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Discussions

High soil solution carbon und nitrogen concentrations in a drained Atlantic bog are reduced to natural levels by 10 yr of rewetting

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Abstract

Artificial drainage of peatlands causes dramatic changes in the release of greenhouse gases and in the export of dissolved carbon (C) and nutrients to downstream ecosystems. Rewetting anthropogenically altered peatlands offers a possibility to reduce nitrogen (N) and C losses. In this study, we investigate the impact of drainage and rewetting on the cycling of dissolved C and N as well as on dissolved gases over a period of 1 yr and 4 month, respectively. The peeper technique was used to receive a high vertical sampling resolution. Within one Atlantic bog complex a near natural site, two drained grasslands sites with different mean water table positions, and a former peat cutting area rewetted 10 yr ago were chosen.

Our results clearly indicate that drainage increased the concentration of dissolved organic carbon (DOC), ammonia, nitrate and dissolved organic nitrogen (DON) compared to the near natural site. Drainage depth further determined the release and therefore the concentration level of DOC and N species, but the biochemical cycling and therefore dissolved organic matter (DOM) quality and N species composition were unaffected. Thus, especially deep drainage can cause high DOC losses. In general, DOM at drained sites was enriched in aromatic moieties as indicated by SUVA[280] and showed a higher degradation status (lower DOC to DON ratio) compared to the near natural site. At the drained sites, equal C to N ratios of uppermost peat layer and DOC to DON ratio of DOM in soil solution suggest that the uppermost degraded peat layer is the main source of DOM. Nearly constant DOC to DON ratios and SUVA[280] values with depth furthermore indicated that DOM moving downwards through the drained sites remained largely unchanged. DON and ammonia contributed most to the total dissolved nitrogen (TN). The subsoil concentrations of nitrate were negligible due to strong decline in nitrate around mean water table depth. Methane production during the winter months at the drained sites moved downwards to areas which were mostly water saturated over the whole year (> 40 cm). Above these depths, the recovery of the

water table in winter months led to the production of nitrous oxide around mean water table depth at drained sites.

10 yr after rewetting, the DOM quality (DOC to DON ratio and SUVA[280]) and quantity were comparable to the near natural site, indicating the re-establishment of mostly pristine biochemical processes under continuously water logged conditions. The only differences occur in elevated dissolved methane and ammonia concentrations reflecting the former disturbance by drainage and peat extraction. Rewetting via polder technique seems to be an appropriate way to revitalize peatlands on longer timescales and to improve the water quality of downstream water bodies.

1 Introduction

Biogeochemical cycling of C and N in peatlands is determined by the water table position. Under pristine conditions C and N fixation exceed gaseous and dissolved losses (Hemond, 1983; Urban and Eisenreich, 1988; Limpens et al., 2008; Blodau, 2002) in these oxygen limited ecosystems. Thus, peatlands are able to accumulate and store C and N over millennia, making these ecosystems an important part of the global C and N cycle (Gorham, 1991; Limpens et al., 2006).

During the last centuries peatlands have been drained for agricultural purposes and peat extraction in Germany. Recently, nearly all peatlands in Germany are affected by drainage and are used mainly as grassland (40 %) and arable land (32 %; Federal Environment Agency, 2013).

The exposure of previously anaerobically stored peat to oxygen is associated with an increase in the C and N mineralization (Bridgham et al., 1998) with a substantial impact on greenhouse gas emission (Schafer et al., 2012; Flessa et al., 1998; Berglund and Berglund, 2011) and concentrations of water soluble C and N (Holden et al., 2004), changing these ecosystems from a net C and N sink to a source.

Compared to mineral soils, peatlands are characterized by high (20–50 mg L^{-1}) DOC pore water concentrations (Blodau, 2002). Even under pristine conditions, DOC can

contribute up to 10 % to the overall carbon export (Limpens et al., 2008) and drainage was shown to increase DOC concentration in peatlands (Wallage et al., 2006; Banas and Gos, 2004; Glatzel et al., 2003; Moore and Clarkson, 2007), causing an additional export of old DOC (Moore et al., 2013). In contrast to the former findings Kalbitz (2001) measured lower DOC concentrations in long term (> 200 yr) intensively used peatlands due to lower stock of soil organic carbon as main source of DOC. Additionally, intensive land use associated with increased peat degradation released aged and strongly microbiologically altered DOC (Kalbitz and Geyer, 2002) with a higher amount of aromatic moieties (Hoell et al., 2009; Kalbitz, 2001). These findings point to the importance of the peat degradation status for the DOC quality. The quality of DOC determines the bioavailability of DOC (Kalbitz et al., 2003) and the adsorption potential to the mineral phase or iron oxides (Kaiser et al., 1996). Thus, the internal cycling and transport of DOC within the soil profile influences the quality and quantity of DOC released to downstream water ecosystems (Fraser et al., 2001; Stutter et al., 2012), where DOC affects the light penetration, the microbial metabolism, the oxygen supply in the water column, the co-transport of metals and other pollutants via complexation, as well as the drinking water quality.

Drainage has a strong impact on N availability in peatlands. The increase in N mineralization with water table drawdown is related to an elevated release of ammonia, nitrate and DON (Venterink et al., 2002) leading to an eutrophication of previously N limited ecosystems. Ammonia is found to generally increase after drainage whereas nitrification, which is strongly pH dependent (Dancer et al., 1973), is thought to be restricted to less acidic peatlands (Holden et al., 2004). The export, especially of nitrate from drained peatlands, depends on flow pathways (Kieckbusch et al., 2006) and the capability for denitrification and therefore on available C and groundwater table position (van Beek et al., 2004). Especially in drained minerotrophic peatlands, nitrate concentrations in ditches were found to exceed drinking water and general water quality limits (Tiemeyer et al., 2007), thus causing a risk of eutrophication of downstream water bodies.

Rewetting is a common strategy to revitalize peatlands after peat cutting. Frequently, the main objective of rewetting is nature conservation and thus the establishment of flora and fauna typical for peatlands. Additionally, successful rewetting creates environmental conditions suitable for peat formation and C and N sequestration as well as for an improved water quality. Several factors determine whether rewetting leads to quasi natural soil solution concentrations. Firstly, the level of peat degradation has a strong impact on the release of DOC after rewetting (Zak and Gelbrecht, 2007). Secondly, a stable high water table is important to reach biochemical processes related to pristine peatlands as pronounced water table fluctuations throughout the year can have a large impact on DOC concentration (Kalbitz et al., 2002). Thirdly, time is an important factor to overcome the disturbance of drainage. Directly after rewetting, Worrall et al. (2007) found increased DOC concentrations compared to drained sites which can be explained by the former disturbances as well as by the enzymatic latch mechanism (Freeman et al., 2001). On longer timescales, Wallage et al. (2006; up to 4 yr) and Hoell et al. (2009; 20 yr) found lower concentrations of DOC compared to drained sites. Despite recovered water tables, the preceding oxygenation may even on longer timescales affect DOC production due to the enzymatic latch mechanism (Wallage et al., 2006).

In the case of N, rewetting may be a suitable way to reduce nutrient losses. This is particularly effective in the case of nitrate as rewetting inhibits nitrification and increases denitrification resulting in lower nitrate losses (Kieckbusch and Schrautzer, 2007). The denitrification potential is controlled by state of peat degradation, N load and temperature (Cabezas et al., 2012). However, even when nitrate could be substantially decreased the additional release of ammonia and DON may cause an overall net N release (Davidsson and Stahl, 2000). Even after rewetting, N mineralization was sometimes found not to decrease significantly (Venterink et al., 2002). Wind-Mulder et al. (1996) measured elevated inorganic N concentrations after rewetting harvested sites compared to pristine sites, and the distribution between ammonia and nitrate was determined by soil moisture. The duration of elevated N concentrations are important,

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because a successful revitalization of peatland typical vegetation requires N states related to pristine conditions.

The separate understanding of the cycling of DOC and dissolved N was improved over last decades, but many studies have either focused on dissolved C or on N. Kalbitz and Geyer (2002) found a different response of DOC and DON to peat degradation and pointed out the importance of combined DOC and DON investigations to understand the cycling of DOM in peatlands. They also addressed the data gap concerning DON in peatlands, which has especially not been filled in bogs. In addition, only few studies have focused the impact of both drainage and rewetting compared to natural conditions (Wallage et al., 2006; Glatzel et al., 2003) to fully understand the risks of drainage or drainage depths and the benefits of rewetting in relation to both natural and drained conditions.

Therefore, in this study we investigated the cycling of dissolved C and N in a peat bog complex for one year and dissolved gases (CO_2, CH_4, N_2O) for four months. Specifically, the impact of two different drainage levels and rewetting of a former harvested site (10 yr) in relation to a near natural bog site were studied to understand the impact of water table drawdown and preceding oxygenation prior to rewetting on the cycling of dissolved C and N. We assume that: (1) an increase in C and N mineralization after drainage increases concentrations of dissolved C and N species in the soil water; (2) the degradation of the uppermost peat layers at drained sites alters the quality of released DOM (DOC to DON ratio and aromaticity); (3) 10 yr of rewetting do not lead to quasi natural soil solution C and N concentrations.

2 Materials and methods

2.1 Site description

Our study was conducted in the "Ahlen–Falkenberger Moor" peat bog complex, which is located 20 km south of Cuxhaven (North Sea; 53°41′ N, 8°49′ E) in the federal state

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Lower Saxony (Germany). The long time averages of temperature and precipitation are 9.4 °C and 905 mm, respectively (period: 1971–2011; German Weather Service, 2012). With an area of 39 km^2 the "Ahlen–Falkenberger Moor" is one of the largest peat bog complexes in northern Germany.

5　The peatland grew in a depression of Pleistocene sands, partly mixed with gravel and clay. The bog formation started on former fen areas approximately 4000 B.C. and afterwards directly on Pleistocene sand (Schneekloth, 1970).

Since the middle Ages, small-scale peat-cutting was performed at the edges of the complex. Intense drainage started at the beginning of the 20th century. From 1939 to
10　1965 over 50 homesteads were established and land use was intensified (Ahrendt, 2012). Nowadays, the "Ahlen-Falkenberger Moor" bog complex is mainly used as grassland. Furthermore, from 1957 to 2002 industrial peat-cutting was conducted. These areas were rewetted after terminating peat extraction in 2002. In the center of the bog complex, there is an area with nearly natural conditions which has never
15　been drained for any purpose.

Four study sites were chosen to investigate the impact of the water table position combined with different land use on C and N cycling in the pore water. An intensive grassland site (*IG*), extensive grassland site (*EG*), a rewetted peat extraction site (*RW*) and a near natural site (*NN*) represent the dominant land use forms. At EG, a former
20　deeply drained extensive grassland site, ditches were partly closed and mean water table rose related to *IG*. In general, the sites differ in drainage, vegetation and management (Table 1) and therefore in the mean water table position. These differences are associated with different peat properties (Table 2).

Prior to our investigations, greenhouse gas emissions were measured from 2007 to
25　2009 at the *IG*, *EG* and *NN* site (Beetz et al., 2013).

2.2　Sampling and chemical analysis

Sampling was conducted between February 2012 and February 2013. Peeper technique (Hesslein, 1976) was used to sample soil solution in a high vertical resolution

via diffusion. This technique is often used to investigate peat pore water (Zak et al., 2004). In our case, the plexiglas peeper allows sampling of soil solution between 10 and 65 cm below the surface. Each peeper contains 12 vertical layered chambers with 5 cm spacing and a chamber volume of 50 mL (Fig. 1). Prior to field installation the chambers were filled with dionised water and covered with a 0.2 µm membrane filter (Gelman HT 200 tyffron, PALL, Pensacola FL, USA). To displace oxygen, the peepers were placed in deionised water, which was purged with N_2 for 2 days. The peepers were transported to the field sites in the same water containers. Peepers were allowed to equilibrate with the soil solution for approximately 4 weeks, leading to 14 dates of sampling within one year of investigation.

Three peepers were installed at each site. One of the three peepers was used for high vertical profile sampling (each chamber). Bulk samples (chambers 1 to 3: 10–22 cm, chambers 3–6: 25–37 cm and chambers 7–12: 39–65 cm) representing a rough vertical resolution were collected from two peepers. After removal the peepers were emptied immediately in the field. Between 10–15 mL aliquots were filled for each DOC/UV-Vis absorption, major ions and total nitrogen (TN) in 20 mL PE-vials and cooled during transport. When measurements could not be conducted within a few days, samples were stored at −18 °C until analysis. No further sample preparation was made.

Between November 2012 and February 2013 (4 times) samples were additionally analyzed for dissolved CO_2, CH_4 and N_2O. For these analyses 10 mL solution was filled in 20 mL air tight glass vials. The vials were flushed prior to sampling with N_2 and prepared with 1 mL $ZnCl_2$ (50 % w/v) to reduce microbial activity (Romero et al., 1999).

The water table position during the sampling period was recorded using Mini-Divers (Schlumberger Water Services, Delft, the Netherlands).

The peat soil organic carbon (SOC) and total nitrogen (N_t) content was measured on a LECO TrueMac CN (LECO Corporation, St. Joseph, Michigan, USA). Peat samples were taken with a resolution of 5 cm using a peat corer and were oven dried (40 °C), sieved and ground prior to measurement.

A DimaTOC 2000 (DIMATEC, Essen, Germany) was used to measure DOC concentration. DOC was calculated as the difference between total carbon (TC; measured via combustion at 850 °C) and total inorganic carbon (TIC; measured via combustion at 165 °C after acidification with H_3PO_4 −40 % w/v). DOC was measured in at least three replicates.

UV-Vis absorbance was measured on a Shimadzu-1800 spectrophotometer (Shimadzu Germany, Duisburg, Germany) between 200 and 700 nm in a 1 cm cuvette. The UV-absorbance is often used to assess the quality of DOM. Appropriate wavelengths are shown in Grayson and Holden (2012). In our study, we chose a wavelength of 280 nm, because Chin et al. (1994) reported a strong correlation between $SUVA^{280}$ and aromaticity derived by ^{13}C-NMR from fulvic acids. Therefore $SUVA^{280}$ is an effective way to receive DOC quality information. $SUVA^{280}$ was calculated as the absorbance at 280 nm divided by the corresponding DOC concentration and is given in $L\,mg^{-1}\,m^{-1}$.

Total nitrogen (TN) of soil solution was determined with a Total Nitrogen Analyzer (TN-100; Mitsubishi, Kanagawa, Japan) by reduced pressure chemiluminescence detection. Concentrations of major ions (NH_4^+, NO_3^-, SO_4^{2-} and Cl^-) were measured with an ion chromatograph 850 Professional (Metrohm, Filderstadt, Germany). Dissolved organic nitrogen (DON) was calculated as the difference between TN and dissolved inorganic nitrogen (DIN: NH_4-N + NO_3-N).

Concentrations of CO_2, CH_4 and N_2O were determined with a gas chromatograph 7890-A (Agilent Technologies, Loveland, USA) using the headspace method. Samples were shaken for 2 h at 21 °C prior to measurement to equilibrate headspace and solution. Concentration of dissolved CO_2, CH_4 and N_2O were calculated according to Henry's law using the solubilities of CO_2, CH_4 and N_2O at 21 °C given by Weiss and Price (1980), Weiss (1974) and Wilhelm et al. (1977) and taking into account headspace pressure and volume and the volume of the liquid phase.

Bulked samples were further analyzed for pH and electrical conductivity (WTW, Weilheim, Germany).

As the peeper technique is only applicable under soil moisture conditions near saturation, chambers above minimum water table were excluded for analysis during each of the 14 sampling periods. Additionally, depth profiles of chloride were used to identify upper chambers with insufficient contact between the membrane and the peat, which
5 were characterized by distinct lower chloride concentrations compared to the mean profile concentrations. The application of cattle slurry at *IG* in April 2012 allows us to analyze all chambers, despite deep water tables during this occasion. These results were discussed separately from the general data analysis.

3 Results and discussion

10 ### 3.1 Peat properties

At NN the vegetation (Table 1) and peat properties (Table 2) clearly indicate the pristine state of the upper peat layers. Additionally, (Beetz et al., 2013) measured recent carbon accumulation. Thus, despite the surrounding drainage and increase in land use intensity over the last decades, the *NN* site may be used as a reference site.
15 The drained sites (*IG*, *EG*) are characterized by a strongly degraded amorphous peat layer with high von Post indices and narrowed C to N ratios at the soil surface (Table 2). Narrow C to N ratios may be caused by a retention of N in these peat layers, possibly due to immobilization of N by microbes during peat decomposition (Wells and Williams, 1996) or by nitrogen fertilization in the past (*EG*) and present (*IG*). Basic
20 peat properties clearly indicate peat mineralization, especially in the uppermost layer, which is supported by substantial gaseous C losses measured by (Beetz et al., 2013). Additionally, elevated bulk density and slightly increased von Post indices at all peat layers compared to *NN* confirm that peat degradation by compaction and mineralization after drainage has reached deeper peat layers.
25 The rewetted site (*RW*) is characterized by floating peat layers with a depth of 65 cm. After rewetting, the upper parts of the degraded bog peat (15–55 cm) started to float

at the soil surface. On this peat *Sphagnum* spp. colonized and built up a 15 cm thick slightly decomposed peat layer since 2002. These upper peat layers are comparable to *NN* in their basic peat properties (Table 2).

3.2 Water table

Measured mean water table positions during the sampling interval differed distinctly between drained sites (*IG*: 45 cm, *EG*: 28 cm) and wet sites (*RW*: −3 cm, *NN*: −1 cm). Positive values represent the water table below surface and negative values above soil surface. Water table fluctuations during the sampling period had higher amplitudes at the drained sites (*IG*: 5 to 87 cm; *EG*: 2 to 62 cm) than at the wet sites (*RW*: −27 to 6 cm; *NN*: −10 to 5 cm; Fig. 2).

The oscillation of the peat at *NN* and *RW* buffered the amplitudes of the water table. Thus, even during dryer summer months, the water tables at these sites did not drop deeper than 6 cm below soil surface. In contrast, the drainage ditches (open at *IG* and partly closed at *EG*) and the active drainage pipes (at *IG*) led to low mean and minimum water table levels and fast water table fluctuations. Rain events in summer months triggered a strong and fast rise in the water table at *IG* and *EG*. This can be explained by physical peat properties after drainage (Silins and Rothwell, 1998). Degraded peat with high bulk densities (Table 2) is characterized by a finer pore size distribution, leading to an increased rise of the water table after rain events (Edom, 2001). Furthermore, the rapid drawdown of the water table after rain event peaks at *IG* showed that the drainage system worked efficiently at this site. Declining water tables after drainage move the oxic-anoxic boundary in peat layers downwards (Estop-Aragones et al., 2012). Deep water table positions during summer at our drained sites (*IG*, *EG*) indicated pronounced oxygen penetration in contrast to the wet sites (*RW*, *NN*).

At the wet sites (*RW*, *NN*) all peeper chambers could be sampled throughout the whole sampling period. As mentioned before pore water from the topsoil at *IG* and *EG* could not be sampled in dry conditions based on the minimum water table crite-

ria which was used to assess whether a chamber got sufficient contact with the soil solution during sampling periods. At *IG*, all peeper chambers could be used in 57 % of the 14 sampling dates, only chambers below 25 cm in 57 % of sampling dates and only chambers below 39 cm in 64 % of sampling dates. At *EG* all chambers could be used in 64 % of sampling dates, chambers below 25 cm in 86 % of sampling dates and chambers below 39 in 86 % of sampling dates. As mentioned earlier 10, 25 and 39 cm represent the upper boundary of the bulk samples. There is thus a lack of information due to sampling constraints for the upper soil solution of *IG* and *EG* during April to September. Differences between drained sites (*IG*, *EG*) and wet sites (*RW*, *NN*) are mainly driven by differences during moist periods of the investigation period.

3.3 Basic properties of the soil solution and concentrations of SO_4^{2-} and Cl^-

PH values around 4 at *IG*, *RW* and *NN* of the soil solution represent the acidic character of bogs (Table 3). The slight increase in pH values at *IG* may be caused by cattle slurry application, which usually has pH values around 7.

Low electrical conductivities (EC) at *NN* is typical for ombrotrophic peatlands. The EC values at *IG* and *EG* are 1.3 and 2.5 times higher than at *NN* due to an increased amount of nutrients by mineralization and/or fertilization. This agrees with Wind-Mulder et al. (1996), who found increased EC associated with higher nutrient and SO_4^- concentrations in post-harvested and natural bogs in Canada. Electrical conductivity at *RW* was comparable to *NN* (Table 3), indicating near natural conditions.

SO_4^{2-} concentrations strongly differed between wet and drained sites. Whereas SO_4^{2-} concentrations were low (usually < 1 mg L^{-1}) and without vertical pattern at the wet sites (Fig. 3a), concentrations at drained sites were 21 (*IG*) and 9 (*EG*) times higher. Sulfate concentrations were highest in summer. At *IG* SO_4^{2-} concentrations remained relatively high in colder and wetter periods, whereas at *EG* SO_4^{2-} concentrations in winter months drop sharply. At *IG* (top 55 cm) and *EG* (top 40 cm), topsoil pattern of SO_4^{2-} concentration are partly masked by lack of summer samples (Fig. 3a). Mean concen-

trations in SO_4^{2-} tended to decrease below a depth of 40 and 55 cm at *EG* and *IG*, respectively, where peat layers were mostly water saturated and SO_4^{2-} reducing processes seem to occur. The vertical and temporal SO_4^{2-} concentration patterns match the water table dynamics and associated oxygen penetration at drained sites, which allows oxidation of reduced sulfur compounds to SO_4^{2-}, whereas oxidation is suppressed at wet sites over the whole profile.

As mentioned before, Cl^- was used to determine whether chambers got sufficient contact throughout the sampling interval. With the exception of *IG* no vertical patterns was found at the sites (Fig. 3b). The constant depth profiles at our sites are in contrast to measurements by Beer and Blodau (2007) in an oligotrophic bog in Canada, who found a strong increase in the upper 65 cm. This profile pattern could mainly be explained by diffusion. Thus in our case, different transport mechanisms such as advective transport have to be considered especially at the drained sites.

3.4 Carbon

3.4.1 Dissolved organic carbon – quantities

Mean dissolved organic carbon (DOC) concentrations were $195.8 \pm 77.3\,mg\,L^{-1}$, $89.4 \pm 31.4\,mg\,L^{-1}$, $38.8 \pm 6.4\,mg\,L^{-1}$ and $48.6 \pm 16.3\,mg\,L^{-1}$ at *IG*, *EG*, *RW* and *NN*, respectively. Mean DOC concentrations at the drained sites were increased by a factor of 1.8 (*EG*) and 4 (*IG*) compared to *NN*. At the drained sites DOC concentrations showed a higher spatial and temporal variability than the wet sites (Fig. 4a). Overall, DOC concentrations ranged from $490\,mg\,L^{-1}$ to $14\,mg\,L^{-1}$. Deep drainage causes twice as high DOC concentrations as shallow drainage. The mean DOC concentrations were negatively correlated with mean annual water table positions. Fairly constant vertical DOC patterns were observed at *EG*, *NN* and *RW*, but concentrations at *IG* were elevated in the upper 35 cm (Fig. 4b). DOC concentrations at drained sites (*IG*, *EG*) exceeded

those of *NN* in upper profile down to 65 cm. Near natural DOC concentrations over the whole profile were found at *RW* after 10 yr of rewetting.

DOC concentrations of the near natural site (*NN*) were comparable to previous studies in natural temperate and boreal peatlands, which often found values of around 50 mg L^{-1} (Moore, 2003; Blodau et al., 2007).

DOC concentrations measured at the drained sites exceeded most values in the literature. Increased DOC concentrations (≈ 70 mg L^{-1}) at drained bogs in Poland were reported by Banas and Gos (2004), which were comparable to our site *EG* and rather low compared to our deeply drained site (*IG*). High DOC concentrations (> 200 mg L^{-1}) were reported in a German bog after an exceptionally dry summer (Glatzel et al., 2006). Our results confirm the hypothesis that long term drainage has raised DOC concentrations up to a depth of at least 65 cm.

Drainage increases oxygen availability (Estop-Aragones et al., 2012) and enzymatic activity (Freeman et al., 1996), activates phenol oxidase (Freeman et al., 2001) and enhances peat decomposition, which have led to elevated DOC concentrations. Thus, drainage has strongly enhanced DOC production over DOC consumption or export at *IG* and *EG*. Wallage et al. (2006) hypothesized that increased DOC concentration in drained UK upland peatlands compared to natural peatlands resulted from increased microbial activity. Furthermore, Clark et al. (2009) showed an increase in net DOC production with water table drawdown during the incubation of peat cores. Deep drainage promotes DOC production, as shown by extreme DOC concentrations in *IG* and a mean DOC concentration of more than twice that of *EG*. However, the DOC depth profile varies seasonally, in particular in the aerated soil layer (Adamson et al., 2001). Due to a lack of samples from the upper peat layers at *IG* and *EG* in dry summer periods, DOC concentrations in the uppermost soil solution may even have been underestimated in our study. The high DOC concentrations in the upper 35 cm at *IG* suggests that DOC was primarily produced in the uppermost peat layers (Mcknight et al., 1985) and seems to be diluted or partly consumed during advective transport to deeper horizons.

In contrast to our hypothesis that 10 yr of rewetting do not yet lead to quasi natural DOC concentrations, mean DOC concentrations at *RW* were comparable to *NN*. 10 yr of rewetting at our site has overcome initial conditions directly after rewetting, where DOC was found to increase in a drain blocked UK peatland (Worrall et al., 2007). An initial increase can be attributed to the mobilization of DOC produced prior to rewetting and to the "enzymatic latch" mechanism, which causes elevated DOC concentrations after the water table is restored (Worrall et al., 2007). On longer time scales (3–4 yr) DOC concentrations were found to decrease (Glatzel et al., 2003; Wallage et al., 2006) as a result of DOC store exhaustion. This decrease depends on the water table position and dynamics after rewetting. The polder technique at our site induces a water table near the peat surface throughout the year, which prevents the ingress of oxygen during summer leading to permanent anaerobic conditions and slows down C mineralization and DOC production as also found by Hoell et al. (2009). If the water table after rewetting is not restored to the soil surface – a common problem in many projects – increased DOC concentrations could still be found which becomes evident when comparing *RW* and *EG*. This is attributed to increased decomposition during the summer months, when the water table drops and allows increased peat mineralization and DOC production, which is mobilized after the water table recovers in moist periods (Kalbitz et al., 2002).

DOC concentrations at *RW* remain constant over depth, even in the more degraded peat layer, which had been aerated during peat harvesting. Zak and Gelbrecht (2007) could show increased DOC release with an increased degree of peat degradation following rewetting, pointing to an enhanced mobile DOC pool of the degraded peat. In contrast to our findings, Wallage et al. (2006) measured increased DOC concentration with depth after drain blocking and suggested a continued disturbance to DOC production/transport via the "enzymatic latch mechanism". A lack of vertical DOC pattern at *RW* suggests that neither an enhanced bioavailabilty nor the "enzymatic latch" mechanism could be found in the degraded peat layer, which is saturated over the whole year and has stopped enhanced DOC formation. Thus, long term rewetting (10 yr) associ-

ated with a near surface water table at *RW* leads to a DOC production and consumption or export which is closely related to *NN*.

3.4.2 Dissolved organic carbon – qualities

Mean SUVA^{280} values were elevated by around 10 % at the drained sites (*IG*: 3.8 ± 0.4 L mg^{-1} m^{-1}, *EG*: 3.6 ± 0.3 L mg^{-1} m^{-1}) compared to the wet sites (*RW* 3.3 ± 0.3 L mg^{-1} m^{-1}, *NN*: 3.3 ± 0.2 L mg^{-1} m^{-1}) with no differences between drained and wet sites, respectively. SUVA^{280} values ranged from 5.57 L mg^{-1} m^{-1} to 2.46 L mg^{-1} m^{-1} (Fig. 4c). Deeper drainage at *IG* has led to much higher DOC quantity but no change in DOC quality compared to *EG*. Within the soil profile, SUVA^{280} values were fairly constant at *EG*, *RW* and *NN*, and slightly elevated in the upper 35 cm at *IG* (Fig. 4d).

These results confirm our hypothesis that prolonged water table drawdown has led to a higher degree of aromaticity of DOC, which must have been mainly released from the highly degraded, intensely humified upper peat layer (high von Post indices and narrow C to N ratio at *IG* and *EG* compared to *NN*). DOC then percolates down the peat profile with only little (*IG*) or no alteration (*EG*) in aromaticity. As in our study, Hoell et al. (2009) also found elevated SUVA^{280} values in a drained, compared to a rewetted fen in Germany and Kalbitz et al. (1999) showed as well, that DOC sampled in more degraded peat was enriched in aromatic moieties. The differences in SUVA^{280} between drained sites (*IG*, *EG*) and wet sites were slightly smaller than those found by Kalbitz (2001) between fen areas of different land use history, possibly as a result of longer land use history (up to 200 yr compared to 60 yr in our case) and therefore a higher peat degradation level.

The stable SUVA^{280} profiles contrast findings in mineral soils under grassland where Don and Schulze (2008) found a decline in SUVA^{280} by around 30 % from 10 to 65 cm depth. This is attributed to missing adsorption of aromatic DOC moieties in bog peat, which lacks a mineral phase or iron oxides. Thus drainage of bog with thick peat layers bears a huge risk of additional DOC losses due to lacking retention capacities.

In contrast to our hypothesis, the DOC quality of *RW* and *NN* was comparable, which supports our findings for DOC concentrations that *RW* has reached a near natural DOC status after 10 yr of rewetting. Comparable pathways of DOC production and consumption under anoxic conditions at *RW* and *NN* lead to a nearly identical DOC quantity and quality, despite differences in land use history reflected in diverging peat properties.

3.4.3 Dissolved CO_2 and CH_4

Dissolved CO_2 was the dominant greenhouse gas during our investigations, but no general difference was found between drained (*IG*, *EG*) and wet sites (*RW*, *NN*; Fig. 5). The mean concentrations of CO_2-C decreased in order *NN*, *IG*, *RW* to *EG* with $16.8 \pm 4.4\,\mathrm{mg\,L^{-1}}$, $15.4 \pm 5.0\,\mathrm{mg\,L^{-1}}$, $10.4 \pm 1.6\,\mathrm{mg\,L^{-1}}$ and $7.8 \pm 4.5\,\mathrm{mg\,L^{-1}}$, respectively. Concentrations of dissolved CO_2-C tended to increase with depth at all sites.

In contrast, CH_4-C concentrations at the wet sites (*RW*: $4.4 \pm 1.3\,\mathrm{mg\,L^{-1}}$; *NN*: $3.0 \pm 0.9\,\mathrm{mg\,L^{-1}}$) were 11 to 22 times higher than at the drained sites (*IG*: $0.2 \pm 0.4\,\mathrm{mg\,L^{-1}}$; *EG*: $0.3 \pm 0.5\,\mathrm{mg\,L^{-1}}$). CH_4-C concentrations tended to increase with depth at all sites. While CH_4 was measured over the whole sampling depth at wet sites, it was only found at drained sites in water saturated zones below a depth of 40 cm (Fig. 5).

In general, the corresponding DOC concentrations were higher as the concentrations of dissolved CO_2-C and CH_4-C. Thus, DOC accounts for 74 % to 93 % of total dissolved carbon at our sites.

However, the soil solutions were enriched in dissolved CH_4-C and CO_2-C compared to equilibrium water concentration (CO_2-C: $0.33\,\mathrm{mg\,L^{-1}}$; CH_4-C: $0.05\,\mathrm{\mu g\,L^{-1}}$) at atmospheric pressure and 2 °C, the average air temperature during sampling between November 2012 and February 2013. The supersaturation of the soil solution leads to a degassing of CH_4-C and CO_2-C into the atmosphere at the water-air boundary.

An accumulation of microbially produced CO_2 and CH_4 with depth was also found elsewhere (Clymo and Bryant, 2008; Blodau and Siems, 2012) and was attributed

to low diffusion coefficients in water associated with a slow ongoing mineralization in deeper peat layers (Beer and Blodau, 2007).

Conditions for methanogenesis were suitable over the whole profile at *RW* and *NN* due to an absence of oxygen or nitrate and low ($< 1\,mg\,L^{-1}$) sulfate concentrations which may act as electron acceptors and suppress methanogenesis. Elevated net methane production can be expected at *RW* compared to *NN*, because CH_4-C concentration exceeds those at *NN*. Temporal water table draw down in summer months at the drained sites prohibits the methanogenesis during periods of high water tables in the winter months in the seasonally unsaturated peat layers at *IG* and *EG* above a depth of 40 cm. Knorr and Blodau (2009) found a lack of methanogenesis during drought and periods of up to 50 days after water table was recovered, due to replenishment of other electron acceptors. Below 40 cm the soil was almost constantly water saturated and the absence of nitrate (Fig. 6c) and declining sulfate concentrations (Fig. 3) seems to be generally suitable for methanogenesis although there may be energetic constraints (Beer and Blodau, 2007). Methanotrophic processes in the upper peat layers are likely to dominate, thus Beetz et al. (2013) did not measure any CH_4 emissions in winter at the drained sites.

3.5   Nitrogen

3.5.1   Dissolved nitrogen

TN decreases in the order *IG* ($11.9 \pm 3.7\,mg\,L^{-1}$) > *EG* ($5.2 \pm 1.8\,mg\,L^{-1}$) > *RW* ($2.2\pm 0.5\,mg\,L^{-1}$) $\approx$ *NN* ($1.6 \pm 0.4\,mg\,L^{-1}$). Overall, TN concentrations ranged from $21.4\,mg\,L^{-1}$ to $0.9\,mg\,L^{-1}$.

At all sites DON was the dominant nitrogen species (Fig. 6), ranging from 33 % to 100 % of TN. Mean DON concentrations were 9.2 ± 3.2, 4.1 ± 1.3, 1.2 ± 0.4 and $1.4 \pm 0.3\,mg\,L^{-1}$ at *IG*, *EG*, *RW* and *NN*, respectively and ranged from 19.88 to $0.6\,mg\,L^{-1}$. The depth profiles of DON were comparable to DOC and remained nearly constant at *EG*, *RW* and *NN*. At *IG*, DON values were higher in the top 30 cm (Fig. 6a).

NH$_4$-N was the dominant inorganic N species, ranging from 0 % to 67 % of TN. Mean NH$_4$-N concentration decreased in the order *IG* $(2.3 \pm 1.4\,\text{mg}\,\text{L}^{-1})$ > *RW* $(1.0 \pm 0.5\,\text{mg}\,\text{L}^{-1}) \approx$ *EG* $(0.9 \pm 0.7\,\text{mg}\,\text{L}^{-1})$ > *NN* $(0.2 \pm 0.3\,\text{mg}\,\text{L}^{-1})$ and ranged from 9.14 mg L^{-1} to detection limit. Comparing the concentrations in 10 cm and 65 cm depth,
5 NH$_4$-N tends to increase at all sites with depth by a factor of 1.5 to 6 (Fig. 6b). At *NN* and *RW* the increase was nearly linear. In contrast, the vertical pattern at *IG* and *EG* showed the lowest NH$_4$-N concentration in upper aerated peat layer and increases constantly at a depth below the mean water table at each site (Fig. 6b).

NO$_3$-N was only present at drained sites (*IG*, *EG*). At *NN* and *RW* no or only
10 trace amounts of NO$_3$-N could be measured. Average NO$_3$-N concentrations were $0.5 \pm 1.0\,\text{mg}\,\text{L}^{-1}$ at *IG* and $0.2 \pm 0.4\,\text{mg}\,\text{L}^{-1}$ at *EG* and ranged from 5.9 mg L^{-1} to detection limit. Highest NO$_3$-N concentrations were found in 10 cm depth and decreased downwards to 40 cm depth (Fig. 6c). Below 40 cm depth no NO$_3$-N was found.

Slurry application at *IG* in April 2012 led to a spike in nitrogen concentrations in
15 the form of NO$_3$-N (Fig. 7). Compared to the colder and wetter periods of the year the concentration of NO$_3$-N was increased by a factor of approximately 37, but only in the soil solution above the mean water table. Surprisingly, concentrations of DON and NH$_4$-N were unaffected by slurry application although considerable amounts of DON and NH$_4$-N had been added to the peat surface. The N response of the peat and the
20 N profile (Fig. 7) can be explained by fast nitrification of applied N followed by strong denitrification in the soil profile above groundwater level.

In this study, the near natural site (*NN*) showed the lowest concentration of all measured N species, reflecting the nitrogen-poor conditions of pristine bogs. As in our study, Bragazza and Limpens (2004) found that DON is the dominant N species in
25 six European bogs. DON concentrations measured during this study were comparable to DON concentrations reported by Moore et al. (2005) for an ombrotrophic bog in Canada. High DON concentrations are typical for peatlands, where anaerobic conditions constrain mineralization of organic nitrogen compounds. Under acidic waterlogged conditions NH$_4$-N is the main inorganic N compound, because nitrification rates

are low due to low pH values and the absence of available oxygen. NH_4-N concentrations during our study are lower than reported by Hemond (1983) in an ombrotrophic bog in Massachusetts and comparable to ombrotrophic bogs in Canada (Wind-Mulder et al., 1996). Despite low ammonification under water logged conditions, over longer

5 time periods NH_4-N accumulates in deeper peat layers at *NN*.

In general, long-term water table drawdown has led to increased concentrations of all measured nitrogen compounds (*IG, EG* vs. *NN*). Measured DON concentrations at drained grassland sites are in the range of agricultural fen areas in Germany (Kalbitz and Geyer, 2002), but exceed concentrations in agriculturally used mineral soils

10 (Siemens and Kaupenjohann, 2002). This emphasizes the importance of the DON export from agriculturally used bogs. The conditions for ammonification are more suitable at our drained grassland sites, leading to elevated NH_4-N concentrations over the whole profile under aerobic conditions. This is supported by incubation experiments by Williams and Wheatley (1988), who found a higher amount of ammonifying bacte-

15 ria under aerobic than under anaerobic conditions. However, the aeration of the upper peat layer during dryer seasons of the year further allows nitrification in the upper peat layer of drained sites. The nitrification potential can be assumed to be lower at our grassland sites compared to a nutrient rich fen area used as grassland in North East Germany, where NO_3-N was the dominating inorganic N compound (Tiemeyer et al.,

20 2007). Lower NH_4-N levels in aerated peat layers compared to more saturated deeper peat layers indicate changes in the net N release. N uptake by plants and nitrification of NH_4-N may have led to the lower NH_4-N concentration, whereas below mean water table depth N uptake by plants is reduced and nitrification does not occur.

The application of cattle slurry in April 2012 indicates a fast nitrification of added

25 NH_4-N and organic N. A strong increase in NO_3-N content in the upper peat layer following fertilization was also shown by van Beek et al. (2004).

Overall, the risk of NO_3-N losses at our drained sites seems to be negligible. The strong decline in NO_3-N after fertilization (Fig. 7) and during periods without fertilization (Fig. 6c) indicates a strong denitrification potential at our drained sites. The areas of

denitrification in the peat profile follows the groundwater table (van Beek et al., 2004) and consumes all NO_3-N up to a depth of 40 (Fig. 6c) to 50 cm (Fig. 7) where no NO_3-N was found.

Furthermore, all nitrogen species at *IG* were twice as high as at *EG*. Thus, the drainage level further determines the height of N concentrations by duration and depth of water table drawdown.

In contrast to DOC, after 10 yr of rewetting the nitrogen state of soil solution still differed between the rewetted (*RW*) and near natural site (*NN*). Despite equal DON concentrations at *RW* and *NN*, NH_4-N concentration at *RW* exceeded those at *NN* by far. Wind-Mulder et al. (1996) also showed that rewetted post-harvested bogs were characterized by elevated NH_4-N concentrations compared to intact bogs. The NH_4-N concentrations doubled from 10 to 25 cm (Fig. 6b), indicating a NH_4-N uptake in the upper less decomposed peat layers. Higher NH_4-N concentrations were especially found in the more strongly degraded peat (15–65 cm) which is in agreement with Zak and Gelbrecht (2007) who showed increased NH_4-N released from degraded peat after rewetting. Thus, it can be assumed that the elevated NH_4-N concentrations are a "memory effect" of the former drainage and mineralization of this peat layer during harvesting and prior to rewetting.

Although mean NH_4-N concentrations of *EG* and *RW* were comparable, different processes lead to these findings. Ammonification is likely higher at *EG* than *RW*, but vegetation with a high biomass and therefore higher N demand by plants as well as nitrification of NH_4-N at *EG* may have decreased NH_4-N levels in the upper peat layers.

3.5.2 Dissolved N_2O

N_2O was only found at drained sites and ranged from 0 to 688 μg L^{-1} at *IG* and 0 to 30 μg L^{-1} at *EG*. N_2O concentrations were highest in November and December 2012 in zones of water table fluctuation during the sampling period, in general above 40 cm depth, where no CH_4 was present (Fig. 5).

Due to water table fluctuations saturated and unsaturated conditions alternate in the upper peat layers at *IG* and *EG*. These alternations favor processes like nitrification with buildup of nitrate and denitrification with consumption of nitrate. Figure 7 clearly indicates the production of nitrate within the unsaturated zones and strong consumption

5 of nitrate in zones of water table fluctuation (10 to 40 cm). Both processes produce N_2O (Wrage et al., 2001) and, indeed, Beetz et al. (2013) observed N_2O emissions at *IG* and *EG* during the winter periods in 2008 and 2009. The production of N_2O was restricted to periods directly after water table rise to soil surface when nitrate is present. During longer periods of water saturation N_2O formation declined, probably due to ceased

10 nitrification and thus exhaustion of available nitrate.

3.6 DOC to DON ratios of soil solution

The DOC to DON ratio of drained (*IG*, *EG*) and wet sites (*RW*, *NN*) were clearly different. Drained sites were characterized by narrower and less variable DOC to DON ratios (*IG*: 21.5±1.9; *EG*: 21.9±1.6) than the wet sites (*RW*: 35.4±7.1; *NN*: 35.6±4.9).

15 At all sites the DOC to DON ratio remains constant over depth (Fig. 8). A lower DOC to DON ratio at *IG* and *EG* compared to *NN* indicates an increased degradation level of the released DOM (Currie et al., 1996) and agrees with the increased SUVA[280] values (Austnes et al., 2010) at the drained sites and supports our hypothesis that drainage alters the DOM quality.

20 Comparing the DOC to DON ratio with the C to N ratio of peat can help to identify the origin and turnover of DOC within the soil profile. At the wet sites, the DOC to DON ratio was lower than the C to N ratio of the peat in the uppermost and corresponding peat layer (Fig. 8). At *NN* the corresponding C to N ratio of the transition zone between peat and organic sediment (39–65 cm) equals the DOC to DON ratio in soil solution.

25 However, DOM enriched in N compared to peat may result from N immobilization by microbes (Bowden, 1987), microbial N recycling (Broder et al., 2012) of a small N pool (Urban and Eisenreich, 1988) and therefore from an increased input of the microbial biomass. Another possibility may be a predominant release of low-molecular-weight

organic compounds, as indicated by lower SUVA[280] values (Chin et al., 1994), from the peat to soil solution, which were enriched in N (Qualls and Haines, 1991).

At the drained sites (*IG*, *EG*), the DOC to DON ratio of the soil solution equals the C to N ratio of the most degraded upper peat layer in 0–10 cm depth (Fig. 8). Equal C to
5 N ratios of the solid phase and DOM in soil solution from the forest floor were found by Michalzik and Matzner (1999). They suggest a predominant release of DOM from the bulk soil organic matter rather than specific sources (e.g., microbial biomass).

Furthermore, throughout the profile the DOC to DON ratio remained narrower than the C to N ratio of the corresponding peat layers (Fig. 7). This finding supports our
10 analysis of SUVA[280] observations that at the drained sites DOM is produced in the degraded topsoil during peat degradation and percolates through the profile without microbial alteration. DOM in peatlands is mainly built of recalcitrant humic substances (Zak et al., 2004) so that DOM from degraded peatlands is not readily bioavailable (Kalbitz et al., 2003). Drained degraded peatlands therefore bear a risk of high carbon
15 losses by ground- and surface waters.

4 Conclusions

In the "Ahlen–Falkenberger Moor" peat bog complex, long-term water table drawdown by drainage has caused severe peat degradation associated with a strong increase in DOC and N species concentrations (*IG*, *EG* vs. *NN*). Mean DOC concentrations at the
20 intensively used grassland site (*IG*) were among of highest ever reported (196 mg L^{-1}) and may lead to an increased DOC export under this type of land use. Equal C to N ratios of the uppermost peat layer and soil solution DOC to DON ratios indicate that DOC was mainly produced in the uppermost peat layer, percolating afterwards through the profile without measurable chemical alteration during advective transport.
25 Deep drainage (*IG* vs. *EG*) strongly increased the amount of DOC and N species in the soil solution but did not affect the quality of DOM (SUVA[280], DOC to DON ratio).

The concentration of subsoil NO_3-N at drained sites was negligible, even after fertilization, due to a strong denitrification potential around the mean water table, which was associated with the formation of N_2O at the drained site. Thus dissolved N export will occur as DON and NH_4-N losses.

After 10 yr of rewetting, the soil solution has clearly reached quasi natural conditions (*RW* vs. *NN*). Main differences occur only in elevated concentration of dissolved methane and an increased nutrient supply (NH_4-N), which was attributed to the historical land use. The former disturbance was overcome due to the colonization of bog typical vegetation at *RW* during the last 10 yr. Bog restoration at former peat cutting areas via polder technique seems to be an appropriate way for the successful revitalization of bogs and improves the water quality which is exported to downstream water ecosystems.

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Table 1. Vegetation, drainage and management information of the study sites.

Site	Vegetation	Drainage	Fertilization	Mowing
NN	*Sphagnum* spp. *Eriophorum angustifolium* HONCK *Drosera rotundifolia* L.	No (Possibly influenced by surrounding drainage)	No	No
IG	*Holcus lanatus* L. *Lolium perenne* L.	Drainage pipes Drainage ditches	Cattle slurry (Apr, May, Jul, Aug)	4 times (May, Jun, Aug, Sep)
EG	*Agrostis stolonifera* L. *Juncus effusus* L. Mosses	Drainage ditches (closed since 2003/2004)	No	Once (Aug)
RW	*Sphagnum* spp. *Eriophorum vaginatum* L. *Molinia caerulea* L.	Polder, Water level regulation at the outlet (rewetted since 2002)	No	No

Table 2. Soil properties (0–100 cm) of the study sites in the "Ahlen-Falkenberger Moor" peat bog.

Depth (cm)	SOC (%)	N_t (%)	C to N ratio	pH	BD (g cm^{-3})	von Post
Near-natural site (*NN*): Ombric Fibric Histosol[a], peat layer thickness: 440 cm						
0–40	46.7	0.9	53	3.9	n.d.	1
40–50	46.1	0.9	60	4.0	0.05	1
50–100[b]	49.3	1.8	28	5.0	0.04	n.d.
Intensive grassland site (*IG*): Ombric Fibric Histosol (Drainic)[a], peat layer thickness: 330 cm						
0–12	46.2	2.3	20	5.3	0.26	10
12–33	48.9	1.2	41	3.9	0.13	5
33–50	49.2	0.9	54	4.1	0.12	7
50–100	50.2	0.9	58	4.0	0.10	3
Extensive grassland site (*EG*): Ombric Fibric Histosol (Drainic)[a], peat layer thickness: 340 cm						
0–10	47.3	2.4	20	5.2	0.31	9
10–19	49.1	1.3	40	5.0	0.12	5
19–50	48.3	1.0	50	4.0	0.09	3
50–60	48.0	1.0	49	3.8	0.10	5
60–100	49.2	0.9	54	4.0	0.07	2
Rewetted site (*RW*): Ombric Floatic Hemic Histosol[a], peat layer thickness: 260 cm						
0–5	42.7	0.9	47	n.d.	n.d.	1
5–15	44.2	1.0	44	n.d.	n.d.	2
15–55	46.5	1.2	40	n.d.	n.d.	6
55–65[c]	51.8	1.0	52	n.d.	n.d.	5

n.d. = not determined; BD = bulk density; SOC = content of soil organic carbon; N_t = content of total nitrogen; C to N ratio = SOC to N_t ratio.
[a] IUSS Working Group WRB (2007).
[b] Organic sediment.
[c] 65–100 open water body.

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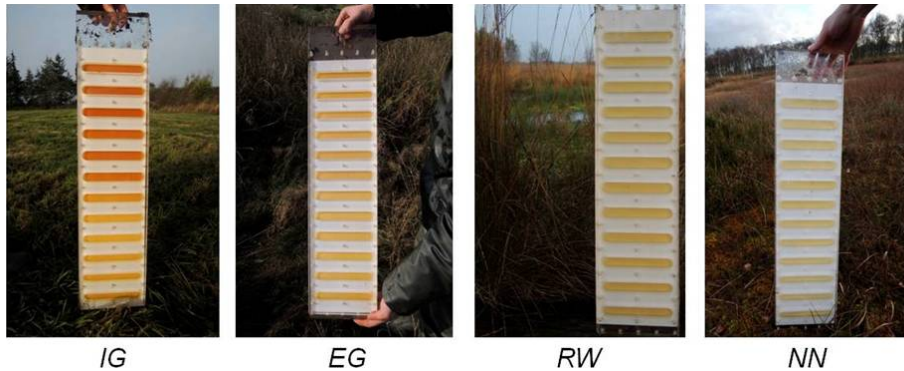
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Table 3. pH-Values, electrical conductivity (EC) and concentration of sulfate (SO_4^{2-}) and chloride (Cl^-) of the soil solution (means ± standard deviations).

Site	pH	EC [$\mu S\,cm^{-1}$]	SO_4^{2-} [$mg\,L^{-1}$]	Cl^- [$mg\,L^{-1}$]
NN	4.0 ± 0.2	83.2 ± 15.0	0.8 ± 0.8	12.7 ± 1.5
IG	4.6 ± 0.3	204.0 ± 71.5	16.7 ± 8.4	19.5 ± 9.4
EG	4.1 ± 0.2	104.1 ± 35.3	7.0 ± 6.2	9.7 ± 4.0
RW	4.1 ± 0.2	77.9 ± 11.6	0.5 ± 0.7	14.0 ± 2.4

Fig. 1. Pictures of peepers at drained (*IG*, *EG*) and wet (*RW*, *NN*) sites (sampling date: October 2012).
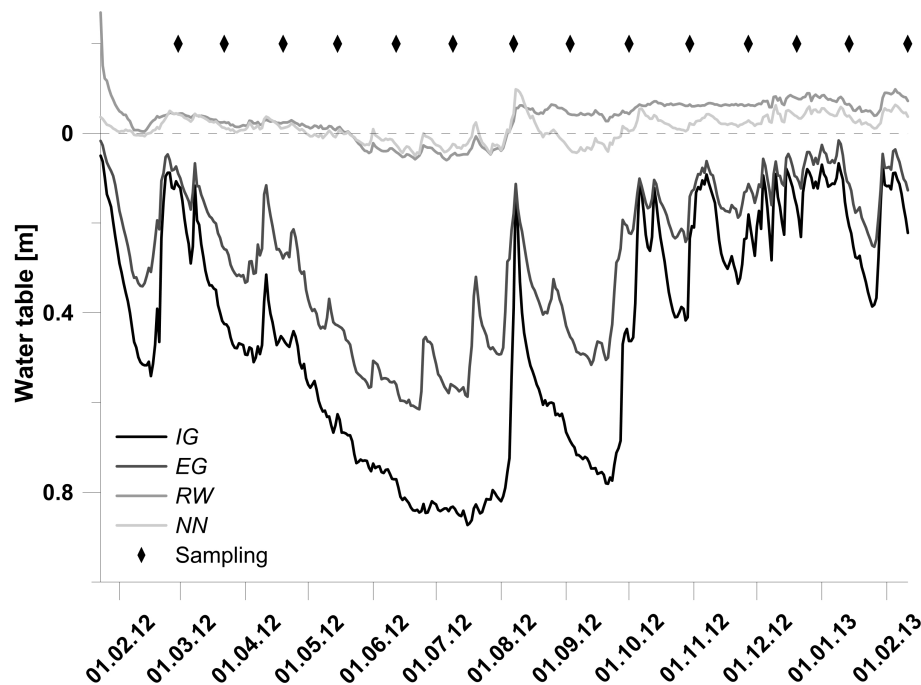
Fig. 2. Water table positions at the study sites during sampling period February 2012–February 2013. The dashed line represents soil surface.

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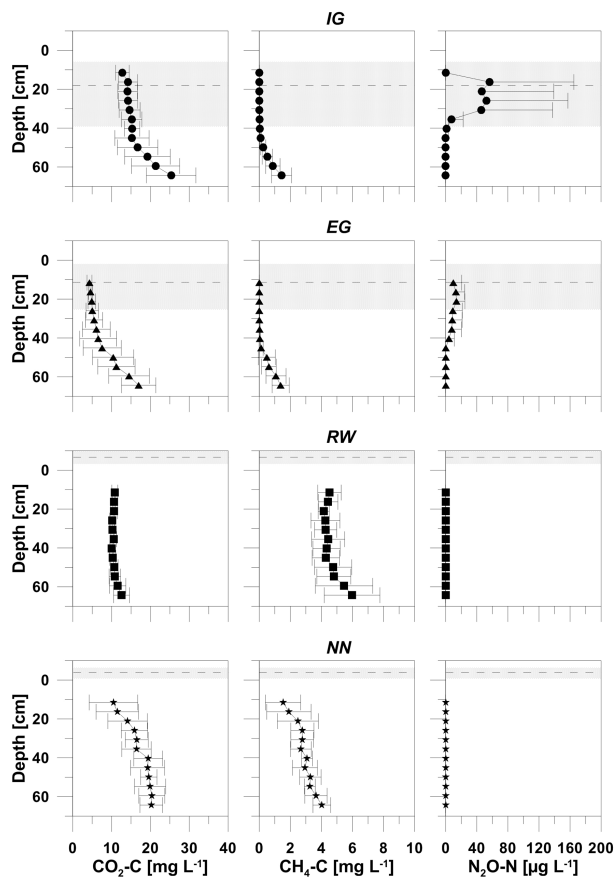
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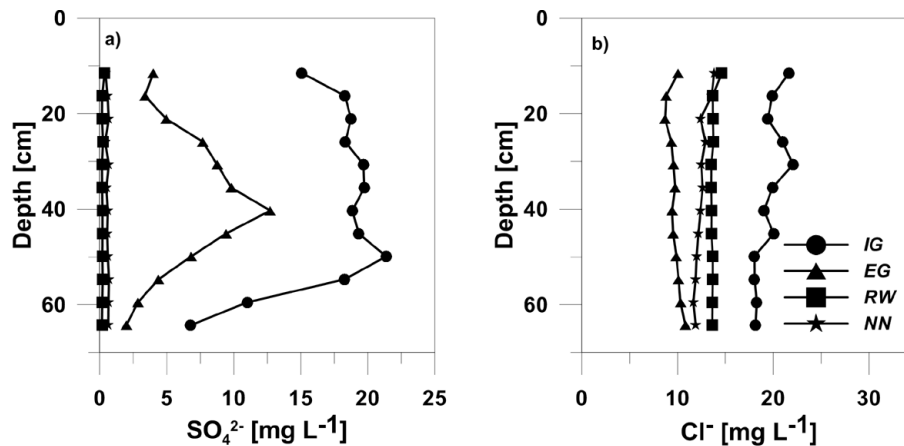
Fig. 3. Profile concentrations of **(a)** sulfate and **(b)** chloride at the study sites (means).
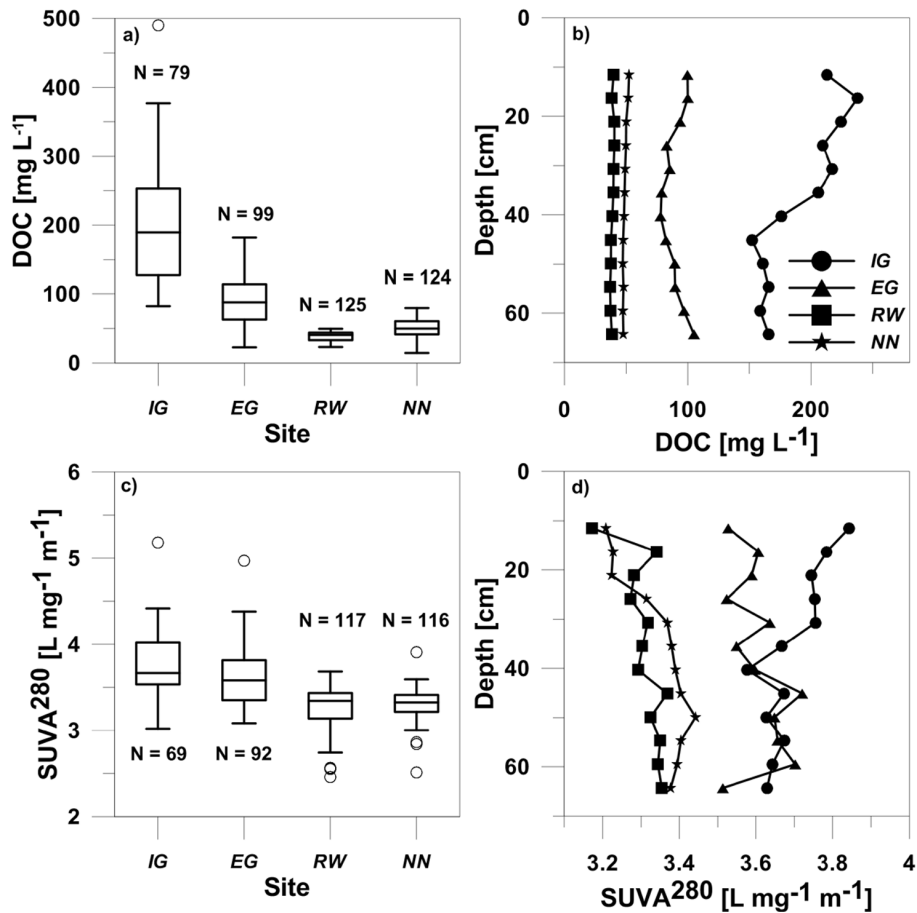
Fig. 4. Variability of **(a)** DOC-concentrations and **(c)** SUVA^{280} at the study sites (Central cross-bars represent the median, the boxes the 75th and 25th percentile and the circles extreme values) of bulk samples and profile of measured **(b)** DOC-concentration and **(d)** SUVA^{280} (means).

Fig. 5. Concentrations of the dissolved gases CO_2-C (left), CH_4-C (center) and N_2O-N (right; mean values per depth and standard deviation). The dashed lines represent the mean water table and the dotted area the range of the water table during the sampling period (November 2012–February 13).

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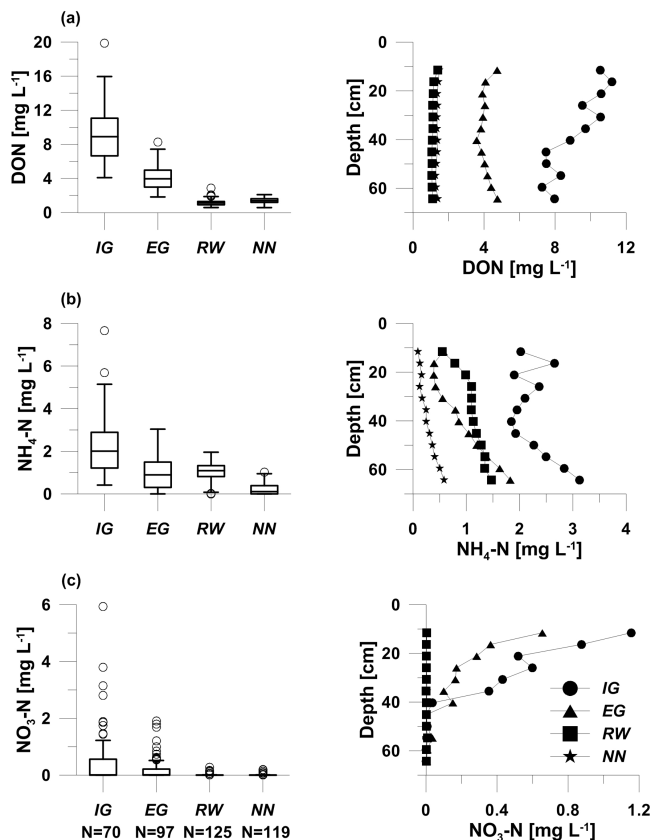Tables   Figures

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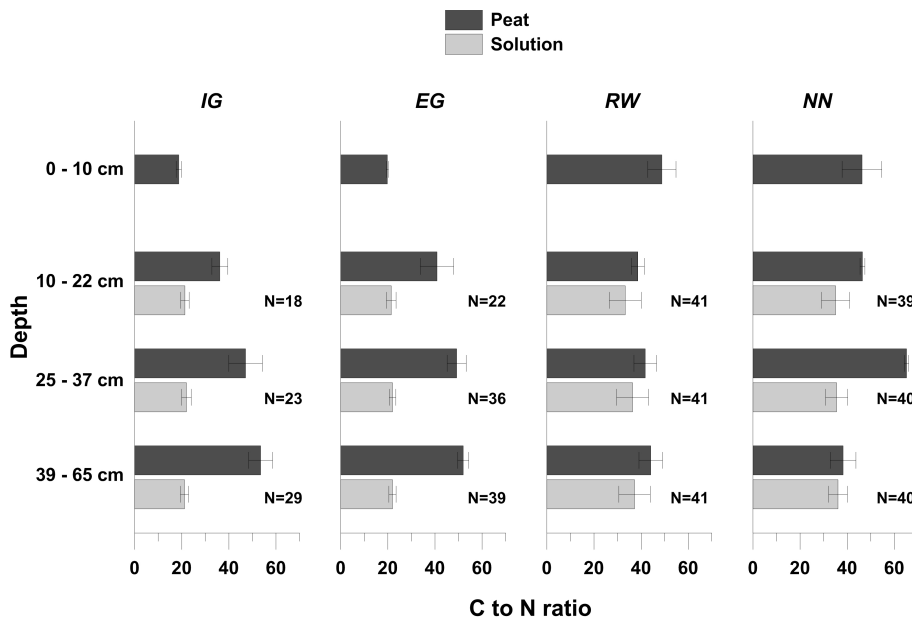Printer-friendly Version

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Fig. 6. Concentration of **(a)** DON, **(b)** NH_4-N and **(c)** NO_3-N at the study sites in the bulk samples (left; Central crossbars represent the median, the boxes the 75th and 25th percentile and the circles extreme values) and vertical profile of concentrations (right; means). Sampling date after cattle slurry application excluded (April 2012).

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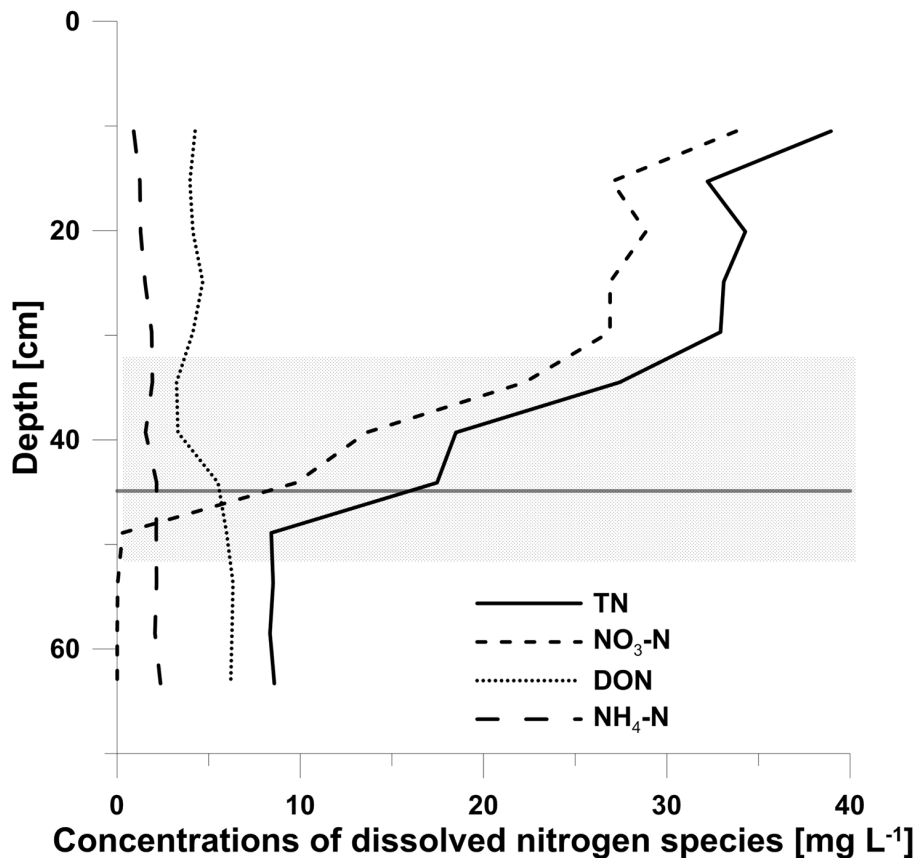
Fig. 7. Vertical profile of total nitrogen (TN), dissolved organic nitrogen (DON), NO_3-N and NH_4-N concentrations after cattle slurry application in April 2012. The horizontal line represents the mean water table position and the dotted area the range of the water table during the sampling interval.

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Fig. 8. Ratio of DOC to DON in the soil solution and the C to N ratio of uppermost and corresponding peat layers ($n = 3$). Bulk solution means of 3 chambers (10–22 cm and 25–37 cm) and 6 chambers (39–65 cm) over sampling period (February 2012–February 2013).