Greenhouse gas balance of an establishing *Sphagnum* culture on a former bog grassland in Germany

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SUMMARY

The cultivation of *Sphagnum* mosses on re-wetted peat bogs for use in horticulture is a new land use strategy. We provide the first greenhouse gas balances for a field-scale Sphagnum farming experiment on former bog grassland, in its establishment phase. Over two years we used closed chambers to make measurements of GHG exchange on production strips of *Sphagnum palustre* L. and *Sphagnum papillosum* Lindb. and on irrigation ditches. Methane fluxes of both *Sphagnum* species showed a significant decrease over the study period. This trend was stronger for *S. papillosum*. In contrast, the estimated CO_2 fluxes did not show a significant temporal trend over the study period. The production strips of both *Sphagnum* species were net GHG sinks of 5–9 t ha^{-1} a^{-1} (in CO_2-equivalents) during the establishment phase of the moss carpets. In comparison, the ditches were a CO_2 source instead of a CO_2 sink and emitted larger amounts of CH_4, resulting in net GHG release of ~11 t ha^{-1} a^{-1} CO_2-equivalents. We conclude that Sphagnum farming fields should be designed to minimise the area covered by irrigation ditches. Overall, Sphagnum farming on bogs has lower on-field GHG emissions than low-intensity agriculture.

KEY WORDS: carbon dioxide, ditches, methane, paludiculture, peatlands, Sphagnum farming

INTRODUCTION

In 2012, approximately 10 % (corresponding, in CO_2-equivalents, to 5.4 Gt a^{-1}) of all global anthropic greenhouse gas (GHG) emissions were released from farmed lands and the absolute magnitude of emissions has been increasing since the 1960s (FAO 2015). The agricultural utilisation of peatlands has been identified as a hotspot of agricultural GHG emissions, because it leads to the release of large amounts of carbon dioxide (CO_2) and nitrous oxide (N_2O) as a consequence of peat drainage and fertilisation (Maljanen *et al.* 2010, Couwenberg *et al.* 2011). Most temperate peatlands have been drained (Joosten & Clarke 2002). In Germany, for instance, 75 % of the peatland area is under agricultural use. Although this area represents only 6 % of the country's agricultural land, it accounts for more than one third of the total GHG emissions from agriculture (including emissions from ruminants, Umweltbundesamt 2014). In contrast, pristine mires sequester CO_2 from the atmosphere while emitting methane (CH_4), leading to a net climatic effect of only a slight warming or a slight cooling, depending on the timescale (Frolking *et al.* 2006, Franzén *et al.* 2012, Beetz *et al.* 2013). In addition to GHG release, the drainage of peatlands also leads to peat mineralisation, subsidence and compaction with a number of associated problems (e.g. increased flood risk, Joosten *et al.* 2012).

In a world with a growing human population the pressure to utilise large parts of the land surface increases. For this reason, alternative land use options for peatlands that are compatible with high water tables are currently being developed to provide a sustainable form of peatland utilisation. The concept of peat-conserving land use (paludiculture) enables biomass production while avoiding peat mineralisation and high GHG emissions (Joosten *et al.* 2015, Wichtmann *et al.* 2016).

Pilot field studies in the temperate zone have investigated the feasibility of paludiculture in fens with reed, cattail, sedges or alder (Wichtmann & Joosten 2007) and in bogs with peat mosses (Gaudig *et al.* 2014b). The harvested (above-ground) biomass may substitute fossil fuel resources as a bioenergy source or as raw material for industrial purposes (Wichtmann & Joosten 2007, Kuhlman *et al.* 2013, Vaičekonyte *et al.* 2014). The reduction of GHG emissions by paludicultures has been tested for reed in a temperate fen ecosystem (Günther *et al.* 2015) and in mesocosm experiments (Karki *et al.* 2014).

The results so far indicate that the harvesting of reed plants does not detract from the positive effects of peatland re-wetting. On re-wetted peat bogs, *Sphagnum* biomass from paludiculture can provide a sustainable alternative to the fossil ‘white peat’ which is extracted on a large scale from pristine peat bogs for use in horticulture (Emmel 2008, Blievernicht *et al*. 2012, Wichmann *et al*. 2012, Pouliot *et al*. 2015). While the biomass of reed has already been harvested on a small scale for many centuries (e.g. as roofing material, Köbbing *et al*. 2013), the cultivation of *Sphagnum* mosses is a new land use strategy. During the first pilot study in Germany, *Sphagnum* mosses were spread on the peat surface of a cut-over bog and covered with straw mulch before the area was re-wetted (Gaudig *et al*. 2013/2014, Beyer & Höper 2015). This resulted in the establishment of a vigorously growing *Sphagnum* lawn after 3.5 years (Gaudig *et al*. 2014a, Gaudig *et al*. 2017). First measurements of the GHG exchange indicate a slight overall cooling effect during the main growing phase (five to six years after installation) of the established *Sphagnum* lawn (Beyer & Höper 2015).

A number of studies find CH_4 emission peaks from peatlands within the first few years after re-wetting (Höper *et al*. 2008, Wilson *et al*. 2009, Hahn *et al*. 2015). For this reason, the GHG balance of the establishment phase of the mosses has to be included in the assessment of the whole production cycle of Sphagnum farming. However, so far no data are available regarding the GHG exchange during this first phase of Sphagnum farming. Also, the GHG emissions of other structural elements of Sphagnum farming fields (such as irrigation ditches and causeways) are unknown, since Beyer & Höper (2015) focused exclusively on the *Sphagnum* production area. It is known that drainage ditches on peatlands often emit large amounts of CH_4 (e.g. Minkkinen & Laine 2006, Wilson *et al*. 2009, Schrier-Uijl *et al*. 2011). Therefore, it seems reasonable to expect that the irrigation ditches of Sphagnum farming fields function similarly. If this were true, high GHG emissions from a small proportion of the total area could counteract the emission reductions of Sphagnum farming compared to conventional agriculture on peatlands.

Here, we provide the first GHG balances for a field-scale Sphagnum farming experiment on former bog grassland, in its establishment phase. We answer the following questions:

(1) Does the Sphagnum farming field show a peak of CH_4 release during the early stages of vegetation establishment, as has been found for other systems during re-wetting (Glatzel *et al*. 2011, Hahn-Schöfl *et al*. 2011, Hahn *et al*. 2015)?
(2) What is the contribution of the single structural elements (e.g. ditches, production strips) of the Sphagnum farming system to its GHG balance?
(3) Does Sphagnum farming reduce GHG emissions compared to the former use as intensively used bog grassland?

METHODS

Study site

A field-scale experiment on Sphagnum farming was established in north-western Germany (53° 15.80′ N, 08° 16.05′ E, Figure 1a). The annual precipitation in this area is 849 mm (reference period 1981–2010, German Weather Service) and the mean annual air temperature is 9.7 °C (reference period 1989–2012, German Weather Service).

After decades of intensive agriculture with deep drainage, the peatlands in this area are strongly degraded with a surface subsidence of up to ~1 m since 1958 (Krebs *et al*. 2012). For field preparation, an area of ~ 4 ha of intensively used fertilised bog grassland was cleared of the uppermost and highly mineralised peat (~ 30 cm depth) in April 2011. The removed peat was used to form dams along the sides of the field which were used as causeways. The *Sphagnum* production area was subdivided into 10 m wide production strips by channels of the irrigation system (Figure 1). Two production systems of Sphagnum farming that represent different field management strategies are being tested in the field (each on 1.5 ha). In System 1 the production area is managed using specialised site-adapted machinery driving on the cultivation strips themselves (e.g. during harvesting), while in System 2 the management is conducted exclusively from the dams around the periphery (Wichmann *et al*. 2014, 2017). As a consequence, the production systems differ in their abundance of dams (Figure 1, Table 1).

For inoculation of the production strips, *Sphagnum* mosses were gathered from a nearby experimental field and a natural site. On 10 May 2011, fragments (~ 1 cm length) of *Sphagnum papillosum* Lindb. (0.5 ha in total) and *Sphagnum palustre* L. (1.1 ha in total) were spread evenly on the plane surfaces of the production strips and subsequently covered with straw mulch (Krebs *et al*. 2012, Wichmann *et al*. 2017). These two species were chosen on the basis of promising results in preliminary experiments regarding production and cultivation (Gaudig 2008, Gaudig *et al*. 2012) and their availability. Eventually, the field was re-wetted and, since then, the water table has been maintained

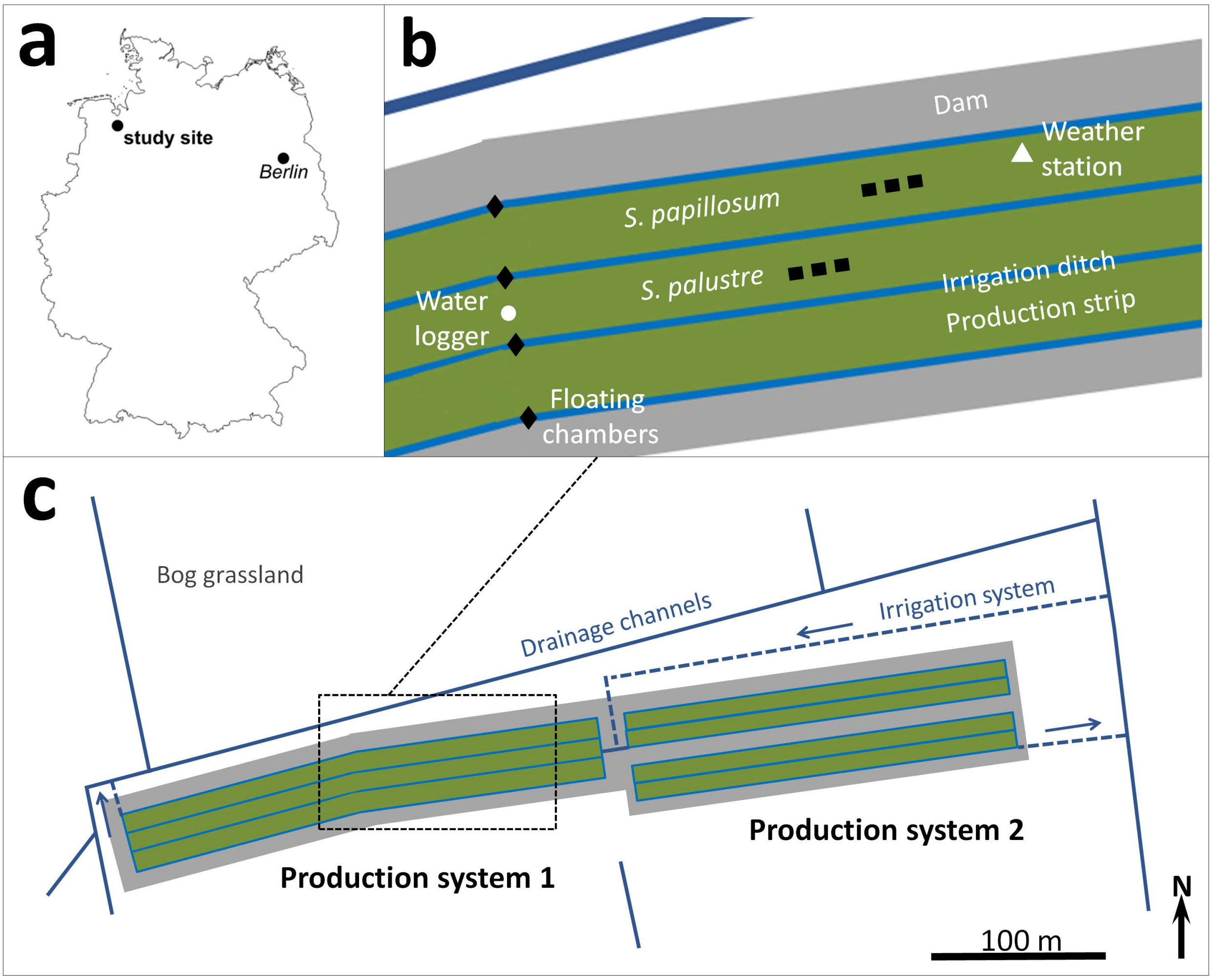

Figure 1 The experimental Sphagnum farming field: (a) location in Germany in the municipality Rastede; (b) position of the weather station (white triangle), water logger (white circle), sampling plots (black squares) and floating chambers (black diamonds); (c) composition and arrangement of the two production systems.

Table 1. Composition of the two production systems in % area of the different structural components (production area, ditches, and dams) for the realised experimental field and planned commercial Sphagnum farming fields. Production System 1: Management with site-adapted machinery directly on the *Sphagnum* production area, Production System 2: Management exclusively from the dams.

	Production system	Production area	Irrigation Ditches	Dams
Rastede experimental field (this study)	1	55 %	5 %	40 %
	2	51 %	5 %	44 %
Planned *Sphagnum* production fields	1	80 %	5 %	15 %
	2	50 %	5 %	45 %

at around ground surface level. The field was not fertilised and the *Sphagnum* biomass has not yet been harvested.

Study set-up

All GHG measurements were conducted in Production System 1 (Figure 1). In June 2011, boardwalks were constructed in that system to minimise disturbance during measurements of GHG exchange. Three measurement plots were established in the production strip of each *Sphagnum* species (*S. palustre*, PA, and *S. papillosum*, PI; six plots in total), with adjacent plots ~ 1 m apart. At each plot a square collar (about 0.75 m × 0.75 m × 0.15 m) was inserted approximately 10 cm into the peat and remained in place during the whole investigation period. Additional sampling plots for floating chambers were established in each of the irrigation ditches (n = 4, Figure 1) by inserting several bamboo sticks into the sediment to mark the plots permanently and to keep the chambers in place during measurements.

During the period from September 2011 to the end of August 2013, we conducted measurements of GHG exchange using closed chambers (Livingston & Hutchinson 1995). In the following, we refer to the time period from 01 September 2011 to 31 August 2012 as ‘Year 1’ and to the period from 01 September 2012 to 31 August 2013 as ‘Year 2’.

Greenhouse gas (GHG) measurements

Exchange of CH_4/N_2O (and CO_2 from the ditches) was determined at fortnightly intervals over the whole two years of the study period by laboratory gas chromatographic analyses of samples from the chamber headspace. In addition, we made monthly field measurements of CO_2 exchange from the *Sphagnum* production strips using a portable infrared gas analyser (IRGA).

For the measurements we used two sets of chambers which differed in shape. The chambers used on the production strips were box-shaped and had an enclosed volume of 0.3042 m^3 (height = 0.5 m). During the measurements, chambers and collars overlapped by ~ 5 cm. This, combined with soft rubber rims glued to the bottoms of the chambers, ensured air-tightness during chamber closure. In the irrigation ditches we used opaque cylindrical chambers (V = 0.0073 m^3, height = 0.2 m) with styrofoam rims that enabled them to float on the water.

For fortnightly measurements of CH_4 and N_2O exchange the chambers on the production strips consisted of white polyvinyl chloride (PVC) and were equipped with venting tubes (diameter 12 mm, length = 0.5 m). During measurements, the chambers were closed for 60 min and four gas samples were taken from the headspace at equal time intervals. Given the large chamber volume, this long closure time was necessary to get measurements of sufficient accuracy. During the closure time, headspace temperature increased, on average, by only 0.4 °C. Due to higher fluxes and the smaller headspace volume of the floating chambers, closure time was reduced to 15 min for measurements on the irrigation ditches. All samples were taken using pre-evacuated glass vials (V = 60 ml) and were analysed within one week for CH_4 and N_2O concentrations using a Gas Chromatograph (GC, Shimadzu, Japan) with a Flame Ionisation Detector (FID) and an Electron Capture Detector (ECD). Samples from the floating chambers were additionally analysed for their CO_2 concentrations using the GC.

At approximately monthly intervals we estimated net exchange of CO_2 (*NEE*) on the *Sphagnum* production strips as the net difference between photosynthesis (gross primary production, *GPP*) and ecosystem respiration (R_{eco}) according to the equation:

$$NEE = GPP + R_{eco} \qquad [1]$$

Chambers for measurements of R_{eco} were identical to those for CH_4/N_2O measurements in terms of size and equipment but additionally contained two fans (flow rate = 160 dm^3 min^{-1}). For measurements of *NEE*, similar but transparent chambers consisting of polymethyl methacrylate (PMMA, “acrylic glass”) were used. On each monthly measurement day, we determined R_{eco} at least once and *NEE* multiple times over a range of natural light conditions starting at sunrise. In this way, we collected on average two valid R_{eco} fluxes and four valid NEE fluxes for each plot and measurement day. The headspace temperature of the transparent chambers was reduced by installing cooling packs in the chambers if required. Through this measure, headspace temperature change during chamber closure averaged 0.6 ± 1.1 °C. The durations of chamber measurements of *NEE* and R_{eco} ranged from 120 to 300 s. During CO_2 measurements, headspace CO_2 concentrations (in ppmv) were monitored using a portable IRGA (Licor-820, Licor Biosciences, USA). At the same time, photosynthetic photon flux density (400–700 nm; PPFD) was monitored with a quantum sensor (Indium Sensor, Germany) and temperatures were recorded inside and outside the chamber, as well as at 2, 5 and 10 cm beneath the peat surface (sensors from F&C GmbH, Germany). All sensor readings were recorded at approximately 1.5 s intervals.

Environmental variables

A weather station (F&C GmbH, Germany) was installed a few metres from the measurement plots. It was equipped with an automated logger to record half-hourly means of soil temperature (at 2, 5 and 10 cm depth), air temperature and humidity, wind speed and direction, as well as PPFD. Owing to technical problems, especially at the beginning of the study period, no environmental variables were recorded over a total of 101 days (≈ 14 % of the study period). These gaps were filled by obtaining data either from mobile loggers that were active at the study site during measurement campaigns or from a weather station installed at ~ 60 km distance. All devices used for gap-filling were equipped with sensors similar to those connected to the weather station at the study site. In addition, precipitation data were obtained from a weather station (53° 14.52′ N, 8° 13.44′ E; operated by the German Weather Service, station name "Rastede") at ~ 4 km distance from our experiment field. Water levels were recorded at 10 min intervals (01 September 2011 to 24 February 2012) and hourly intervals (25 February 2012 to 31 August 2013) by an automated logger (Type 575, HT Hydrotechnik GmbH, Germany) installed at the centre of the production area. Vegetation development was monitored by visually estimating plant species cover-abundance (Braun-Blanquet 1951) once *per* year.

General data analysis

All calculations were performed in R 3.1.2 (R Core Team 2014). Mean values are given as ± SE. We follow the atmospheric sign convention, where positive values represent fluxes from the soil to the atmosphere. For estimating gas fluxes we used the functions *flux* and *fluxx* of the R package 'flux' (Jurasinski *et al*. 2014) as described by Günther *et al.* (2015).

Estimation of CH_4 and N_2O fluxes and annual balances

Flux estimation for CH_4 and N_2O (and CO_2 in the floating chambers) was based on at least three out of four concentration measurements. To eliminate the influence of extreme values on annual CH_4 balances, a distribution-based outlier test was performed prior to further analyses of CH_4 fluxes for both the production strips and the ditches. Based on the most appropriate distribution (lognormal), CH_4 fluxes which exceeded the most extreme 1.25 % at either end of the distribution (2.5 % total) were removed from the dataset. This resulted in deletion of 57 fluxes from the ditches and nine fluxes from the two production strips (total number of fluxes before deletion: 192 from ditches, 303 from production strips). The deleted fluxes from the ditches mainly resulted from measurements with extremely high headspace concentrations of CH_4 (maximum CH_4 concentrations > 15,000 ppbv), indicating ebullitive fluxes. We excluded these values to avoid errors when integrating over longer time periods.

Since our data did not allow for statistical modelling using environmental parameters, we used a combination of bootstrap and jackknife methods for the estimation of annual balances as described by Günther *et al.* (2015). In short, we randomly picked one of the available fluxes from the three spatial replicates for each measurement day (bootstrap method). After repeating this procedure 100 times, we estimated 100 CH_4 balances for the datasets, each time omitting one measurement day (jackknife method, 10,000 balance values in total). We then calculated the jackknife error $SE_{JK}(T_n)$ for each of the 100 bootstrap datasets (Köhler *et al*. 2007). The reported annual balances represent the means of all jackknife estimates from the 100 bootstrap datasets, and the errors of the annual balances represent the means of all 100 jackknife errors. This approach allows us to report robust annual estimates, including estimates of the uncertainty associated with linear interpolation.

Because the CH_4 flux data were not normally distributed we used Mann-Whitney tests for species / season comparisons. For calculation of the net climatic effect for the field, CH_4 and N_2O emissions were converted to CO_2-equivalents using their respective global warming potentials (GWP) of 28 for CH_4 and 265 for N_2O. These values are for a 100-year timespan (Myhre *et al.* 2013, excluding climate-carbon feedbacks because they are highly uncertain). The errors in the net GHG balances ($Err_{Balance}$) were obtained by integrating the individual errors (in CO_2-equivalents) according to the following equation:

$$Err_{Balance} = \sqrt{(Err_{CO_2})^2 + (Err_{CH_4})^2 + (Err_{N_2O})^2} \quad [2]$$

where Err_{CO2} is the error of the CO_2 balance, Err_{CH4} is the error of the CH_4 balance and Err_{N2O} is the error of the N_2O balance.

Modelling of CO_2 exchange

R_{eco} was estimated separately for both *Sphagnum* species using all CO_2 fluxes determined with the opaque chamber as well as those determined with the transparent chamber when PPFD was lower than 50 µmol m^{-2} s^{-1}. We modelled R_{eco} (g m^{-2} h^{-1}) for each species over the whole study period by optimising the parameters R_{ref} and $E0$ of Equation 3:

$$R_{eco} = R_{ref} \times e^{E0\left[\frac{1}{T_{ref}-T_0} - \frac{1}{T_S - T_0}\right]} \times ETI \qquad [3]$$

where R_{ref} is CO_2 release (g m^{-2} h^{-1}) at the reference temperature T_{ref} (283.5 K), $E0$ is the activation energy (K), T_0 is the temperature constant for the start of biological processes (237.48 K), T_S is the soil temperature (5 cm depth) during the measurements, and ETI is the effective temperature (>5 °C) sum index. The latter was included in the response function as a technical term to indicate vegetation development over the year (Alm *et al.* 1999). However, unlike Alm *et al.* (1999) we defined ETI as the slope of the curve of the effective temperature sum (threshold 5 °C) of each year because it better represented the seasonal variation in our all-season study.

We calculated GPP fluxes from the estimated R_{eco} and NEE fluxes according to Equation 1. For each *Sphagnum* species, we fitted a non-linear model to the GPP fluxes of the main growing season (MGS) of each measurement year. Due to an unusually cold spring in both measurement years, we assumed the MGS to range from May to October. We based this assumption on the occurrence of minimum air temperatures close to 0 °C, which has been shown to represent a threshold for the growth of many *Sphagnum* species (Gerdol 1996, Asada *et al.* 2003). Also, the times outside the MGS corresponded to times when the accumulation of the effective temperature sum ceased, which is an indicator for stagnating growth in *Sphagnum fuscum* (Schimp.) H. Klinggr. (Lindholm 1990). The general form of the functions to estimate GPP fluxes (g m^{-2} h^{-1}) between measurements followed Equation 4:

$$GPP = \frac{GP_{max} \times \alpha \times PAR}{\alpha \times PAR + GP_{max}} + b_0 + b_1\, ETI + b_2\, T_S + b_3\, T_A \qquad [4]$$

where α is the initial slope of regression, GP_{max} (g m^{-2} s^{-1}) is the boundary value of GPP at infinitely high PPFD, and T_S and T_A represent soil and air temperature. T_S was obtained by averaging the measured soil temperatures at 2, 5, and 10 cm depth. T_A represents the mean temperature of chamber headspace and surrounding air.

Coefficients of the environmental variables were determined by non-linear least-squares parameter estimation. Since no satisfying model could be fitted outside the MGS, we applied a simple linear regression to these GPP fluxes of each *Sphagnum* species following Equation 5:

$$GPP = b_0\, PAR + b_1\, T_S + b_2\, ETI \qquad [5]$$

We predicted half-hourly GPP and R_{eco} by applying the above models to the weather station data. Modelled GPP fluxes which were larger than 0 (therefore, indicating negative photosynthesis) were set to equal 0. Also, GPP fluxes were defined to be 0 when there was no irradiation (PPFD = 0). The half-hourly estimates were then linearly integrated over time to obtain estimates for each study year.

We determined modelling errors (E_{mod}) by evaluating the difference between observed (NEE_{obs}) and modelled NEE values (NEE_{mod}) for each crop and year following Equation 6 (Aurela *et al.* 2002, Adkinson *et al.* 2011):

$$E_{mod} = \sqrt{\frac{(NEE_{obs} - NEE_{mod})^2}{(n-1)\cdot n}} \qquad [6]$$

We then assigned E_{mod} to each modelled, half-hourly NEE value. By linear integration we obtained annual NEE errors.

As the headspace of the floating chambers used on the ditches did not contain any plant biomass, we assumed that the only process driving CO_2 exchange was R_{eco}. Therefore, we did not apply transparent chambers on ditches. R_{eco} fluxes were determined from the same gas samples that were used for CH_4/N_2O flux estimation and using the same approach. Annual CO_2 balances of the ditches were determined using the bootstrap/jackknife methods described above for CH_4/N_2O fluxes.

RESULTS

Environmental conditions and vegetation development

Year 1 was slightly wetter and Year 2 drier than the long-term mean (863 mm and 696 mm compared to 849 mm long-term mean). However, this was not reflected in differences regarding water level, since it was artificially maintained at a constant level (Year 1: 4.4 ± 7.2 cm, year 2: 5.2 ± 5.4 cm below the peat surface). In Year 1 the mean annual air temperature resembled the long-term mean; while Year 2 was cooler than the average (9.7 °C and 8.3 °C compared to 9.7 °C long-term mean, see Figure 2).

In October 2011, the measurement plots still showed patches of bare peat covering ~ 48 % of the surface of *S. palustre* (PA) plots and ~ 31 % of the surface of *S. papillosum* (PI) plots. At this time, *S. palustre* lawn covered on average 46 % of the area of the PA plots, while *S. papillosum* lawn had established less quickly with only 32 % mean

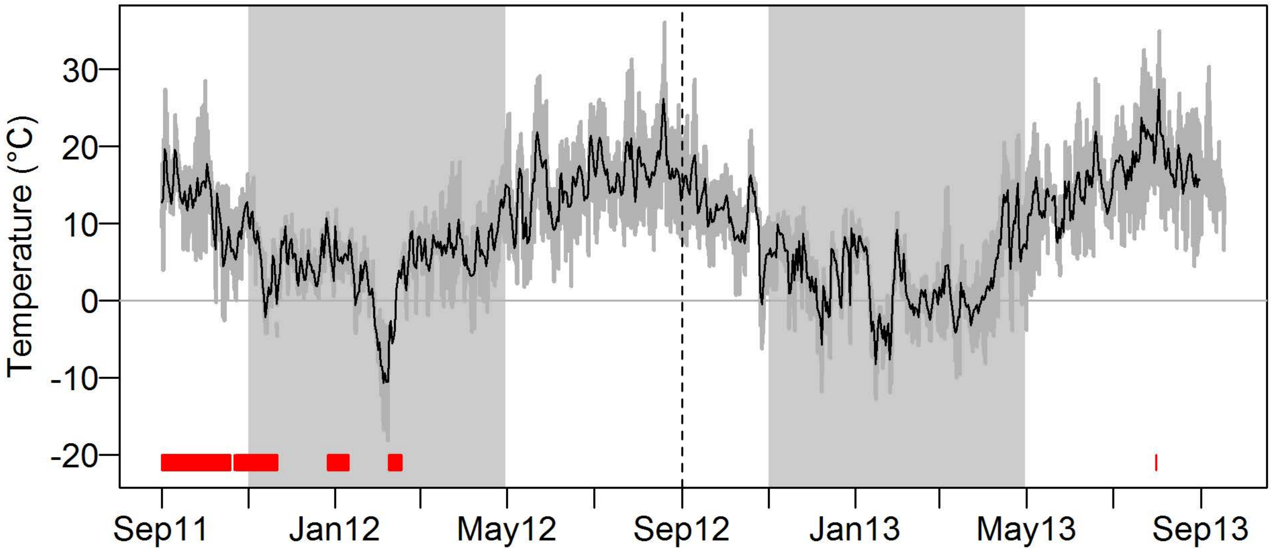

Figure 2. Air temperatures at 2 m height over the investigation period. Half-hourly averages (grey bars) are shown together with the daily mean (black line). The main growing seasons (May to October) are indicated by a grey background. The red bars at the bottom of the figure indicate times where, due to technical failure of the weather station at the study site, temperature data were obtained from other sources (see section 'Environmental variables' in the text).

coverage on the PI plots. By August 2012, all plots were completely vegetated and the two moss species covered on average 79 % (PA) and 71 % (PI) of their respective measurement plots which still did not represent a completely established peat moss lawn (mean cover > 90 %; cf. Gaudig *et al.* 2014a). Apart from the two target species, a number of vascular wetland plants such as *Juncus effusus* L. and *Drosera rotundifolia* L. as well as additional *Sphagnum* species (*Sphagnum angustifolium* (Warnst.) C. E. O. Jensen, *Sphagnum cuspidatum* Ehrh. ex Hoffm, *Sphagnum fallax* (H. Klinggr.) H. Klinggr.) established on the measurement plots. After initial fast establishment, the cover of *J. effusus* steadily decreased over the study period (from ~ 40 % cover in 2011 to ~ 10 % cover in 2013) as an effect of active management (mowing) and increasing moss establishment.

Greenhouse-gas exchange of production strips and ditches

The measurement plots on *S. palustre* displayed significant seasonal patterns of CH_4 fluxes in both study years ($P < 0.05$, tested with Mann-Whitney test), with higher fluxes during the vegetation period than during the rest of the year (Figure 3). In contrast, CH_4 fluxes of *S. papillosum* plots were more variable and did not show a significant seasonal pattern. CH_4 fluxes of both *Sphagnum* species decreased significantly over the two study years (PA: $P = 0.02$, PI: $P = 0.01$; tested with linear regressions of the log-transformed fluxes). This trend was stronger for *S. papillosum*, which is reflected in the strong reduction of the estimated annual CH_4 balances that are more than halved in Year 2 compared to Year 1 (Table 2). Overall, CH_4 fluxes of the two *Sphagnum* species were significantly different during the study period ($P < 0.001$, tested with Mann-Whitney test) with higher mean fluxes occurring on *S. papillosum* (PA: 0.1 ± 0.2 mg m^{-2} h^{-1}; PI: 0.2 ± 0.3 mg m^{-2} h^{-1}).

CO_2 fluxes of both *Sphagnum* species show typical seasonal variation with higher average R_{eco} fluxes during the MGS (PA: 0.39 ± 0.03 g m^{-2} h^{-1}, PI: 0.42 ± 0.03 g m^{-2} h^{-1}) than outside the MGS (PA: 0.08 ± 0.01 g m^{-2} h^{-1}, PI: 0.06 ± 0.01 g m^{-2} h^{-1}; Figure 4). Similarly, the average NEE fluxes indicate higher CO_2 uptake during the MGS compared to the rest of the study year for both *S. palustre* (during MGS: -0.64 ± 0.04 g m^{-2} h^{-1}, outside MGS: -0.16 ± 0.02 g m^{-2} h^{-1}) and *S. papillosum* (during MGS: -0.64 ± 0.04 g m^{-2} h^{-1}, outside MGS: -0.30 ± 0.03 g m^{-2} h^{-1}). Over the study period, neither NEE nor R_{eco} fluxes were significantly different between the two *Sphagnum* species (*NEE*: $P = 0.4$, R_{eco}: $P = 0.5$, Mann-Whitney tests). CO_2 fluxes do not follow a significant temporal trend over the two years (PA: $P = 0.75$, PI: $P = 0.23$; tested with linear regressions). The models for obtaining half-hourly *NEE* values show good correspondence with measured *NEE* fluxes (Figure 5).

Fluxes of both CO_2 and CH_4 were more erratic from the irrigation ditches compared to those from the production strips and did not follow seasonal trends (Figure 6). Also, neither CH_4 nor CO_2 fluxes

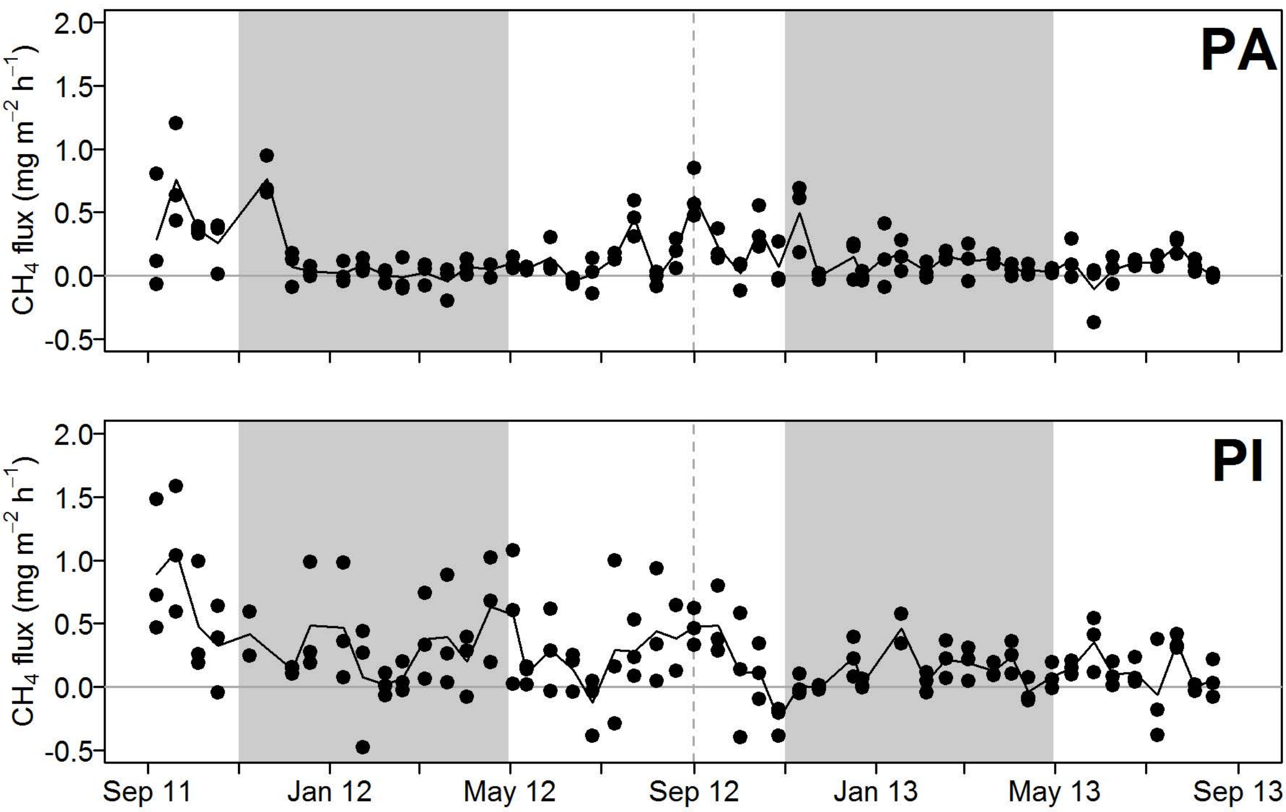

Figure 3. Methane fluxes (mg m^{-2} h^{-1}) of the two production strips with *S. palustre* (PA) and *S. papillosum* (PI) during the establishment phase. Individual flux values (solid circles) are shown together with the mean of each measurement day (black line). The main growing seasons (May to October) are indicated by a grey background.

Table 2. Estimated annual balances of CO_2, CH_4 and N_2O together with combined climatic effect (all in g m^{-2}) for the production strips and irrigation ditches. Values are given ± SE. Note that the annual balances of the GHGs refer to the released mass of gas, while the combined balances of all gases have been converted to CO_2-equivalents.

		CO_2	CH_4	N_2O	Sum GHGs (CO_2eq)
Year 1	*S. palustre*	-629 ± 188	1.4 ± 0.5	0.0 ±0.3	-578 ±209
	S. papillosum	-898 ± 196	2.7 ± 0.7	0.1 ± 0.2	-790 ± 221
	Ditches	608 ± 393	14.4 ± 6.2	0.3 ± 0.4	1101 ± 577
Year 2	*S. palustre*	-547 ± 92	1.0 ± 0.4	0.0 ± 0.1	-506 ± 98
	S. papillosum	-875 ± 100	1.2 ± 0.5	-0.1 ± 0.1	-857 ± 108
	Ditches	910 ± 604	4.8 ± 4.9	0.6 ± 0.4	1135 ± 631

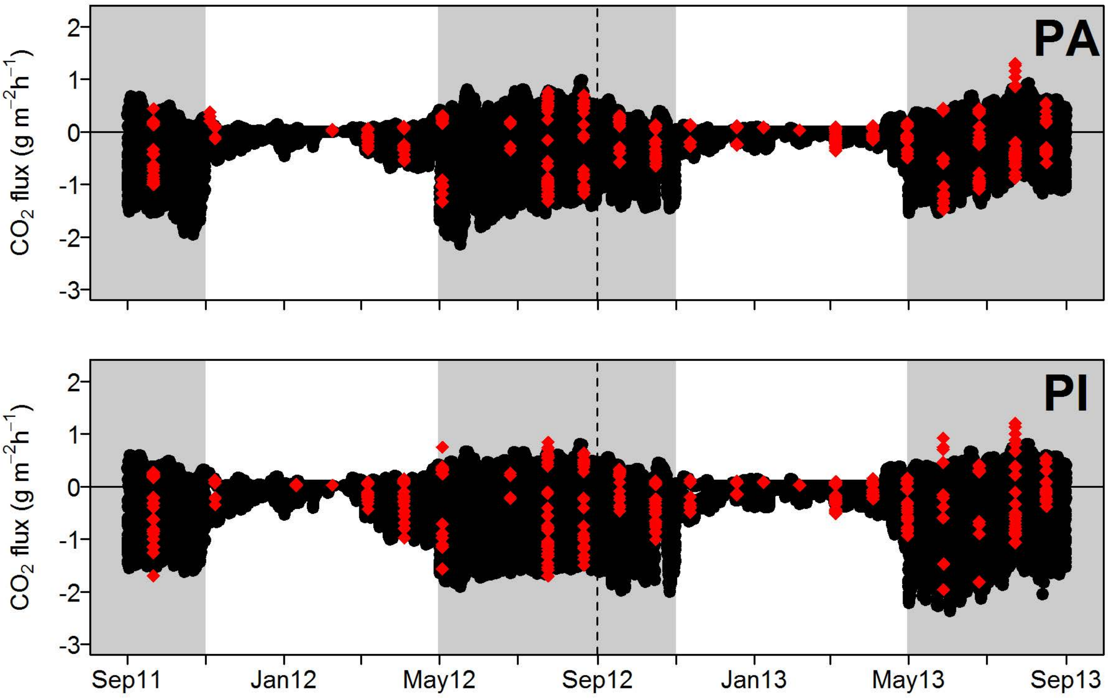

Figure 4. NEE fluxes (g m^{-2} h^{-1}) of the two production strips with *S. palustre* (PA) and *S. papillosum* (PI) during the establishment phase. Modelled (black circles) and measured (red diamonds) values are given. The main growing seasons (May to October) are indicated by a grey background.

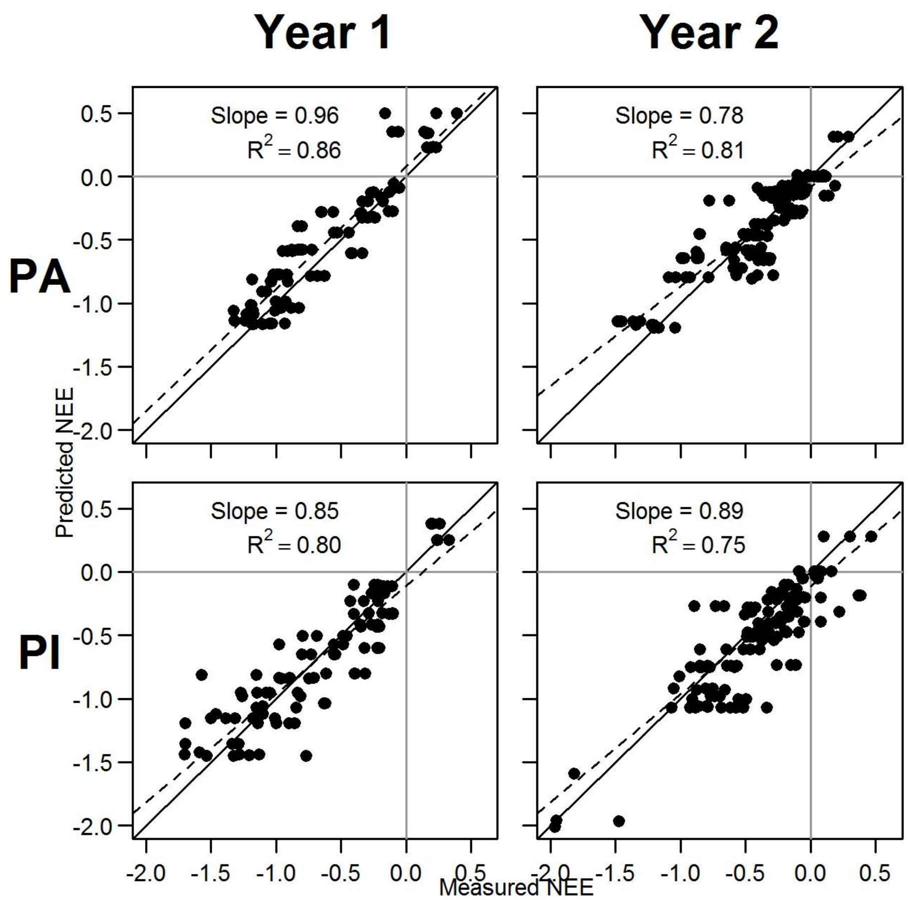

Figure 5. Modelled against measured NEE fluxes (g m^{-2} h^{-1}) for *S. palustre* (PA) and *S. papillosum* (PI) together with a linear regression (dashed line) and 1:1 line (solid line). "Year 1" corresponds to the period from September 2011 to August 2012 and "Year 2" to the period from September 2012 to August 2013. The slope and R^2 of the linear regression fit are also given.

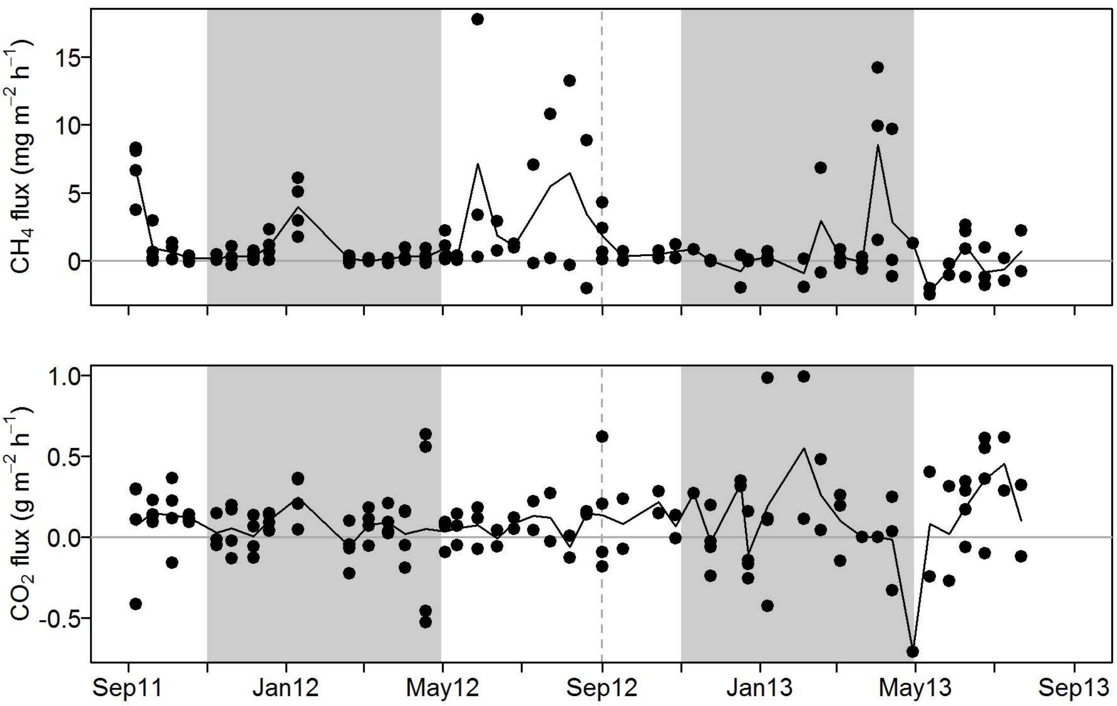

Figure 6. Methane fluxes (mg m^{-2} h^{-1}) and carbon dioxide fluxes (g m^{-2} h^{-1}) with the irrigation ditches. (Note the different mass units.) Individual flux values (solid circles) are shown together with the mean of each measurement day (black line). The main growing seasons (May to October) are indicated by a grey background.

show significant temporal trends over the study period (CH_4: P = 0.09, tested with linear regression of the log-transformed fluxes; CO_2: P = 0.23, tested with linear regression). Compared to the production strips, the ditches were a CO_2 source instead of a CO_2 sink and emitted larger amounts of CH_4 (Table 2).

N_2O exchange of both the production strips and the irrigation ditches remained very low during the study period and did not follow any temporal trends. Nonetheless, we included the N_2O balances in the overall GHG balances to avoid discrimination against small fluxes (Table 2).

The production strips of both *Sphagnum* species were net annual GHG sinks during the establishment phase, while the ditches were slight GHG sources (Table 2). The small difference in the GHG sink between the two mosses is within the uncertainty of the annual balances.

DISCUSSION

CH_4 emissions during the establishment phase

We found no evidence for the occurrence of a strong CH_4 peak following field installation. However, there seem to have been some establishment effects. CH_4 fluxes of both *S. papillosum* and *S. palustre* showed a small decrease over the two years. Higher CH_4 effluxes at the beginning of the study period could have resulted from a higher availability of labile organic compounds. Despite the removal of the upper degraded peat layer down to the pristine peat, the earthworks during the installation of the field may have led to some erosion-induced carbon loss. Another explanation could be that, due to the decreasing cover of the aerenchymatous species *J. effusus*, CH_4 was transported less effectively to the atmosphere towards the end of the study period. Since aerenchymatous species are known to be strong drivers of CH_4 transport (Chanton & Whiting 1995, Thomas *et al.* 1996, Askaer *et al.* 2011), the cover of such ‘shunt’ species should be minimised on Sphagnum farming fields. This is also advisable for reasons of maximising *Sphagnum* moss production.

The trend of decreasing CH_4 fluxes over time was stronger in *S. papillosum* than in *S. palustre*. This might be explained by physiological differences, or by spatial differences between the production strips since the measurement design was not randomised (Figure 1b). We found that *S. papillosum* established more slowly than *S. palustre*. This is in accordance with Krebs *et al.* (2012) who found less growth by *S. papillosum* than by *S. palustre* during the first six months after moss spreading. Nonetheless *S. papillosum* showed slightly higher rates of CO_2 uptake than *S. palustre* in our study, indicating a faster carbon turnover. Therefore, it is possible that more fresh organic compounds may have been available due to the higher metabolic activity at the *S. papillosum* strip during the early stages of establishment. Since the peat was constantly water saturated during this time (Krebs *et al*. 2012), the available substrate would plausibly have been transformed to CH_4 rather than to CO_2. Accordingly, CO_2 exchange over the study period did not follow a significant temporal trend. Beside species differences, another explanation for the different development of CH_4 emissions could be slight differences in peat quality, substrate availability or hydrology between the centre (the *S. palustre* strip) and the border (the *S. papillosum* strip) of the field. Since the position of measurement plots could not be randomised (for technical reasons) we cannot finally attribute the differences in CH_4 emissions to either species differences or spatial effects.

Total GHG emissions of production area, ditches and dams

The production area of both *Sphagnum* species was already a net GHG sink in the first year after field installation. In contrast, the irrigation ditches were GHG sources in both study years. Previous studies also found high GHG emissions by open water bodies (such as ditches and lakes) in peatlands (Minkkinen & Laine 2006, Wilson *et al.* 2009, Schrier-Uijl *et al.* 2011, Hiraishi *et al.* 2014). While the GHG emissions of the irrigation ditches in our study were also raised compared to those of the production strips, they were lower than the values in the literature. However, the magnitude of GHG emissions from ditches seems to be related to vegetation, with fewer emissions for ditches with *Sphagnum* cover (Minkkinen & Laine 2006, Cooper *et al.* 2014). In addition, GHG emissions from ditches are often released as bubbles. The question of how to account for these erratic emissions during the estimation of annual GHG balances is still not settled. Here, we excluded extreme fluxes to avoid errors by assigning ebullitive fluxes to time spans of several weeks. It is possible that this approach led to the slightly lower emissions from the ditches compared to the literature.

In any case, it is not clear whether the emissions from the ditches truly represent GHG emissions of the Sphagnum farming field, because the water used for irrigation originates mainly as drainage from the surrounding intensively-used peatlands. It is known that intensive drainage of peat soils can lead to high GHG emissions downstream, e.g. due to the enhanced release of DOC (Holden 2005, Hiraishi *et al*. 2014, Luan & Wu 2015). Therefore, it is possible that the emissions from the ditches are driven by the intensive management practices of the surrounding fields rather than by the management of the *Sphagnum* farming field itself. Unfortunately, we did not measure DOC at the water inlet of the field; therefore, we do not know whether the inflow of carbon determined the high CO_2 emissions of the ditches. Suitable measurements should be made in further studies, because the information is important for generating a complete climatic balance of *Sphagnum* farming products. In any case, it seems advisable to minimise the area occupied by ditches to improve the net climatic effect of the *Sphagnum* farming system. This might be achieved by reducing the size of the ditches if possible and by evaluating the hydraulic conductivity of the peat in order to maximise the distances between ditches.

During installation of the *Sphagnum* farming field, the uppermost layer of degraded peat was removed and used to construct dams surrounding the production strips. This peat will eventually be converted mainly to CO_2 because it is well aerated. Therefore, the emissions originating from the peat of the dams must be added to the emissions of the production strips and ditches to evaluate the GHG effect of a whole *Sphagnum* farming field. If we assume CO_2 effluxes similar to those of agriculturally-used but unfertilised, drained bogs in Germany (12.6 t ha^{-1} a^{-1}, taken from Drösler 2005) for the area covered by the dams, the two production systems in their present configuration at the experimental field probably represented a slight GHG source of 2.5 t ha^{-1} a^{-1} CO_2-equivalents during the establishment phase. However, the composition of the experimental field is different from that of future commercial fields regarding the relative proportions of ditches, dams and production strips (see Table 1). Following the configuration of planned *Sphagnum* farming fields with Production System 1, the area would have been a net GHG sink of -1.9 t ha^{-1} a^{-1} and -4.1 t ha^{-1} a^{-1} during the establishment phase if cultivated with *S. palustre* or *S. papillosum*, respectively. Under Production System 2, which has a higher relative abundance of dams, a future Sphagnum farming field would emit 3.5 t ha^{-1} a^{-1} (*S. palustre*) and 4.1 t ha^{-1} a^{-1} (*S. papillosum*, both in CO_2-equivalents) to the atmosphere. However, for final conclusions about the net effects of a complete Sphagnum farming field, future studies should include direct measurements on the dams surrounding the field, as well as measurements of fluvial imports to and exports from the Sphagnum farming field. Also, GHG measurements should be repeated on fields with the final structural composition and dimensions of commercial Sphagnum farming.

Later phases of the production cycle

So far, there are no existing data on the GHG exchange of the study site or similar Sphagnum farming fields during later phases of the production cycle. The only available study on the GHG exchange of Sphagnum farming in Germany during the production phase (Beyer & Höper 2015) originates from a field with very different properties compared to our study site. In that study, among other things, the *Sphagnum* mosses were cultivated directly on strongly humified peat (unlike our study where the strongly degraded peat layer was removed) and the water table was lower and more variable than in our study. Beyer & Höper (2015) found a lower sink strength of the production area during the production phase (-270 ± 79 g m^{-2} a^{-1} in CO_2-equivalents) due to a lower net CO_2 uptake compared with our study. The discrepancy is most likely to result from higher soil respiration due to the lower water table (6–9 cm below the peat surface) and, thus, faster carbon turnover. This explanation is supported by the fact that both annual *GPP* and annual R_{eco} are, on average, numerically larger in the study of Beyer & Höper (2015) (*GPP* -2020 g m^{-2} a^{-1}, R_{eco} 1657 g m^{-2} a^{-1}) than in our study (-1616 g m^{-2} a^{-1} and 984 g m^{-2} a^{-1}).

To evaluate whether Sphagnum farming fields are net sinks or sources of carbon over a complete production cycle the harvested biomass must be included in the balances. Up to the present, the *Sphagnum* biomass at the experimental field has not been finally harvested. It is likely that the net GHG footprint of Sphagnum farming would be that of a small GHG source when including the harvested biomass, because most of the CO_2 assimilated by the mosses is ultimately released to the atmosphere when the biomass is used as a horticultural substrate. For this reason, an evaluation of the net climatic effect of a complete production cycle of Sphagnum farming should be the subject of future studies, when the biomass yields of the experimental field are known. Furthermore, no studies have yet accompanied the harvest or other management events with measurements of GHG exchange. Also, the climatic effect of the regeneration phase following the harvest is unknown. More work is needed to evaluate the climatic effect of a whole production cycle of Sphagnum farming.

Sphagnum farming as land use alternative to conventional agriculture on bogs

Due to the deep drainage of the peat (> 50 cm below ground surface) that is typical for conventional land use, temperate bogs typically emit large quantities of GHGs under intensive grassland use or crop production, mainly as CO_2 and N_2O (van Beek *et al*. 2010, Elsgaard *et al*. 2012, Beetz *et al*. 2013). Including N_2O emissions, the IPCC Wetlands Supplement (Hiraishi *et al*. 2014) lists a net emission factor of 21.5 t ha^{-1} a^{-1} (in CO_2-equivalents) for temperate drained bogs under grassland use. Note that this does not include emissions by drainage ditches. Accordingly, total GHG emissions (including N_2O) of 15–35 t ha^{-1} a^{-1} (in CO_2-equivalents) are reported for temperate peatlands that are under intensive cropland or grassland use with deep drainage in Germany and The Netherlands (Beyer 2014, Schrier-Uijl *et al*. 2014). In contrast, reported average values of net GHG release of peatlands under low-intensity grassland use are in the range of only 6–14 t ha^{-1} a^{-1} (in CO_2-equivalents, Beyer 2014, Schrier-Uijl *et al*. 2014, Beyer *et al*. 2015).

Emission data for the establishment phase (this study) and growing phase (Beyer & Höper 2015) indicate that Sphagnum farming, when applied on bogs, might have even lower on-field GHG emissions than low-intensity agriculture. However, compared to low-intensity utilisation of peatlands, Sphagnum farming has the additional advantage that the generated *Sphagnum* biomass helps to reduce the mineralisation of pristine peat and, therefore, the emission of further GHGs, by providing an alternative growing substrate for horticulture. In contrast, assuming emissions of 15 t ha^{-1} a^{-1} as a conservative estimate, the combined 3 ha of the two production systems in our study would have released ~ 45 t a^{-1} of CO_2-equivalents under intensive grassland use (the former land use). Compared to this scenario, the experimental Sphagnum farming field had already avoided GHG emissions of up to 32 t a^{-1} of CO_2-equivalents during the establishment and growing phases, depending on the magnitude of emissions resulting from biomass export.

We did not observe a CH_4 emission peak during the first two years after installation of the experimental field. However, the irrigation ditches were a GHG source mainly through emissions of CO_2 from the water body. The production strips themselves were GHG sinks regardless of the cultivated *Sphagnum* species. If we assume literature values for emissions by the dams, a complete Sphagnum farming field represents a slight sink or a slight source of GHG, depending on its composition regarding irrigation ditches, dams and production area. In this way, Sphagnum farming mitigates several tons of GHG emissions (in CO_2-equivalents) each year compared to the practice of intensive agriculture on bogs.

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