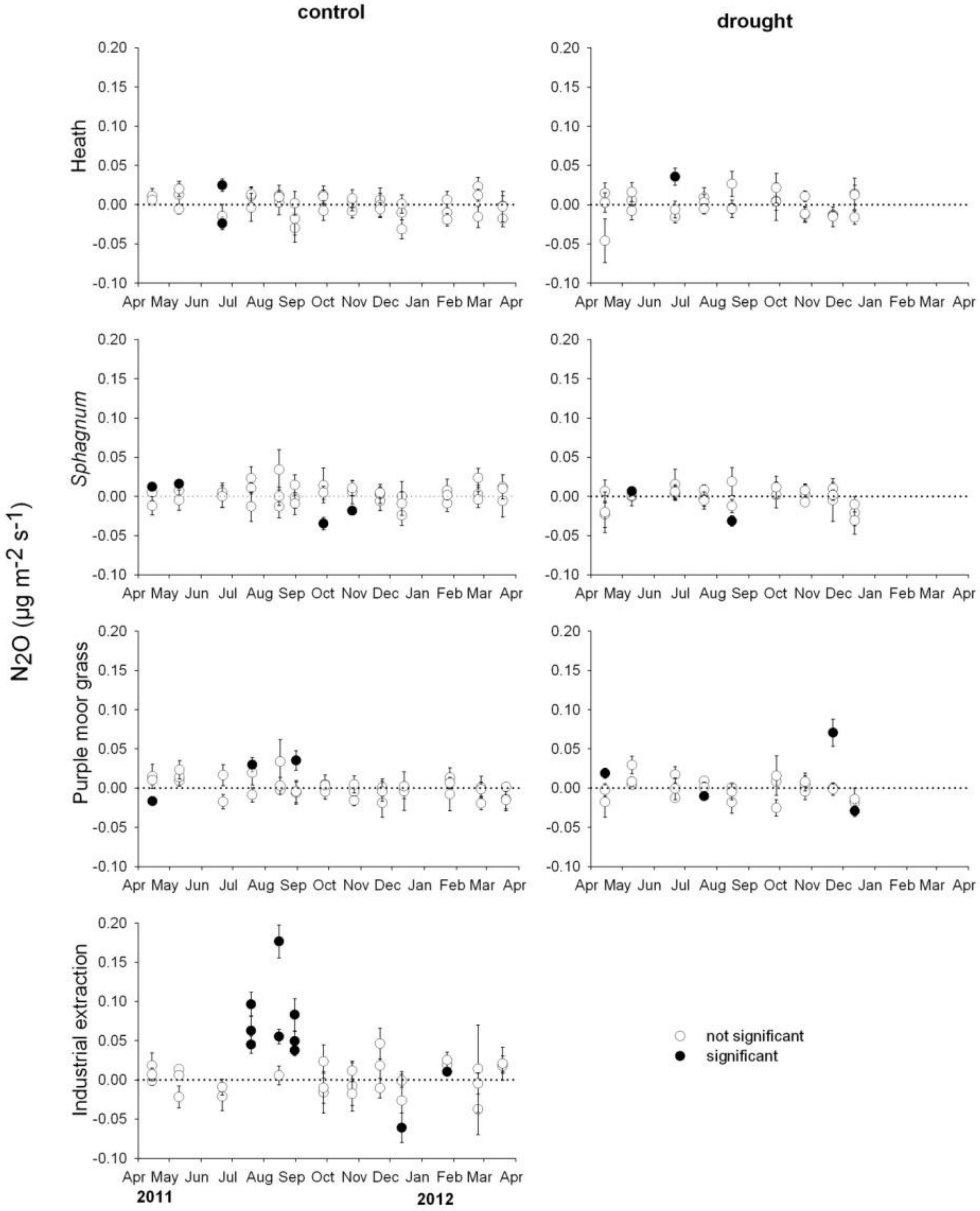


Figure 33: Comparison of N_2O fluxes at four different subsites in the Himmelmoor with and without drought treatment from April 2011 until March 2012. All three replicates per measurement day are shown individually. Vertical bars indicate the standard error of the of the flux calculation. Significant fluxes are marked black.

Influence of environmental conditions on N_2O fluxes

As most of the fluxes were not significant and ranged around zero, no correlation with soil temperature and water table can be made. Solely at the industrial extraction site, a slightly soil temperature dependency of the significant summer fluxes was seen (Figure 34). Here, at three measurement events in summer the soil temperature was higher than 15 ℃ but no dependency of water level was evident as soil was facing a huge amplitude of water table levels (-5 cm up to -29 cm under soil surface).

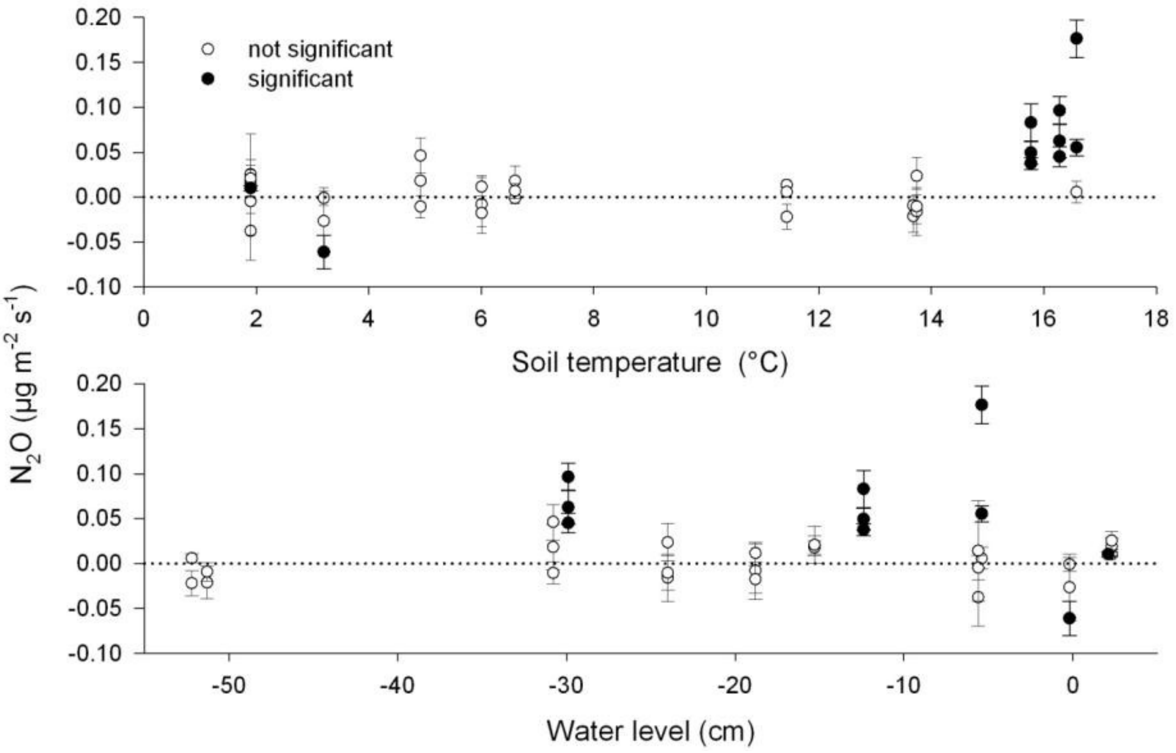


Figure 34: N_2O fluxes at the industrial extraction site in the Himmelmoor in dependency of soil temperature and water level. Significant fluxes are marked black.

Nitrous oxide fluxes were not different between day and night time (Figure 35), but differed highly significant between the subsites (two-way ANOVA, $p < 0,001$) displaying that the industrial extraction site was a significant N_2O source at day and at night time at this measurement event (30/31 August 2011). N_2O fluxes of the industrial extraction site differed highly significantly from all of the vegetated subsites ($p < 0,001$ each) while there is no difference within the vegetated subsites ($p > 0,05$, Holm-Sidak post hoc test).

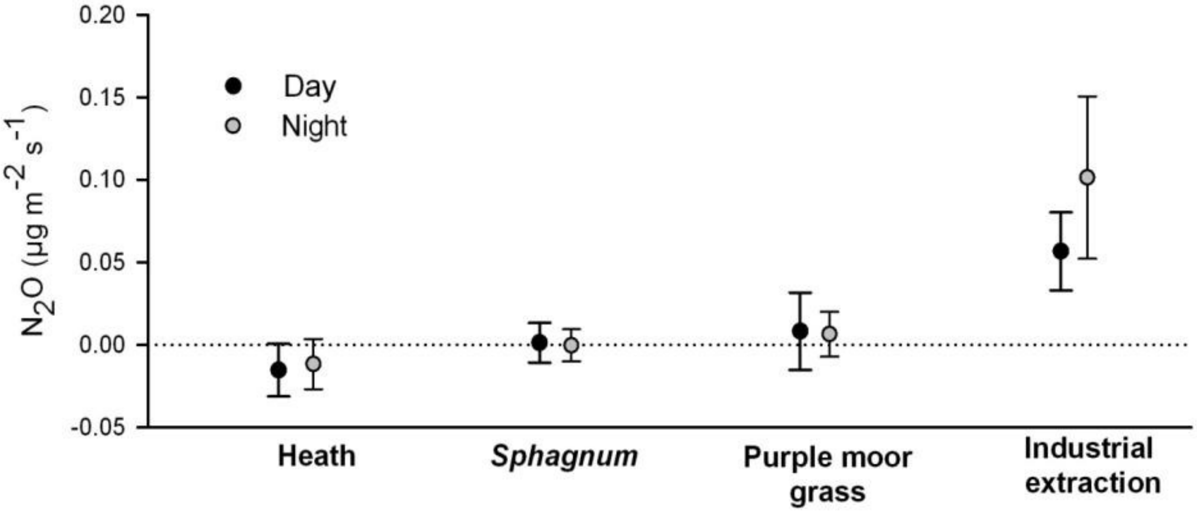


Figure 35: Comparison between day and night N_2O fluxes at four different subsites in the Himmelmoor measured at 30 to 31 August 2011. Values represent means and standard deviations of three replicates.

4.6 Greenhouse gas budget (GHG budget)

All CH_4 and N_2O fluxes were calculated into CO_2 equivalents and summed up with the CO_2 fluxes as a GHG budget (Table 15; for a better visualisation values are demonstrated in Figure 36). It was shown, that all subsites are significant sources of greenhouse gases, while CH_4 emissions have the quantitative greatest effect on the greenhouse gas budget at all vegetated subsites (77-99 % of the total GHG CO_2-eq emissions). The portion of N_2O is negligible at the vegetated subsites, while it plays a significant role at the industrial extraction site (1.54 ± 0.80 t CO_2-eq ha^{-1} $year^{-1}$). The uncertainty of the GHG budget is high due to high spatial variability between the replicate plots of each subsite and thus no significant differences between the subsites and due to the drought treatment was detectable (ANOVA). The highest emissions (39.69 ± 26.36 t CO_2-eq ha^{-1} $year^{-1}$) were detected at the purple moor grass site (P). Here, the emissions of the drought treatment were considerable lower, as the portion of CH_4 was stronger reduced by drought than CO_2 was increased (26.00 ± 22.16 t CO_2-eq ha^{-1} $year^{-1}$). Contrary, the emissions of the drought treatments at the heath (H) and the *Sphagnum* (S) subsite were higher than the control treatments. The lowest emissions are present at the industrial extraction site (8.90 ± 1.06 t CO_2-eq ha^{-1} $year^{-1}$). As the GHG budget of the industrial extraction site is not complete without taking into account the amount of peat that is extracted, it was added to the calculation. Including these C-losses, the emissions of the industrial extraction site are an order of magnitude higher than those of the vegetated sites (122.92 ± 6.78 t CO_2-eq ha^{-1} $year^{-1}$).

Table 15: Greenhouse gas budget (GHG) of four subsites in the Himmelmoor. GHG budget of the industrial extraction site is composed of measured fluxes on site (control) and the extracted peat in 2011 (peat mining), caluculated from the C-contrent of the amount of extracted peat per area. Values represent means of four (CO_2 and CH_4) or three (N_2O) replicates ± standard deviation.

Subsite	Treatment	CO_2	CH_4	N_2O	Sum
		(t CO_2.eq ha^{-1} $year^{-1}$)			
Heath	control	3.09 ± 3.86	15.89 ± 3.27	-0.07 ± 0.36	18.91 ± 5.07
	drought	-3.10 ± 1.16	25.00 ± 10.07	0.02 ± 1.04	21.93 ± 10.19
Sphagnum	control	0.60 ± 1.43	24.91 ± 17.38	0.15 ± 0.20	25.66 ± 17.44
	drought	0.09 ± 8.92	27.72 ± 14.35	-0.25 ± 0.47	27.56 ± 16.90
Purple moor grass	control	2.47 ± 3.30	37.1 ± 26.15	0.12 ± 0.36	39.69 ± 26.36
	drought	5.87 ± 13.16	20.01 ± 17.82	0.12 ± 0.60	26.00 ± 22.16
Industrial extraction	control	7.30 ± 0.67	0.05 ± 0.16	1.54 ± 0.80	8.90 ± 1.06
	peat mining	114.02 ± 6.70			114.02 ± 6.70
	total				122.92 ± 6.78

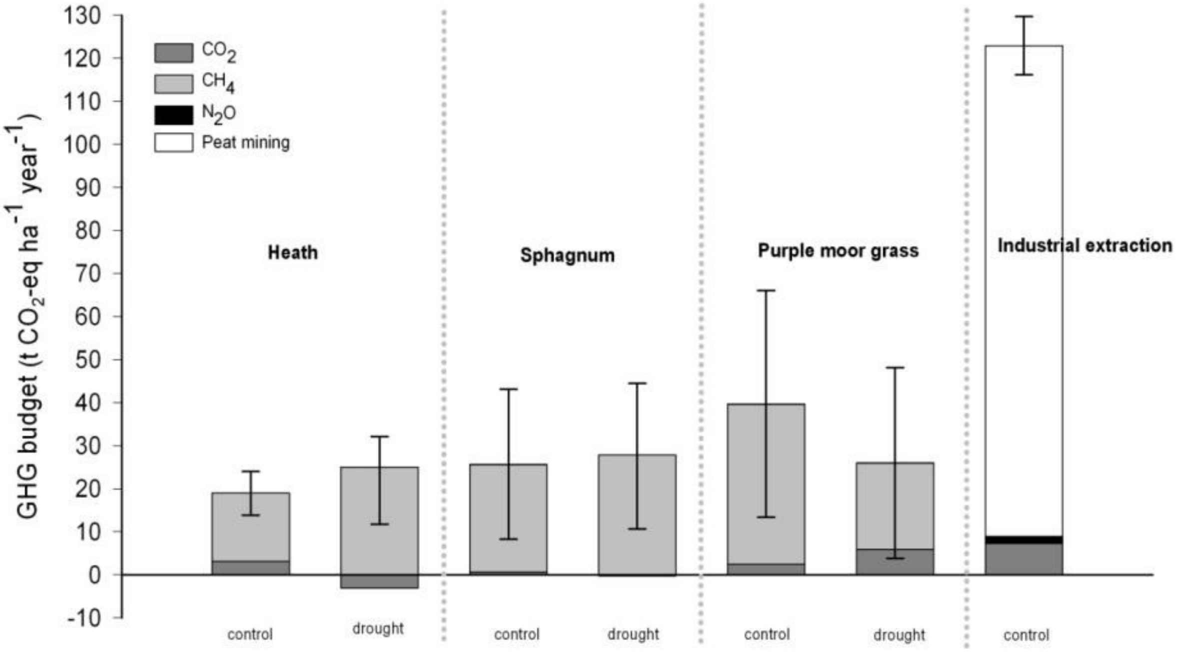


Figure 36: Greenhouse gas budget in CO_2 equivalents of four subsites in the Himmelmoor with and without drought treatment of 2011. Bars represent means of four (CO_2 and CH_4) or three (N_2O) plots per subsite ± standard deviation.

4.7 Characterisation of N-availability and net-mineralisation

4.7.1 Effect of summer drought on N-availability in the peat pore water

Total dissolved nitrogen (TDN) and dissolved organic nitrogen (DON)

The TDN in the soil pore water consists mainly (approx. 90%) of dissolved organic nitrogen (DON) and only little of dissolved inorganic nitrogen (DIN, Figure 37). TDN strongly varied over the year and spatially over the subsites, demonstrated by the high standard deviation of the mean of the four plots per subsite (Figure 38). Additionally interannual differences were found between the measurements in 2010 and in 2011.

In **2010** TDN, DON and DIN in the soil pore water differed significantly between the subsites (two-way ANOVA, $p = 0{,}01$, $p < 0{,}01$ and $p < 0{,}001$) with the highest values at the purple moor grass site and the lowest at the *Sphagnum* site (Figure 37). Drought highly significantly increased the concentrations of TDN, DON and DIN with $p < 0{,}001$, $p < 0{,}01$, $p < 0{,}001$ respectively. TDN increased most at the *Sphagnum* site, while the heath subsite showed the greatest increase in DIN due to drought. Additionally a significant interaction ($p<0{,}05$) was found between treatment (drought/control) and subsite implying that drought is unequally pronounced depending on the kind of vegetation type (subsite). The effect of drought was long-lasting, as it was detectable several weeks after the end of the drought treatment (Figure 38).

In **2011** no difference in TDN and DON was found between the subsites, while the heath subsite showed significantly higher DIN values ($p = 0{,}001$) than the other two subsites. No significant impact of drought on the dissolved nitrogen components in the soil pore water was found for the whole year as well as for the sheltered period only (Figure 37). However, it was shown that drought significantly increased the TDN concentrations at the purple moor grass site at several months in 2011 (Figure 38).

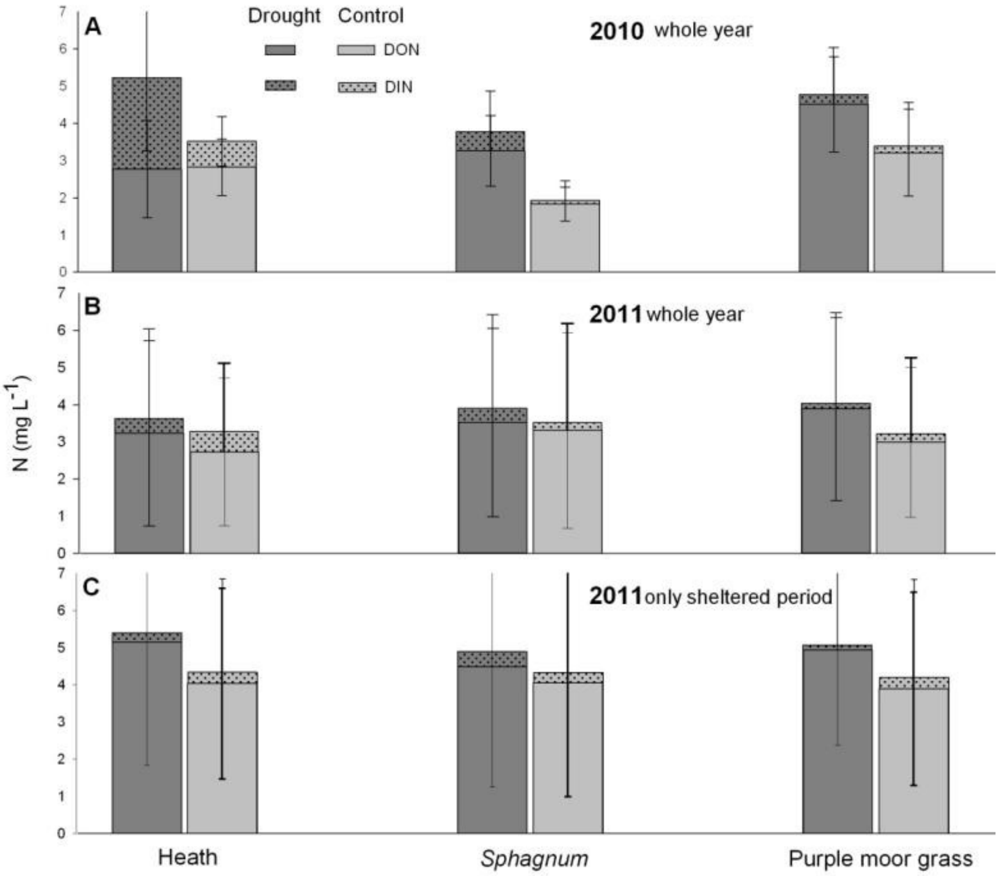


Figure 37: Effect of drought on dissolved organic and inorganic nitrogen (DON/DIN) at three different subsites in the Himmelmoor. The top edge of the bars represents the concentration of total dissolved nitrogen (TDN). Bars refer to means and standard deviations of 5 measurements between October and December in 2010 (A) and from12 measurements between January and December in 2011 (B). Panel C shows values of 5 measurements during the drought treatment period in 2011.

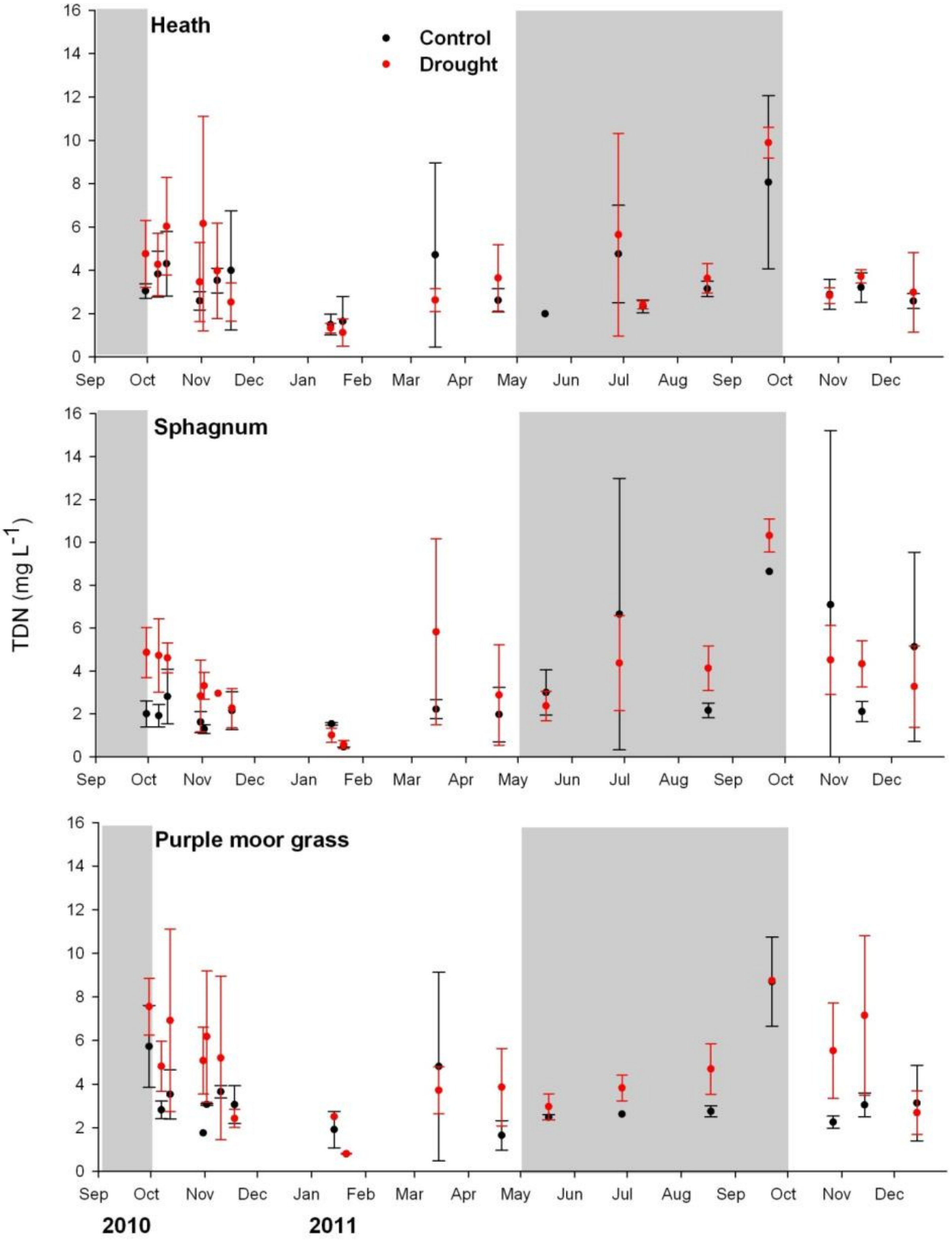


Figure 38: Effect of drought on total dissolved nitrogen (TDN) in the soil pore water from September 2010 until December 2011. Values represent means and standard deviations from up to four replicates per subsite and treatment. Grey background indicates periods of rainfall exclusion.

Dissolved inorganic nitrogen (DIN)

As DIN significantly differed between the subsites and was affected by drought, the three compounds of DIN (ammonium, nitrate and nitrite) are demonstrated separately from each other in the following.

Ammonium

For the year 2010 it was shown that the ammonium-N concentration in the soil pore water highly significantly depend on the type of subsite/vegetation ($p < 0,001$, Table 16) and at which time of the year it was measured ($p = 0.002$). The highest ammonium values were found at the heath subsite (0.66 ± 0.53 mg L^{-1}) followed by the purple moor grass site (0.14 ± 0.13 mg L^{-1}) and the lowest values at the *Sphagnum* site (0.1 ± 0.15 mg L-^{1}) (Figure 39). The high standard deviations indicate high special variability between the plots of a subsite.

The induced summer drought caused significantly higher ammonium-N concentrations in the pore water ($p = 0.002$) and the effect of the drought is unequally pronounced depending on the vegetation type ($p = 0.014$) and at which date was measured ($p = 0.011$). The highest increase in ammonium-N concentrations due to drought was found at the heath and the *Sphagnum* site, while the purple moor grass site was only little affected (Figure 39).

In 2011 the ammonium-N concentration differed between the vegetation types (subsites) when all measurements of the year were included in the calculation ($p < 0,001$; Table 16), but not if only the drought period is considered. In contrast to the year 2010, no influence of the induced drought on the ammonium-N content in the soil pore water was found in 2011.

Nitrate and nitrite

The nitrate-N concentration of the soil pore water was considerably lower than the ammonium-N concentration (Figure 39 and Figure 40) and no significant contents of nitrite were detected. In contrast to ammonium-N, no effect of drought on the nitrate-N concentration in the soil pore water was found in both years and there was no significant difference between the vegetation types (Table 16). The main factor for explaining nitrate concentration variability is the date of measurement (2010: $p < 001$, 2011: $p < 0,01$).

Table 16: Probability values for the effect of subsite, treatment, date and their interactions on the ammonium-N and nitrate-N concentration in the soil pore water for the drought treatment period (Three-way ANOVA). Values in bold indicate significant p-values (< 0.05). Whole year: Calculations include drought and rain periods.

ANOVA p-values for **ammonium-N**

	2010		**2011**	
Source of variation	drought	whole year	drought	whole year
Subsite (vegetation type)	**< 0,001**	**< 0.001**	0.343	**0.001**
Treatment (control/drought)	**0.002**	**< 0.001**	0.164	0.061
Date	**0.002**	0.061	0.471	0.057
Subsite x Treatment	**0.014**	**0.004**	0.600	0.348
Subsite x Date	**0.002**	0.108	**< 0.001**	**< 0.001**
Treatment x Date	**0.011**	0.337	0.450	0.725
Subsite x Treatment x Date	**0.030**	0.482	0.348	0.804

ANOVA P-values for **nitrate-N**

	2010		**2011**	
Source of variation	drought	whole year	drought	whole year
Subsite (vegetation type)	0.782	0.818	0.918	0.172
Treatment (control/drought)	0.258	0.148	0.143	0.172
Date	**< 0,001**	**< 0.001**	**0.008**	**< 0.001**
Subsite x Treatment	0.161	**0.026**	0.718	0.687
Subsite x Date	0.072	**0.047**	0.170	**0.022**
Treatment x Date	0.107	0.089	0.194	0.243
Subsite x Treatment x Date	0.893	0.862	0.962	0.998

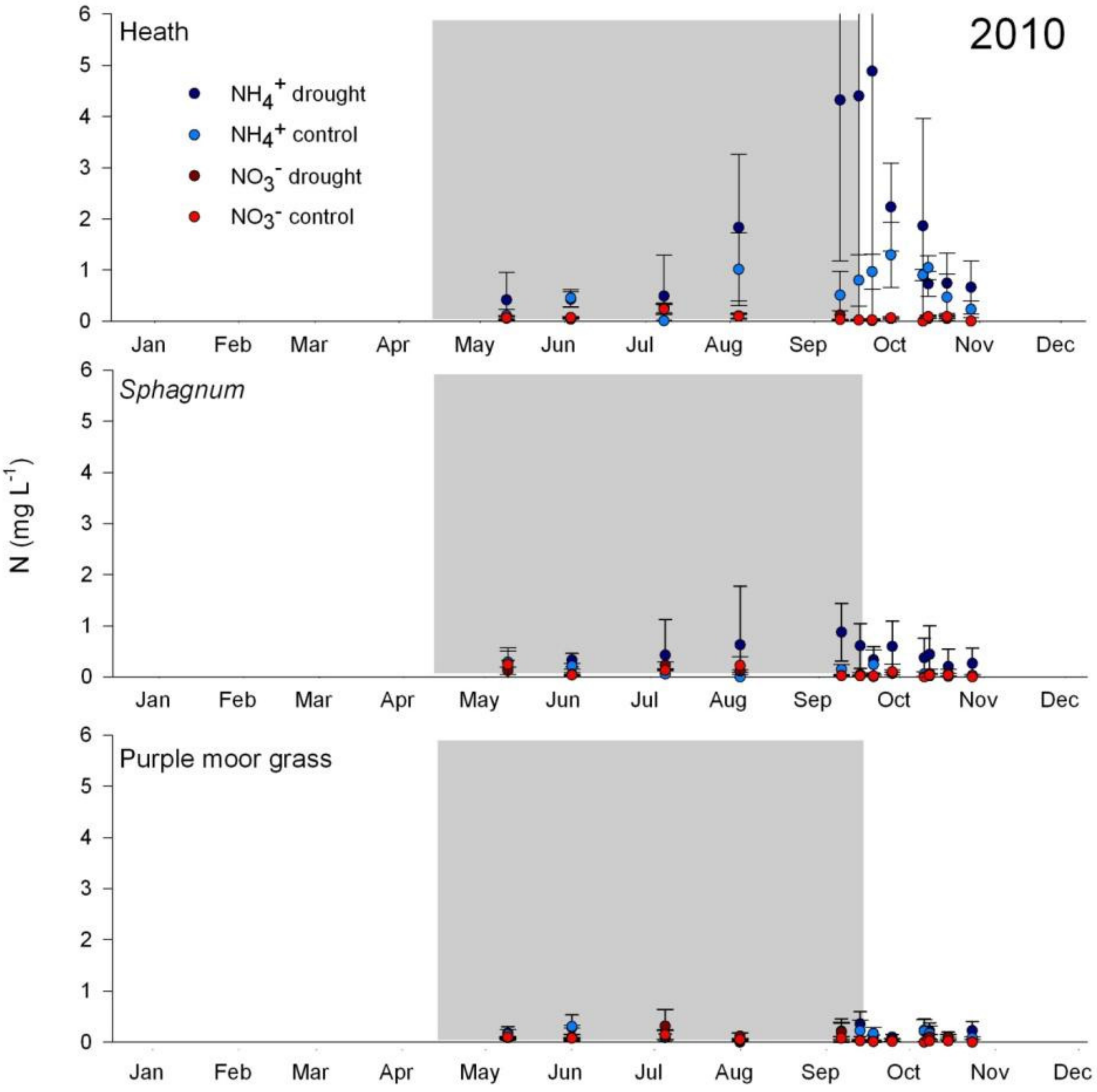


Figure 39: Effect of drought on ammonium-N and nitrate-N in the soil pore water in 2010. Values represent means and standard deviations of four replicates per subsite and treatment. No data available before May and after November. Grey background indicates period of rainfall exclusion.

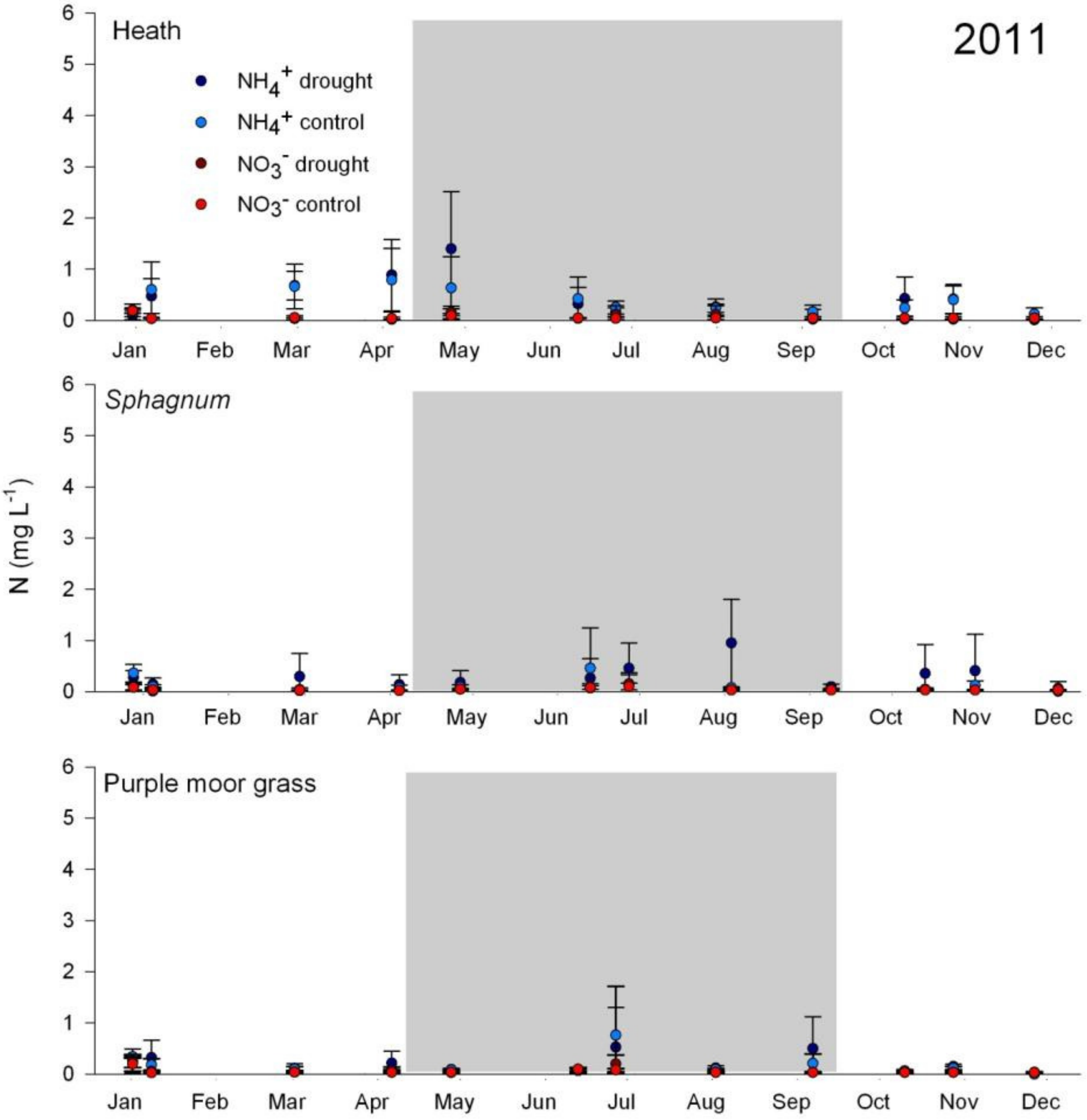


Figure 40: Effect of drought on ammonium-N and nitrate-N in the soil pore water in 2011. Values represent means and standard deviations of four replicates per subsite and treatment. Grey background indicates period of rainfall exclusion.

4.7.2 Net N-mineralisation of peat soil with differing moisture contents in an incubation experiment

Initial conditions of the soil samples for incubation

The soil samples for the incubation experiment, which were taken once per season in 2011, featured differences in initial concentrations of dissolved inorganic nitrogen (DIN), in the percentage of nitrate in the DIN and in water content.

The initial concentrations of DIN in the incubation experiment differed highly significant between the subsites (Figure 41, two-way ANOVA, $p = 0.001$). Highest values were detected at the heath subsite followed by purple moor grass and the industrial extraction site, while the contents at the *Sphagnum* subsite were negligible low. DIN showed seasonal variations with a highly significant increase along the growing season ($p < 0.001$). The seasonality is unequally pronounced depending on the vegetation type/subsite ($p < 0.01$).

The composition of DIN differs highly significant between the subsites (Figure 41, two-way ANOVA, $p < 0.001$): while DIN at the vegetated subsites mainly consists of ammonium, the percentage of nitrate at the industrial extraction site ranged between 40 and 90 %.

The initial water content of the base peat material differed highly significant between the vegetation type/subsite (Figure 41, two-way ANOVA, $p < 0.001$). A significant interaction between the season and the subsite was found, indicating that the effect of the season on the water content varies depending on subsite ($p < 0.01$): *Sphagnum* water content increased during the season, while it decreased at the industrial extraction site.

After reducing the amount of water in the peat by 15 % and 30 %, the water content of the soil material for the incubation ranged between 79-95 % and 76-94 % respectively.

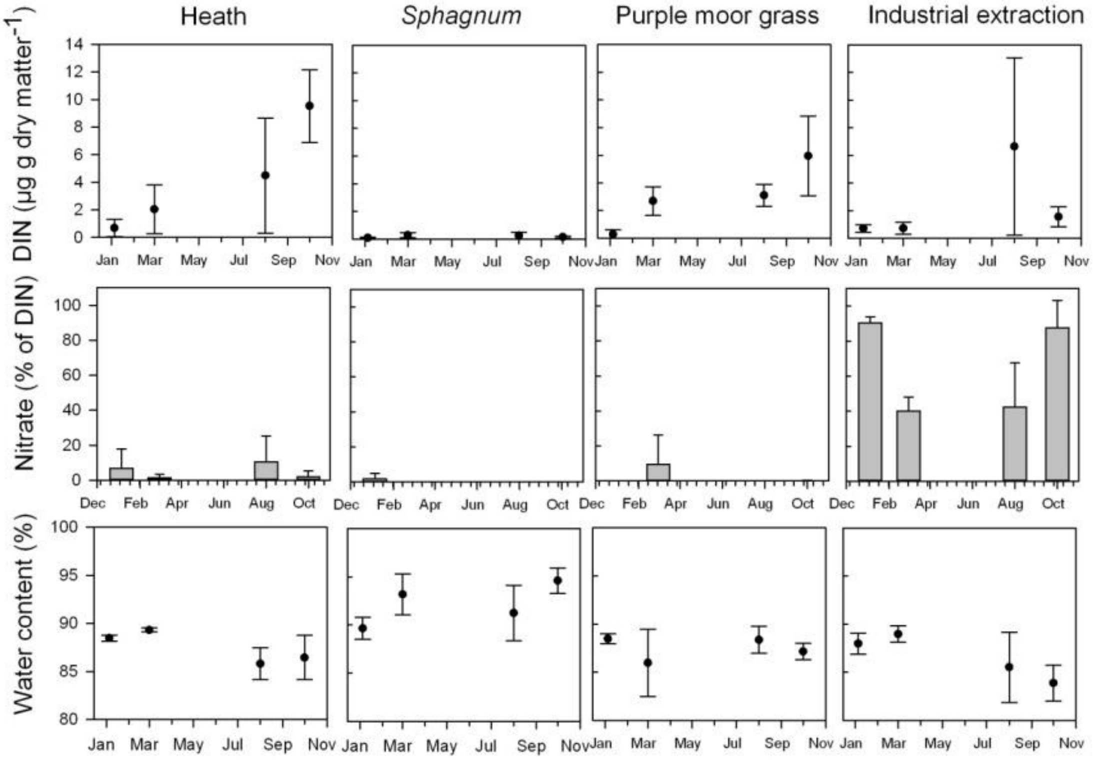


Figure 41: Upper panel: Seasonal variation in dissolved inorganic nitrogen (DIN) in the soil in 2011. Middle panel: Percentage of nitrate on dissolved inorganic nitrogen (DIN). Lower panel: Initial water content. Values refer to means of three soil samples per subsite ± standard deviations.

Net nitrogen mineralisation

Soils of all four subsites were incubated once a season in 2011 for the determination of net nitrogen mineralisation rates. The net mineralisation rate is positive in most cases. Highly significant differences were found in mineralisation between the vegetation types/subsites (Figure 42, three-way ANOVA, $p = 0.002$) with greatest differences between the heath and the purple moor grass subsite. However, the order of the subsites from highest to lowest mineralisation rates varied every season. Thus, temporal variability was high as well as spatial variability which can be seen by the high standard deviation of the mineralisation rates.

Additionally, a significant influence of the season on the mineralisation rates was found ($p < 0.05$) and there is a highly significant interaction between the subsite and the season ($p < 0.001$), indicating that the differences between the subsites depend on the season. The drought treatment had an ambiguous effect on the mineralisation rates. For example, the incubation of soil from the heath subsite in October showed a distinct decrease in mineralisation rates resulting in a negative net mineralisation (net immobilisation), while the rates of the *Sphagnum* sample of January increased. However, most of the mineralisation rates were not affected by the water content treatments, thus no significant influence is evident.

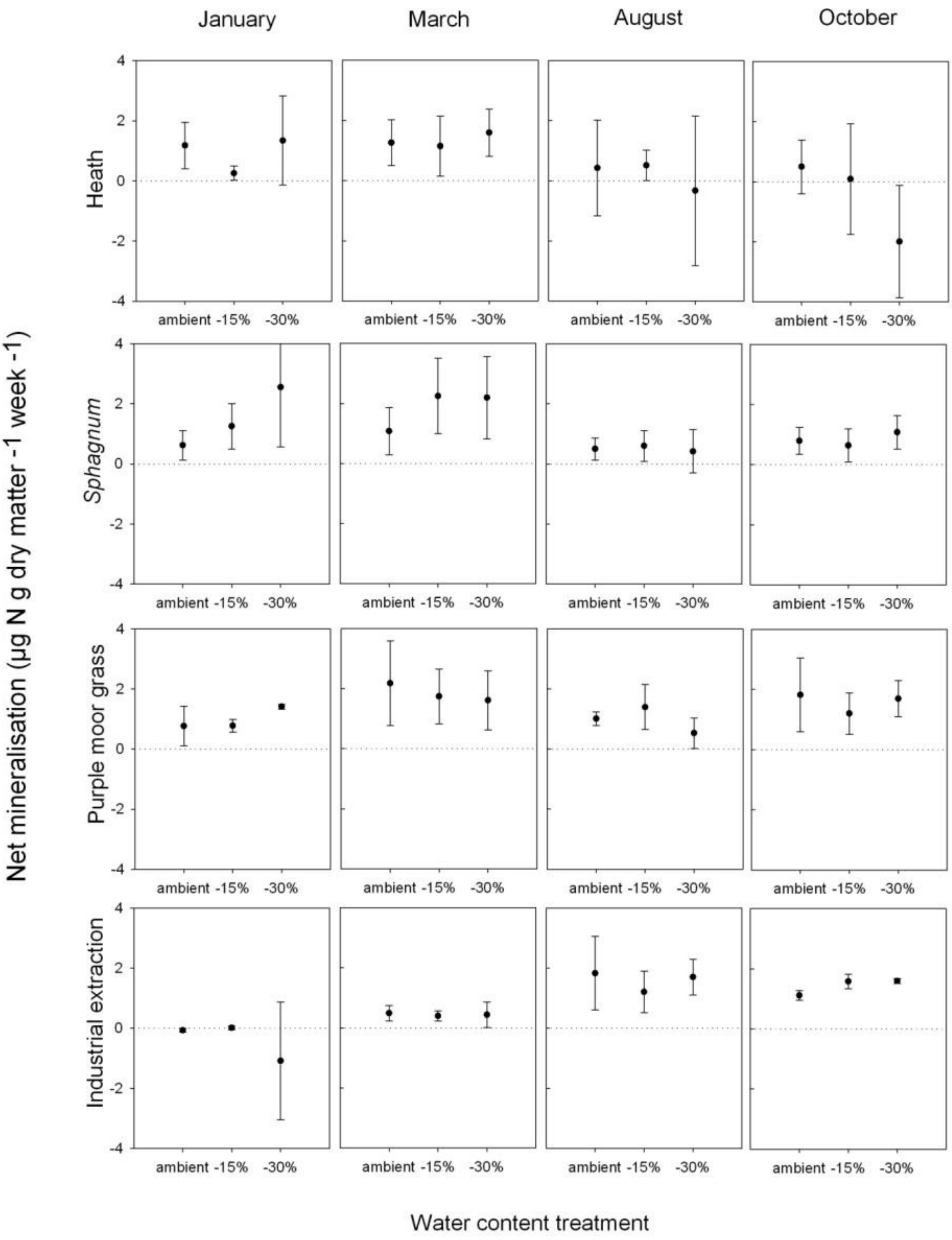


Figure 42: Seasonal variations (2011) in mineralisation rates at the four different subsites with differing moisture contents in an incubation experiment. Values refer to means of three soil samples per subsite ± standard deviations. Negative values imply a net N-immobilisation. Ambient water content can be seen in Fig.XY.

5. Discussion

5.1 Critical review of used methods

To estimate greenhouse gas fluxes at four different subsites in the Himmelmoor, the chamber technique was applied. Additionally, a summer drought was simulated by the use of rainout shelters to estimate the impact of future climate change. Quality and success of these approaches are discussed in the following.

5.1.1 Chamber Measurement

The use of closed chambers is the most common method to measure surface-atmosphere gas exchange. However, it is facing some well documented problems. Most of them derive from chamber construction: small chambers exhibit a large degree of flux underestimation (Pihlatie et al., 2013) and the use of non-vented and non-mixed chambers also leads to underestimations of fluxes (Christiansen et al., 2011). Another reason for a biased flux determination is the application of linear flux calculation (Kutzbach et al., 2007), while the use of non-linear flux calculation significantly improves the flux estimation (Pihlatie et al., 2013). According to these findings, the chambers used in this project have been equipped with a pressure vent and a fan and are big enough in size as recommended by Philatie et al. (2013). Thus, underestimation of fluxes was reduced to a minimum by chamber design and non-linear flux calculation so that the advantages of this method predominate, e.g. in allowing the assessment of a great spatial variability and different treatments.

5.1.2 Simulating climate change

The use of 3 by 3 m rainout shelters was a requirement of the KLIMZUG-Nord project. As the regional climate model REMO (Umweltbundesamt, 2006) predicts for northern Germany a maximum reduction of summer precipitation by 30 % the requirement was to reduce summer precipitation by 25 %. But, as this shelter construction was planned for ecosystems with well drained soils it was feared that this amount of rainfall reduction on a relatively small area will not affect the hydrological regime of a waterlogged ecosystem and hence will result in non-detectable effects. Thus, the simulated rainfall exclusion of 100 % done in this study was a compromise solution between unsing the shelters and inducing a drought in a peatland. Therefor the possibility of inducing a drought in water logged soils with these small rainout shelters was maximised by this 100 % rainfall exclusion. As this simulation will obviously not happen due to climate change the quantitative results of the drought treatment must be interpreted with a certain degree of caution. However the simulation provides valuable qualitative effects.

The evaluation of the shelter effect demonstrated, that the rainout treatment did not affect the water table. However, as significant amounts of rain were retained from vegetation and soil surface, effects on vegetation, GHG fluxes and N-availability were detectable in drought treatment plots and hence, the development of a drought situation is proved. It can be expected that the rainout shelter treatment had measurable effects on soil moisture and oxygen supply in the uppermost centimetre of the soil, which, unfortunately was not measured. Additionally it was stated by Robroek et al. (2009), that precipitation is at least equally important as water table drawdown and a lack of precipitation negatively affects CO_2 uptake by Sphagnum species. They showed further, that precipitation can compensate for negative effects in times when water level is reduced. As water table fluctuation at the measurement site were high and low water levels up to -20 cm under soil suface appeared during summer at the vegetated sites, frequent precipitation is especially important. Hence, the lack of precipitation in summer month is likely to be meaningful for the peatland vegetation irrespective of wether drought treatment influenced water table or not.

As climate change does not only affect changes in precipitation, it can be questioned why the predicted temperature increase is not simulated in this thesis. Actually, it is partly indirectly included in the drought treatment as soils will become drier under a warmer climate unless increased evapotranspiration will not be balanced by increased precipitation (Macrae et al., 2013). Since increased precipitation is not predicted for northern Germany by the regional climate model REMO a rainout treatment can additionally be seen as a proxy for drying due to increased temperatures as describe by Macrae et al. (2013).

5.1.3 Measurement artefacts of the rain exclusion experiment

Besides inducing a summer drought, the rainout-shelters had some undesirable side effects. Measuring the photosynthetically active radiation (PAR) under a shelter and comparing it with the ambient values showed an unexpected high relatively reduction especially in the summer months, when the sun reaches its highest point and the incident radiation was highest. This shading explains the cooling effect of the shelter on the air temperature at high temperature ranges and was reported by Yahdjian and Sala (2002) as well. Furthermore, as the sheets have been used in 2011 for the second year, it cannot be ruled out that their transmittance was affected by surficial weathering. Unfortunately, there is no PAR data available for 2010 when the material was pristine. Additionally, transmittance of the sheets can be reduced by water condensation, raindrops or dust from the nearby peat extraction site. The latter could explain the increase in transmittance after august 2011 as the high amounts of rainfall might have cleaned the surfaces.

Much more astonishing is that the shelter racks alone, without greenhouse sheets, account for a 10 % reduction of PAR on average, although it is a minimalist skeleton, which gives cause for questioning the reliability of the measured PAR values. As the PAR sensor under the shelters is close to the vegetation canopy it was observed that fauna (e.g. spiders or lizards) disturbed the PAR measurement and snow accumulated on this sensor more than at the control sensor due to lower wind speed close to the soil surface. Due to differences in the placements of the PAR sensors, the sensor under the shelter might be more shaded by the birch stands of the elevation in the south of the measurement site, than the PAR sensor at the meteorological station wich is approximately in 2.5 m height. Additionally, as the PAR sensor under the shelter could not be placed with a levelling unit, the accuracy of placement was limited.

As treatment artefacts of manipulation experiments are unavoidable, their dimensions should be monitored and kept as small as possible and they should be included in calculations. For this reason the construction and the placement of the shelters and the choice of the greenhouse sheets were made very carefully. For example, the UV-A (78-85 %) and UV-B (70 - 80 %) transmittance of the used greenhouse sheets was even better then the material used by Yahdjian and Sala (2002) with only 70 % and 50 % transmittance, respectively.

The effect of altered temperatures and shading under the shelter on the GHG fluxes was directly measured and calculation of the fluxes was based on the actual temperature and PAR values under the shelter but modelled with the ambient ones. Thus the shelter effect is included in the calculations and upscaling was done with the ambient conditions. However, the desired shelter effect (drought) cannot be separated from the undesired effects (PAR and air temperature). It is thus not possible to exclude that reduced temperature (in high temperature ranges) and irradiance under the shelter partially contributes to the measurd differences that were found at the drought treatment plots. But it is unlikely that these reductions account for increases in processes where enzymes are needed or in microbial activity (like mineralisation or plant biomass increase), as these processes generally increase with increasing temperatures. It might have played a role, where productivity of plants or their photosynthesis was reduced. However, as the plants of the three different communities are ranging in the same habitat preferences concerning light abvailability (Ellenberg et al., 2001), shading under the shelter should have had the same effect at all three subsites. As productivity and photosynthesis of the purple moor grass site increased, while it decreased at the heath subsite, it is hence unlikely that the decrease was caused by the shelter effect. However, *Sphagnum cuspidatum* has the highest Ellenberg indicator value for light (9) and a high cover at the *Sphagnum* subsite. It is hence possible that the decrease in NEE and Reco was caused partially by the decrease in light availablility. The results of the *Sphagnum*

subsite must thus be treated with some caution, keeping in mind that they might have been influenced by treatment artefacts.
Despite these constrains, the value and need of experimental field manipulations e.g. in peatlands was underlined by other authors (Dise, 2009; Dise and Phoenix, 2011; Poll et al., 2013).

5.2 Subsite differentiation by plant communities

Within the restored measurement site three subsites were identified by visual differentiation between plant communities and were studied separately. This distinction proved to be an appropriate decision, as several anticipated differences between these three subsites were confirmed by measurements.

First of all the vegetation analyses of Sebastian R. Schmidt quantified the differences in species composition and coverage. The subsites are dominated by plants of differing life-forms emphasising diverse functions and adaptations in the ecosystem: evergreen, ericaceous shrubs (heath subsite), mosses of the Genus *Sphagnum* as well as sedges (*Sphagnum* subsite) and monospecific stands of the perennial deciduous grass *Molinia caerulea* (purple moor grass site). The invasion of *M. caerulea* in Dutch bogs was explained by high N-deposition levels rather than by desiccation due to low water tables (Tomassen et al., 2003; Tomassen et al., 2004a). However, in this study all subsites facing the same amounts of N-deposition as they are directly located next to each other. It is more likely that the differing vegetation communities are the result of different hydrology, as the subsites are divided by former drainage ditches and their blocking as restoration measure had a varying degree of success, resulting in different water table levels. The lowest water table position was detected at the **purple moor grass site,** and the interaction between the relatively drier surface and the appearance of the non-typical bog plant results in many statistical significant differences compared to the other two vegetated subsites, namely in soil characteristics (high pH value, bulk density and ash and water content), N-status (highest TDN and DON concentrations in 2010) and GHG-fluxes (highest CH_4 fluxes, highest gross primary production and highest maximum photosynthetic activity). It can be expected that the relatively lower water level causes a drier soil surface with increased decomposition resulting in increased bulk density, ash content and TDN (Boelter, 1968; Martikainen et al., 1993b; Holden et al., 2004; Kløve et al., 2010; Macrae et al., 2013). Less *Sphagnum* mosses mean less acidification of the soil and less unfavourable conditions for non-typical bog plants such as the purple moor grass (Clymo, 1964). Increased dominance of *M. caerulea* results in an increased flow of carbon and nutrients into the litter department as it has a high growth rate and 48 % of its biomass is located in the leaves (Heil and Bruggink, 1987; Aerts and Berendse, 1988; van Breemen, 1998).

The fast growth of the leaves and their autumn senescense can explain the annual dynamics of the modelled NEE and the measured GPP values at the purple moor grass site. As a deciduous grass, there is nearly no photosynthesis in winter time (except from green basal internodes as described by Jefferies (1915)) and photosynthesis steeply increased with the growth of the green leaves and abruptly stopped with the dieback of the grass leaves. Contrary photosynthesis was detected in winter time at the *Sphagnum* and the heath subsite as these sites were dominated by evergreen plants.

At the ***Sphagnum* subsite**, which has 100 % coverage of *Sphagnum* mosses, the characteristic and main peat forming plant in bogs, the soil conditions are like it is usually described for an ombrotrophic bog: high water content, low pH and dry bulk density values (Boelter, 1968; Sjörs and Gunnarsson, 2002). The lowest DIN, DON and TDN concentration were found at this subsite as Sphagnum has a high N-uptake rate in nutrient poor conditions limiting the N-availability for vascular plants (Tomassen et al., 2003) and thus gives *Sphagnum* a competitive advance in natural peatlands (Malmer et al., 1994).

The **heath subsite** has an intermediate water table level between the purple moor grass and the *Sphagnum* subsite. It is dominated by ericaceous shrubs which have as evergreens long-term competitive advantages over herbaceous species in low nutrient environments as they can retain moisture and nutrients for a long time (Bubier et al., 2003) and has a longer growing season (Heil and Bruggink, 1987). As a plant with low nutrient requirements it generally has a low growth rate (Aerts and Caluwe, 1989). It invests only 12 % of its aboveground biomass in leaves, has a low litter production and thus low nutrient losses (Aerts and Berendse, 1988; van Breemen, 1998). Additionally ericoid mycorrhizal symbionts enhance the ammonium absorption at low concentrations and provide its hosts with access to other resources such as amino acids, peptides and proteins as nitrogen substrates for growth (Read and Bajwa, 1985). As this mycorrhiza makes the ericaceous shrubs independent from available DIN by using organic N-sources, this can explain the higher DIN/DON ratio found at this site.

In contrast to *Molinia caerulea,* ericaceous shrubs have high lignin concentrations (*Erica tetralix*: 33 %, *M. caerulea* 24 %) and thus have low decomposability of the plant litter (van Breemen, 1998; DeLuca et al., 2002). This might explain that the greatest difference in net-mineralisation rate was found between the samples of the heath and the purple moor grass site.

5.3 CO_2 and GHG budget - assessing the restoration success

The GHG budget estimates of the industrial extraction site are dominated by CO_2 emissions with a significant contribution of N_2O emissions. The annual net ecosystem exchange of CO_2 of 7.3 t CO_2 ha^{-1} $year^{-1}$ is ranging in the dimension of 11 reported studies from European peat cut sites with a median of 2 and a maximum of 13 t CO_2 ha^{-1} $year^{-1}$ (Drösler et al., 2008). In the report of Drösler et al. (2008), only one study is included providing N_2O emissions from a peat cut site; however, it is similar to the emissions found in this study (approx. 1 t CO_2-eq ha^{-1} $year^{-1}$). The importance of a future rewetting of the industrial extraction site after abandonment is stressed by the high CO_2 emissions (236 t CO_2 ha^{-1} $year^{-1}$) at a bare peat site in a temperate European bog which was abandoned in 1994 and no rewetting measures took place (Couwenberg, 2011). In comparison to that, the actual CO_2 emission from the peat cut site is relatively low, as the peat material at the soil surface starting to mineralise is several times per year removed and microbial decomposition has to start again with older, even more recalcitrant peat.

Annual net ecosystem exchange of CO_2 of the vegetated subsites are lower than from the peat cut site, while none of them act as CO_2 sink as expected. The *Sphagnum* subsite has the lowest CO_2 emissions ranging with its uncertainty associated to the spatial variability close to a CO_2 neutral status. The magnitude of the CO_2 budget is consistent with the analyses of 53 publications evaluated by Couwenberg (2008), where temperate European peatlands with similar water table positions emit less than 3 t CO_2 ha^{-1} $year^{-1}$ (utilisation and restoration status not mentioned). Although the CO_2 budget of the restored side can thus be considered a positive in respect to the restoration success, this valuation does not apply for the complete GHG budget due to high CH_4 emissions. In contrast to the industrial extraction site, the GHG budget of the restored site is dominated by CH_4 fluxes (up to 99 %). This is in agreement with other studies reported from bogs in comparable regions: Couwenberg et al. (2008) analysed annual CH_4 emissions of peatlands in the temperate region of Europe from 130 sites. Their report shows for bogs with a water level close to the surface that the positive global warming potential derives only from CH_4 emissions. Contrastingly the CH_4 emissions and thus the GHG budget of the present study are much higher than documented in the literature. The maximum rate reported for a European bog is approx. 11 t CO_2-eq ha^{-1} $year^{-1}$ while use and restoration status were not mentioned (Couwenberg et al., 2008). In the Himmelmoor, mean annual CH_4 fluxes between 15 and 37 t CO_2-eq ha^{-1} $year^{-1}$ were found.

However, similar fluxes were reported from Crill et al. (1988) in a temperate, undrained open bog in Minnesota (US) with a mean of 26.8 ± 2.7 t CO_2-eq ha^{-1} $year^{-1}$. Couwenberg et al. (2008) found that highest CH_4 fluxes in bogs appeared when peat had a C/N ratio around 40 and water level was close to the soil surface. These conditions apply for the vegetated subsites in this study and can thus partially provide an explanation for the high annual CH_4

emissions. Additionally, large fluctuations in water level at the measurement site, as typical for degraded peatlands (Schouwenaars, 1993; Tuittila et al., 1999), might explain high CH_4 emissions as young below and above ground plant litter is thereby from time to time inundated and undergoes anoxic fermentation (Augustin and Joosten, 2007; Couwenberg et al., 2008; Paul and Alewell, 2013). This applies especially for the purple moor grass site. Here, CH_4 fluxes were higher than at the *Sphagnum* and the heath subsite (2.3 and 1.5 times, respectively) as *M. caerulea* produces large amounts of easily decomposable litter in comparison to the other plants present at the study site. A similar effect was reported for stands of reed canary grass (*Phalaris arundinacea*) in a German fen causing extremely high CH_4 emissions (Augustin and Joosten, 2007; Hahn-Schöfl et al., 2011). Furthermore, it is stated in the literature that *M. caerulea* has aerenchyma (Jaiswal et al., 2000) and thus can act as gas conduit. It hence allows CH_4 to bypass the oxic acrotelm, which reduces CH_4 oxidation. Although the plant mediated transport of CH_4 through *M. Caerulea* is lower compared to *Eriophorum angustifolium* (Bhullar et al., 2013) this effect is increased by the high cover of this grass species at the purple moor grass site. Thus, it was shown, that CH_4 emissions increase with increasing coverage of shunt species at the subsites in the following order: heath < *Sphagnum* < purple moor grass with a mean coverage of 23, 67 and 71 % respectively.

The particular soil conditions of this degenerated peat site, in combination with water tables fluctuations and a high cover of vascular plants with aerenchyma seem to form specific conditions for both, high methanogenesis and efficient soil atmosphere CH_4 transport. This results in extremely high CH_4 emissions in comparison to other bog ecosystems in temperate Europe and in high GHG budgets for all vegetated subsites. With respect to climate change mitigation, the restoration of the study site so far was not successful as the annual GHG emissions are at least twice as high as the rates of the industrial extraction site (without extracted peat) and the CO_2 sink function was not re-established yet. As this restored site was strongly affected by drainage, peat cutting and mixture of peat layers, it might need a longer time to establish near to natural conditions in e.g. water table and CO_2 sink function. Additionally, a comprehensive restoration success must be valued by other ecosystem functions as well and by the establishment of typical bog species communities.

5.4 Vulnerability of the restored peat sites for summer drought

5.4.1 Increase in nutrient availability

Although no clear increase in net-mineralisation rates in the incubation experiment was found, drought treatment was expected to increase net mineralisation rates as the results of the soil pore water analysis in 2010 showed significantly higher TDN, DON and DIN concentrations in all vegetation communities. However it must be mentioned, that an increase in available nitrogen could also be explained by reduced uptake, for example when plants under the shelter were negatively affected by the lack of precipitation or by the reduced irradiance under the shelters. Interestingly the increase of DIN was very low at the purple moor grass site and very high at the heath subsite. For the interpretation of the measured values, it must be noted that they do not display the gross mineralisation due to simultaneous immobilisation by plants and microorganisms. High immobilisation is expected with soil C/N ratios greater 25 (Westbrook and Devito, 2004; Bengtson, 2006). It can thus be expected that the mineralisation at the purple moor grass site was higher, but DIN was directly incorporated into biomass. It was shown in other studies that *M. caerulea* profits more from an increase in nutrient availability than e.g. *Calluna vulgaris* and *Erica tetralix* (Heil and Bruggink, 1987; Aerts and Berendse, 1988; Tomassen et al., 2003). This high nutrient uptake efficiency is reflected by the significant increase of *M. caerulea* plant biomass due to drought treatment within only one year, stressing that this grass is a better competitor for nutrients and can replace other bog plants in response to a drier soil surface (Aerts and Caluwe, 1989; van Breemen, 1998; Tomassen et al., 2003; Tomassen et al., 2004a).

5.4.2 Changes in greenhouse gas fluxes

CH_4 fluxes were hypothesised to decrease due to drought, as oxidation will increase in the drier soil surface (Freeman et al., 1993; Moore and Dalva, 1993; Hughes et al., 1998; Couwenberg, 2009a; Bridgham et al., 2013). However, no significant effect of drought on CH_4 fluxes was detectable although CH_4 emissions were considerably reduced at the purple moor grass site. As this site has the lowest water table level from all vegetated subsites, the rainfall exclosure seemed to reach enough aeration for a detectable increase in methane oxidation. The decrease in CH_4 emissions due to drought accordingly decreased the GHG budget of the purple moor grass site, reaching the level of the *Sphagnum* subsite. On the first sight, this may appear as a positive effect of climate change. It has to be considered however, that this short-term benefit might be outweighted by the intensified mineralisation in the following years due to an intensified deep soil mineralisation and an increased flux of carbon and nutrients into the easy decomposable plant litter of *M. Caerulea,* causing

increased CO_2 emissions after senescense (Aerts and Berendse, 1988; Aerts and Caluwe, 1989; van Breemen, 1998; Gogo et al., 2011).
Additionally it is likely that *M. caerulea* increases its coverage on the *Sphagnum* and the heath subsite due to climate change and will thus deteriorate the GHG budget of the newly colonised areas.

The undetectable effect of rain exclusion on CH_4 fluxes of the *Sphagnum* and heath subsite indicates that these sites remain too wet for increased CH_4 oxidation as they had higher water levels than the purple moor grass site. It is thereby unclear, if an actual summer drought due to climate change would not affect CH_4 fluxes as well, or if these results reflect the limits of this rainout shelter method (Chapter 5.1.3), as water table was not reduced by the drought treatment. The same applies in respect to N_2O fluxes which did not show the expected increase due to drought treatment. However significant N_2O fluxes due to drought proved to be unlikely by the results of the soil properties and nutrient status: With a C/N ranging between 30 and 59 and low percentage of nitrate in the DIN, only negligible amounts of N_2O can be emitted (Martikainen et al., 1993b; Klemedtsson et al., 2005; Couwenberg et al., 2008; Drösler et al., 2008). Although ammonium significantly increased in the rain exclusion treatment, likely due to increased mineralisation, no increase in nitrate and thus in N_2O emissions was detectable. It can be hypothesised that ammonium is directly taken up by plants when it is oxidised to nitrate by nitrification, as peat vegetation controls the availability of nitrate for N_2O producing microorganisms (Silvan et al., 2005). This is supported by the fact, that DIN is mainly consistent of nitrate in the absence of vegetation (industrial extraction site) as there is no uptake by plants.

For CO_2 emissions, opposing effects of summer drought on plant communities were detected. The plants at the *Sphagnum* subsite suffered from drought by reducing productivity, which can be seen by reduced Reco and GPP resulting in no differences in NEE. At the heath subsite GPP increased while Reco decreased and hence NEE became negative, turning from a CO_2 source to a sink. Contrastingly, Reco and GPP increased at the purple moor grass site indicating an increase in mineralisation, plant respiration and productivity. The latter is reflected by the significant biomass increase within only one year. These results imply that *M. caerulea* profits from drought by increased nutrient availability as discussed above (Chapter 5.4.1), which is concordant with the results of other studies (Heil and Bruggink, 1987; Aerts and Caluwe, 1989; Aerts and Ludwig, 1997; Tomassen et al., 2004a).

5.5 Implications for restoration and climate change

Industrial extraction site

Restoration status and greenhouse gas budget of the vegetated subsites (H, P and S) does not directly allow a prediction for restoration of the industrial peat mining site (A). While the H, P and S subsites developed on a block-cut zone (Chapter 3.2), the progress of restoring the milled peat area will be more slowly due to the absence of viable seeds and unfavourable conditions for plant growth (Tuittila et al., 1999; Triisberg et al., 2011). As the vegetative spread of plant species, predominating in bog vegetation is not possible on the cut-away area (Quinty and Rochefort, 2003; Triisberg et al., 2011) these areas will remain plant-free for a relatively long time (Tuittila et al., 1999). This can be seen at those parts of the industrial extraction site in the Himmelmoor, which were ceased since 2008 and which remain mostly unvegetated until now. To enhance restoration progress, special techniques are required such as active introduction of plant species and levelling of sloping fields (Quinty and Rochefort, 2003). Additionally, it must be carefully monitored, if *M. caerulea* tends to dominate on the industrial extraction site after abandonment as it has some advantages over typical bog vegetation in colonizing this unvegetated soil surface. In contrast to the vegetative spread prevailing in bog plants, the seeds of the purple moor grass are dispersed by wind, and cracks in the peat surface provide ideal conditions for germination. The developing seedlings are capable to grow on highly exposed and inhospitable surfaces (Jefferies, 1915) and with its large root system *M. caerulea* can exploit a large volume of soil to gain nutrients (Taylor et al., 2001). With regard to climate change and increased frequencies and duration of drought periods, a dominance of *M. caerulea* must be expected as it was shown that it can significantly increase its biomass within only one year due to drought. This is concordant with other studies showing that *M. caerulea* can outcompete other bog plants by increased nutrient supply (van Breemen, 1998; Tomassen et al., 2004a).

Vegetated subsites

Dise and Phenix (2011) suggest that warmer temperatures combined with summer drought will alter species composition by accelerated decomposition toward non-peat forming graminoid and shrub communities. This is concordant with the results of Jassey (2013) showing that warming reduces *Sphagnum* cover and leads to an increase in vascular plants cover. It was shown in this study that *Sphagnum* is threatened by reduced precipitation as it decreased its productivity within one year due to drought treatment which is supported by Robroek et al. (2009) who showed in their study that a lack of precipitation negatively affected CO_2 uptake by *Sphagnum* and that recovery after drought was low. Additionally, a decrease of productivity at the heath subsite was found, indicating that plants of this community suffer from rain exclusion, too. The opposite effect was found within the stands of

M. caerulea which benefited from drier conditions as it increased its productivity and significantly gained more biomass from 2010 to 2011 due to drought treatment. It is unlikely, that this increase derives from the undesired shelter artefacts as this grass is used to high light conditions and would more likely impaire from reduced irradiance and temperatures (see chapter 3.5.1). It is thus from particular concern that this invasive grass species becomes more widespread in the future due to its competitive advance over typical mire plants if soil moisture is reduced. *Sphagnum* species are therefore directly threatened by drought and increased temperatures and indirectly by the spread of vascular plants (in particular *M .caerulea)* which in consequence seriously compromises the capacity to sequester carbon (Malmer et al., 1994; Gogo et al., 2011; Limpens et al., 2011). Management measures stimulating growth of *Sphagnum* are therefore required to reduce negative effects of climate change such as optimizing water table level, reducing water level fluctuations and removal of shading plants like invasive birches (Tomassen et al., 2004a). These measures are also recommended to reduce the present GHG emissions by decreasing the coverage of *M. caerulea*. This will provide less easy decomposable plant litter for methanogenesis and less shunts for CH_4 emissions. Reduced water level fluctuation will in consequence reduce inundation of fresh plant litter resulting in high CH_4 emissions. An orientation for water table management should be the mean water level of -10 cm under soil surface. This critical value is reported to increase methane oxidation and hence significantly reduce CH_4 emissions (Drösler et al., 2008). As a lower water level can deteriorate the growth of *Sphagnum* species, enhance the growth of *M. Caerulea* and increase peat mineralisation, this measure must be weight up carefully and must be restricted to areas having already dense stands of this grass species. It might also be worth considering the option of active removing of *M. caerulea* plant litter in autumn reducing substrate availability for methanogenesis or a total one-time sod removal before optimising water table and hence increase favourable conditions for a potential *Sphagnum* colonisation. Biomass removal was previously proposed by Augustin and Joosten (2007) in case of high CH_4 emissions due to stands of *Phalaris arundinacea* (Chapter 5.3), while sod cutting is a common measure to establish target species in different ecosystems (Jansen and Roelofs, 1996; Dorland et al., 2003; Fleischer et al., 2013).

6. Conclusion and Outlook

Evaluation of proposed hypothesis

To deepen the understanding of greenhouse gas fluxes from restored bog ecosystems and to compare between differing vegetation communites which established after rewetting, three subsites in a restored part of the Himmelmoor and one site with active peat mining were investigated.

It was shown, that the three differently vegetated subsites differ additionally in their hydrological regime, physical soil parameters and N-availability. These differences are anticipated to derive from former drainage ditches dividing the subsites, which were more or less successfully blocked as a restoration measure. The combination of theses different parameter results in significantly differences in CO_2, CH_4 and N_2O fluxes within the restored sites and between the restored and the active peat mining sites. These findings support the **Hypotheses 2 - 4** that GHG fluxes differ between the subsites and that N_2O is only emitted at the industrial extractions site and significant amounts of CH_4 are only emitted at the vegetated subsites.

The reestablishment of the CO_2 sink function by restoration was not successful, as all of the vegetated sites have net CO_2 emissions. Only the CO_2 budget of the *Sphagnum* subsite is close to a CO_2-neutral state. The hypothesis (**Hypothesis 1**) that the resored peat sites act as carbon sinks can thus not be confirmed.

As peatlands in temperate Europe are particularly vulnerable to a decrease in summer precipitation due to climate change (Charman et al., 2013), a drought treatment experiment was set up. The **Hypothesis (5)** that the CO_2 balance will be altered by drought depending on vegetation community can be confirmed as it was shown, by reduced Reco and GPP, that plant communities of the *Sphagnum* and the heath subsite are susceptible to drought while the purple moor grass site increased its productivity and *M. caerulea* gained significant amounts of biomass within only one year due to rainout treatment. Contrastingly, **Hypothesis 6** that the rainout treatment will have a significant effect on CH_4 and N_2O can not be supported. Here the missing effect of the shelter experiment on water table is a likely explanation as both fluxes depend on water table levels. However, CH_4 emissions of the purple moor grass site declined appreciably. In case of N_2O, it can be assumed from the results of the N-availability in the soils that N_2O fluxes would not increase even if water table decreases, due to N-imitation and the competition between plants and microorganisms.

It was hypothesised that summer drought will enhance peat mineralisation (**Hypothesis 7**).

As the N-availability increased due to rainout treatment, as well as Reco at purple moor grass site, it can be expacted that mineralisation of the peat was stimulated by drought. (Hypothesis 7 is accepted).

Implications for peatland management

This study demonstrates that despite restoration, the ecosystem functions of the vegetated sites are still affacted by the former human activies.

The GHG budgets of the restored sites are dominated by CH_4 fluxes resulting in extremely high emissions between 15 and 37 t CO_2-eq ha^{-1} $year^{-1}$ which was, to the present knowledge, never reported before for a bog ecosystem in temperate Europe. This can be explained by a combination of high water level, huge water level fluctuations and a high availability of fresh plant litter particularly from *M. caerulea*. As the purple moor grass site has the highest GHG emissions and *M. caerulea* is likely to gain biomass and spread out due to climate change, it is likely that the GHG emissions of the restored sites will even increase in the future. Additionally, it is likely that *M. caerulea* will spread at the industrial site after cessation of peat extraction and hence will determine the GHG fluxes at this site. Here an active introduction of target species is required as viable seeds are probably absent in the bare, old peat which is additionally highly recalcitrant (Quinty and Rochefort, 2003).

With regards to the extraordinary high GHG emissions it must be stated that improvement in restoration is needed. The results of this study showed, that climate change mitigation and adaptation mostly require the same measures, which may be:

- Lowering the water table to a level of -10 cm under soil surface to increase methane oxidation.
- Reducing water table fluctuations to impede inundation of fresh plant litter as a substrate for methanogenesis.
- Active decreasing of *M. caerulea* stands by sod cutting to establish target species communities instead.
- Enhancement of favourable conditions for the growth of typical bog plants, notably for *Sphagnum* species, by e.g. removal of shading plants like birches.
- Establishing *Sphagnum* dominated plant communities at the abandoned industrial extraction sites by special techniques for bare peat sites, such as the active introduction of target species and levelling of slopes to achieve an even distribution of water.

Applicability of the results and demand for research

In this study it was shown that there can be a huge spatial variability of GHG fluxes within a restored site and that for the estimation and upscaling of GHG budgets it is imperative to distinguish subsites if there are obvious differences in plant communities indicating differing hydrological regimes and soil parameters. The absolute values of the estimated GHG budgets in this study must be treated with some caution as only one year was studied and GHG fluxes can underlie large annual variations. This can also be expected from the differences in air temperature and precipitation between 2010 and 2011 and the fact that N-availability was increased by rainout treatment in 2010 but not in 2011. Thus for a better reliability of annual GHG budgets it is essential to extend the measurements to a multiannual time span. Additionally further research is needed to get a deeper insight in the special conditions that are responsible for the extremely high CH_4 fluxes dominating the GHG budget at the restored sites.

However, it can be assumed that the relatively differences between the plant communities surveyed in this study are applicable for similar plant communities in other restored bog ecosystems. The same holds true for the detected changes due to rainout treatment. This means for example that areas with a high coverage of *M. caerulea* will have higher GHG emissions than those with a high *Sphagnum* cover and that *M .caerulea* will profit from climate change also in other bogs. This study thus provides valuable implications for restoration and adaptation to climate change for the Himmelmoor in particular and for degenerated bog ecosystems in temperate Europe in general.

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Acknowledgement

First of all, I would like to thank my doctoral advisor Prof. Dr. Eva-Maria Pfeiffer for the academic supervision of this work and for supporting me in the choice of the field site and in deviation of initial plans of the experiment.

My special thanks go to Prof. Dr. Lars Kutzbach for guidance and support in the field of flux measurement and modelling.

I am most grateful to Dr. Claudia Fiencke, who supported me in any stages of this thesis from the beginning to constructive discussion of my manuscript.

Additionally my gratitude goes to:

- Sebastian R. Schmidt from the Group of Applied Plant Ecology (University of Hamburg) for fruitful cooperation within the KLIMZUG-Nord project. He gratefully performed the vegetation analysis at my measurement site in the Himmelmoor and kindly provided his data to me.
- Familiy Czerwonka from the peat company at the Himmelmoor and their staff for excellent cooperation, providing information on the history of the Himmelmoor and for their never-ending practical support of my field experiment.
- Birgit Grabellus, Birgit Schwinge, Susanne Kopelke, Madleen Greven, Mathias Schwarzer, Steffi Langer, Kira Kalinski, Norman Rüggen and Michael O´Connor for assistance in field and lab.
- Christian Wille, Volker Kleinschmidt and Peter Schreiber for technical support.
- The “School of Integrated Climate System Sciences (SICSS)” for accepting me as a member.
- Inken, Renuka, Sarah, Pete and Sebastian for sharing ups and downs of our PhD projects.
- My family in particular to my husband, who spend some weekends with me at the field side and in the lab whenever needed.

This PhD project was carried out within the climate research project “KLIMZUG-Nord”, which was funded by the Federal Ministry of Education and Research (BMBF).