Managing wet fen meadows for nature conservation leads to a moderate warming effect

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SUMMARY

Wet fen meadows under traditional land use are nowadays mainly preserved through nature conservation measures. Recent discussions suggest that this land use may also be regarded as a form of paludiculture - that is, use of wet peatland with preservation of the peat body. However, the climate effect of this land use type is largely unknown. This study presents full two-year greenhouse gas (GHG) balances for two previously unexplored, long-term rewetted fens under a nature conservation management regime. Closed-chamber GHG fluxes were measured biweekly at two north-east German sites with acute sedge (*Carex acuta*) and at one site with creeping bentgrass (*Agrostis stolonifera*). Including harvest and dissolved carbon export, the CO_2-eq emissions of the three sites were between 10.4 and 16.3 t ha^{-1} yr^{-1}, with mean annual water levels between -10 and -19 cm relative to ground level. Emissions consisted mainly of CO_2 uptake and release and were influenced by the timing and frequency of harvests as well as by periods of surface flooding during the growing season. CH_4 emissions also contributed to the net GHG balances at two of the sites due to inundation in late summer 2014. N_2O emissions were of minor importance at all three sites. This study shows that, with proper water management, the climate effect of fen meadows under management for nature conservation can be similar to that of other fen paludicultures, with a CO_2-eq GHG mitigation potential of 15–20 t ha^{-1} yr^{-1} compared to drainage-based grassland use of fens. Therefore, for GHG reporting purposes, it may be justifiable to treat existing wet fen meadows under nature conservation management as paludicultures.

KEY WORDS: *Agrostis*, *Carex*, climate change mitigation, peat grassland, precipitation, wet land use

INTRODUCTION

Anthropogenic climate change is considered to be the greatest global threat and challenge facing humanity (WEF 2020, Robert Koch Institute 2023). To mitigate the worst effects of climate change such as more intense storms, increased droughts and rising sea levels, it is imperative to limit the increase in global average temperature to below 2 °C or, preferably, 1.5 °C above pre-industrial levels (IPCC 2022). To achieve this goal, emissions of the three main greenhouse gases (GHGs) - namely CO_2, CH_4 and N_2O - must be reduced to net zero by 2050, with each source group contributing to this reduction. In the land use sector (covering land use, land use change, and forestry; known as LULUCF), rewetting of drained peatlands has been recommended as one of the best strategies to mitigate greenhouse gas emissions (Freibauer *et al*. 2009, Humpenöder *et al*. 2020) on account of their small area but high carbon and nitrogen storage and high CO_2 and N_2O emissions (Augustin *et al*. 1996, Abel *et al*. 2019).

Rewetting of drained fen peatlands in Germany can achieve a CO_2-eq GHG mitigation of up to 40 t ha^{-1} yr^{-1} at low cost (on average €70 t^{-1}; Drösler *et al*. 2011). However, from a biodiversity perspective, the agricultural legacy of high nutrient levels and altered soil physical properties often leads to domination by species-poor reed or cattail stands after rewetting (Günther *et al*. 2015), which in many cases do not resemble the original plant communities of pristine temperate fen peatlands (Kreyling *et al*. 2021). A minimum area of wet meadow may be required to maintain fen vegetation communities of higher biodiversity value (Martens *et al.* 2023).

Compared to newly established wetland crops, wet meadows already cover a substantial area (15,000 ha) in Germany and have been proposed as a form of paludiculture (Nordt & Tanneberger 2021). Paludiculture is defined as “the agricultural or silvicultural use of wet and rewetted peatlands.” It “uses spontaneously grown or cultivated biomass from wet peatlands under conditions in which the peat is conserved or even newly formed”

(Wichtmann *et al*. 2016). CO_2-eq GHG emissions from paludiculture have been estimated at 0–8 t ha^{-1} yr^{-1} (Tanneberger *et al*. 2022).

Recent estimates suggest that there are approximately 15,000 ha of wet fen peat meadows under nature conservation management in Germany (Nordt & Tanneberger 2021). In this practice, groundwater levels (GWLs) are maintained close to the soil surface throughout the year and are lowered to 35+ cm depth (Blankenburg *et al.* 2001) during harvesting to allow the operation of machinery. This implies that mean annual GWLs are in the range 10–20 cm below ground surface.

While nature conservation management alone may not be sufficient to significantly mitigate GHG emissions from drained peatlands (Huth *et al*. 2012), and combining raised GWL with high-intensity land use may even increase GHG emissions (Tiemeyer *et al*. 2024), reduced N loading in combination with raised GWL has been shown to reduce GHG emissions in peat meadows across Germany (Tiemeyer *et al*. 2016).

Although a number of GHG balances for wet or longer-term rewetted (> 5 years) groundwater-fed peatlands have become available over the last decade (e.g. Drösler *et al*. 2013, Günther *et al*. 2015, Huth *et al*. 2018, Koebsch *et al*. 2020, Tiemeyer *et al*. 2020), none of them have included long-term low-intensity grassland use with deployment of conventional machinery and GWL drawdown during harvest. Therefore, the GHG balances of pre-existing wet fen meadows under nature conservation management are currently unknown (Harthan *et al*. 2018) and emission factors usually have to be derived from low-intensity grassland use of other peatland types (Drösler *et al*. 2013, Leiber-Sauheitl *et al*. 2014, Tiemeyer *et al*. 2016, 2020).

This study presents GHG balances for two rewetted fen meadows that have been under nature conservation management for 8 and 17 years, respectively, with stable vegetation communities dominated either by acute sedge (*Carex acuta*) or flood meadow with creeping bentgrass (*Agrostis stolonifera*) and spike rush (*Eleocharis palustris*). In this article we aim to:

i) determine the complete climate balance of a long-term wet fen meadow typical of areas under management for nature conservation in north-east Germany;

ii) compare its GHG mitigation potential with drainage-based grassland use, and

iii) consider whether this type of land use can be regarded as paludiculture.

METHODS

Study areas

The study areas 'Körziner Wiesen' (52.221806 °N, 13.096058 °E, WGS84) near Körzin and 'Krielower Wiesen' (52.422269 °N, 12.851542 °E, WGS84) near Krielow (Germany) are typical eutrophic terrestrialisation fens (Joosten *et al*. 2017) located 20 km south and south-west of Berlin as part of the nature reserves 'Nuthe-Nieplitz-Niederung' and 'Krielower See' (Figure 1). The climate of this region is temperate in a transition area between the oceanic and continental climate zones, with warm summers and mild to cold winters. The mean annual precipitation is 586 mm and the mean annual air temperature reaches 9.3 °C (German Meteorological Service Potsdam Weather Station, climate period 1980–2010; DWD 2018).

The Körziner Wiesen were used for decades as extensive meadows without manuring and a maximum of two mowing operations per year. On the other hand, the Krielower Wiesen were used intensively until 2003 with manuring, and were mown more than twice per year. They were changed to extensive meadows without manuring and mown twice per year after 2003. The mowings are used in stables as litter and subsequently applied to arable land as manure. The soils of both study areas are characterised as "Erdfen" (with a soil classification of rheic sapric histosol) with a peat thickness of up to 1.3 m. The upper 50 cm are highly decomposed (H7–H10; von Post 1924), containing high total nitrogen and sulfur concentrations of up to 3.7 and 4.5 % dry mass (Münich 1998), but show low values for plant-available phosphorus and potassium of up to 55 and 40 mg kg^{-1} (dry mass basis; Rabe 2013). The underlying soils consist of weakly decomposed *Carex* and *Phragmites* peat together with pieces of *Alnus* wood (H3– H7; Münich 1998, Gall 2004, 2005). The water sources in the Körziner Wiesen study area are the ditch 'Königsgraben' and the rivers 'Nieplitz' and 'Pfefferfließ' which both flow from the south through the lake 'Blankensee' to the north (Figure 1, lower right image). The Krielower Wiesen are half-surrounded from south to north by the 'V008' ditch (Figure 1, upper right image). At the northern end of each study area, weirs have held back water for higher groundwater levels since 1997 at the Körziner Wiesen (Landgraf 2011) and since 2008 at the Krielower Wiesen (Flächenagentur Brandenburg 2021). Halophytes - such as *Triglochin maritima* and *Plantago major* subsp. *winteri* - grow at single locations on the Körziner Wiesen. This is because gypsum-enriched groundwater ascending from a shallowly situated anhydrite layer through breaks in the overlying clay layer enriches the groundwater

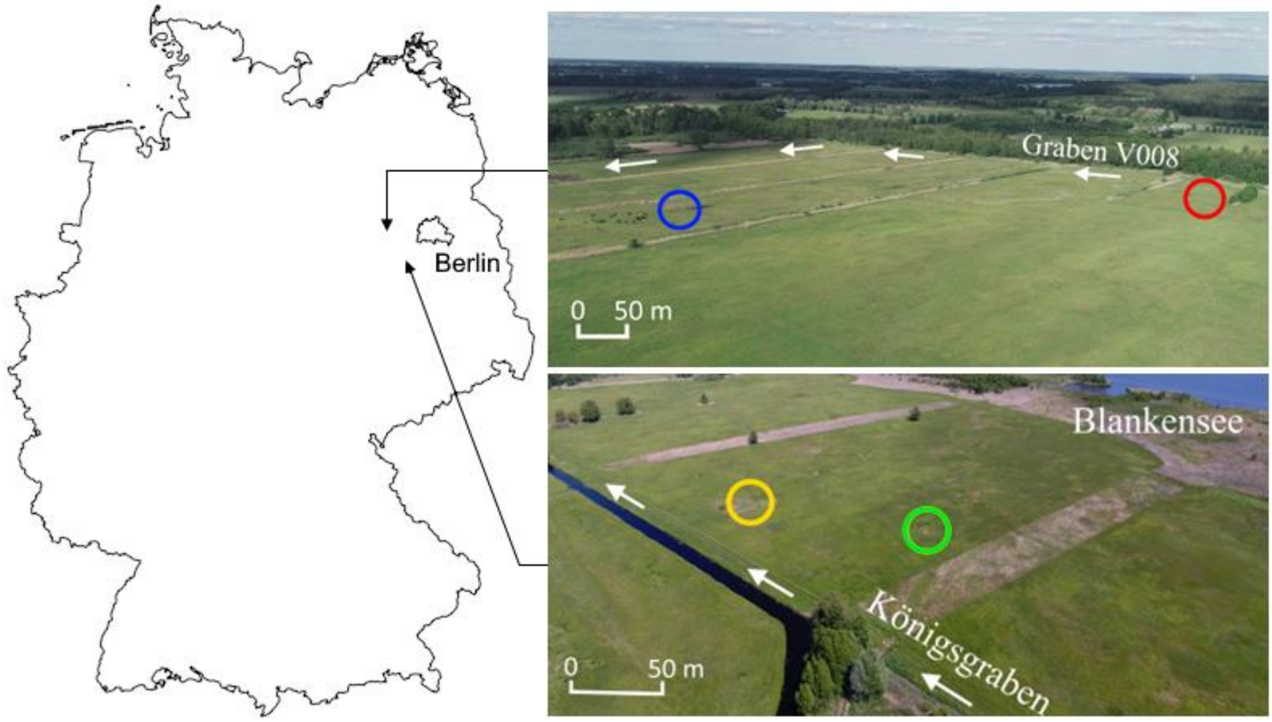

Figure 1. Map showing locations of the study areas near Berlin in Germany (left) with views across the measurement site near Krielow (upper right: Carex-Krielow O and Agrostis-Krielow O) and the site near Körzin (lower right: Carex-Körzin I O and II O); photos: R. Wolf.

with calcium sulphate in alternation with sodium chloride that enters by geogenic inland salt water intrusion (Hermsdorf 2010). This does not occur on the Krielower Wiesen due to weaker-to-absent ascending gypsum-enriched groundwater owing to an incompletely cleared clay layer and a more deeply situated anhydrite layer (Hermsdorf 2010). While the vegetation of Körziner Wiesen is dominated by acute sedge (*Carex acuta*), at Krielower Wiesen there is an alternating mosaic of *Carex acuta* and flood lawns with creeping bentgrass (*Agrostis stolonifera*) plus creeping spike rush (*Eleocharis palustris*). The Körziner Wiesen are classed as important protected habitats for threatened animals, especially for the birds that breed and roost there. Furthermore, salt meadows, wet fen meadows and the littoral zone of the Blankensee (a lake) are nature conservation areas (Landesregierung Brandenburg 1995). The development aims of the Krielower Wiesen are to improve water retention, stop soil degradation and nutrient export, and improve habitats for animals (Flächenagentur Brandenburg 2021).

Experimental set-up

During winter 2013/2014, one measurement site was established at Körziner Wiesen (named 'Carex-Körzin I'), and two measurement sites were established at Krielower Wiesen (named 'Carex-Krielow' and 'Agrostis-Krielow'). In February 2014, three in-line replicate plots (3.8 m × 3 m) were chosen for measurements at each site. One square PVC frame (75 cm × 75 cm × 25 cm) per plot was inserted 13 cm deep into the peat with 3 m distance to the next frame. Each measurement site was equipped with four mini-piezometers and four suction cups per plot, one groundwater pipe, one data logger box and three soil sensors. Boardwalks were built around the frames to avoid physical disturbance of the soil during fieldwork. Measurements with manual static closed chambers were done biweekly for all GHGs, resulting in 49 measurement days for each measurement site. On the 7th and 14th July 2017, measurements were taken for two weeks at an additional measurement location at Körziner Wiesen ('Carex-Körzin II') to explore maximum methane emissions under sulphate reduction following extraordinarily heavy rainfall and freshwater flooding of the site. Here, four plots were chosen where only four suction cups per plot were pre-installed and the chambers were deployed without frames or boardwalks.

Sites Carex-Körzin I and II were dominated by *Carex acuta* with moderate (30 %) cover of *Drepanocladus aduncus* on highly degraded peat (H10; von Post 1924). Peat thickness was 1.1 m. Site Carex-Krielow was also dominated by *Carex acuta*, but moss cover reached 95 % of mostly *Drepanocladus aduncus* on highly degraded peat of thickness 1.3 m. At Agrostis-Krielow, *Agrostis stolonifera* grew together with *Eleocharis palustris* in the herb layer of a flood lawn where *Drepanocladus aduncus* cover was 100 %. The peat layer was highly degraded and 1.1 m thick.

Measurements and calculations

Environmental conditions

At all three study sites a weather station recorded air temperature (VP-3, Decagon), precipitation (ECRN-100, Decagon) and photosynthetically active radiation (PAR, QSO-S, Decagon) at 2 m height and 30-min intervals. Groundwater levels were measured in groundwater pipes at 30-min intervals by a differential pressure transducer (CTD-10, Decagon). Volumetric water content (VWC), electrical conductivity and soil temperature were monitored at 30-min intervals by soil sensors at 5, 10 and 20 cm depth. VWC values were later transformed into values of water-filled pore space (WFPS). Alongside this, WFPS values from sensors were amended according to WFPS values generated from manually retrieved soil samples at four different GWL depths. All devices were connected to data loggers (EM50, Decagon) that stored all of the data. For the analyses of dissolved organic carbon (DOC), NH_4^+, NO_3^- and SO_4^{2-}, soil pore water and groundwater samples were collected at each plot during the (afternoon) gas measurements, using suction cups at 15, 30, 50 and 70 cm depth and mini-piezometers at 30, 55, 80 and 160 cm depth. During rainfall events, precipitation was collected in a 10 L bucket per site for measurement of DOC, NH_4^+ and NO_3^- concentrations. Water samples were treated with HCl (pH < 2) in the field and stored at -18 °C afterwards. After water filtration with a 0.45 µm filter (Minisart RC 25, VWR), water samples were analysed for DOC concentration with Liqui-TOC (Elementar Analysensysteme) and for NH_4^+ plus NO_3^- with a continuous flow analyser (Skalar). SO_4^{2-} concentrations were measured after precipitation by means of a BaCl-cuvette test (LCK 353, Hach Lange) with a photometer (DR 2800, Hach Lange). Frequent occurrence of H_2S was measured with an H_2S sensor (Gas Badge Pro, Siegrist). Soil sampling was performed after cessation of all GHG measurements, from 29 July to 03 August 2016 below the GHG frames and on one additional plot 3 m distant from the other plots per measurement site. pH values were measured on seven dates from May 2015 to June 2016 in the soil pore water collected at 15, 30, 50 and 70 cm depth and once for the soil samples from the soil profiles, all in $CaCl_2$ solution. Mapped vegetation was named after the nomenclature of Wisskirchen & Haeupler (1998) for vascular plants, and after Koperski *et al*. (2000) for mosses.

GHG exchange measurements

For GHG exchange measurements, rectangular PVC chambers (78 cm × 78 cm × 50 cm) were deployed on the pre-installed PVC frames. Airtight closure between chamber and frame was ensured by rubber gaskets fixed to the edges of the chamber bottoms. When the vegetation exceeded the chamber height, PVC chamber extensions (78 cm × 78 cm × 50 cm) were added. The CO_2 fluxes were measured in flow-through non-steady-state mode (Livingston & Hutchinson 1995) biweekly from May 2014 to May 2016. Measurements started before sunrise and ended at late afternoon to cover the full diurnal range of PAR and temperature (Beetz *et al*. 2013, Leiber-Sauheitl *et al*. 2014, Günther *et al*. 2015). Transparent chambers with 90 % light transmission were used for net ecosystem exchange (NEE) measurements, alternating with opaque chambers to measure ecosystem respiration (R_{eco}). Each chamber was equipped with a temperature sensor and a fan to avoid stratification. The chambers were connected to an external pump that provided a continuous air flow, and a portable infrared CO_2 analyser (LI-820, LI-COR) that measured CO_2 concentrations at 1 s time intervals. Furthermore, a PAR sensor (type X.3, Indium Sensor) positioned 0.5 m above the canopy on a stand and a mini laptop (Scieneo Amplio) were connected to the CO_2 analyser. The software program 'CO2 Flux 1.0.3' (J. Hofmann, CALMEX, Rostock) recorded and synchronised all data and depicted it in graphical form as a real-time diagram. A measurement (corresponding to the chamber closure time) for NEE lasted from 90 s to 3 min and for R_{eco} from 120 s to 5 min. Measurements were disregarded whenever PAR changed by more than 10 % of the starting value, or when chamber temperature changed by more than 1.5°C, during chamber closure.

For CH_4 and N_2O concentrations we sampled manually in non-flow-through, non-steady-state mode with opaque PVC chambers (Livingston & Hutchinson 1995). Every plot was sampled once biweekly from May 2014 to May 2016, on the day of the CO_2 measurements (between 1:00 p.m. and 3:00 p.m.). Each chamber was equipped with a vent on the upper side wall to avoid the occurrence of air-pressure artefacts during deployment, another vent on top of the chamber for air sampling, and one thermometer. During a chamber deployment (of duration one hour), seven samples were taken at intervals of 10 min with 60 ml syringes and stored inside 20 ml vials at 5 °C. The samples were analysed the following week for CH_4, N_2O and CO_2 concentrations using a gas chromatograph (GC-2010 Plus, Shimadzu Scientific Instruments) equipped with a flame ionisation detector for CH_4 and an electron capture detector for N_2O and CO_2.

CO_2 flux calculation and annual balance estimation
The CO_2 flux calculation was performed using the software program 'CO_2 Flux 1.0.3', which identified the best fit of a linear increase or decrease in CO_2 concentration over time, with a moving window considering the highest determination coefficient (R^2). Near-zero CO_2 fluxes (during winter or at light compensation points) with low R^2 (< 0.5) were not discarded (Alm *et al.* 2007). Fluxes were determined following the formula proposed by Flessa *et al*. (1998):

$$F = k \frac{273}{T} \frac{V}{A} \frac{\mathrm{d}c}{\mathrm{d}t} \qquad [1]$$

where F is the calculated CO_2-C flux (mg m^{-2} h^{-1}), k is a unit conversion factor (0.536 kg m^{-3} C in CO_2), T is the temperature inside the chamber (K), V is the total volume of the chamber (L), A is the area of the frame (m²) and dc/dt is the linear CO_2 concentration change over time in the chamber (µl L^{-1} h^{-1}). All fluxes and exports from the ecosystem to the atmosphere are defined as fluxes with a positive sign and all fluxes/inputs into the ecosystem are defined as fluxes with a negative sign (using the atmospheric sign convention). To estimate annual CO_2 balances, artificial neural networks (ANNs) were used to model net ecosystem CO_2 exchange (Huth *et al*. 2022) using the package 'neuralnet' (Fritsch *et al*. 2019) in R (R Core Team 2021). A backpropagation algorithm was deployed to create half-hourly time series of CO_2 fluxes. For each plot and year, 100 ANNs (two hidden layers with four and two nodes) were trained separately, each time randomly using 80 % of the measured fluxes and environmental input data to account for uncertainty associated with discontinuous flux measurements. Environmental input data were PAR, air temperature, soil temperatures at 5 and 10 cm depth, groundwater level and vegetation height. ANNs were validated with the remaining 20 % of the measured fluxes, yielding an adjusted R^2 of between 91 and 95 %. ANNs and half-hourly environmental data were used to calculate 100 time series of half-hourly CO_2 fluxes for each plot and year, from which the mean annual CO_2 balance ± 1 SE was calculated.

CH_4 and N_2O flux calculation and annual balances
CH_4 and N_2O concentrations were checked by the package 'HMR 0.4.1' (Pedersen 2015) for linear, non-linear or zero increase over time. In cases of linearity, Equation 1 was used with k_{CH4} = 0.536 kg C m^{-3} or k_{N2O} = 1.25 kg N m^{-3}, accepting only fluxes of $R^2 \geq 0.75$. With non-linearity, fluxes were determined by the Hutchinson-Mosier regression model of the HMR package (Hutchinson & Mosier 1981). For estimating annual balances, CH_4 and N_2O fluxes were tested on the basis of best correlation with environmental data, resulting in linear and polynomial regression models with groundwater level and soil temperature at 10 cm depth (Spearman's rank correlation $r_S \geq 0.7$, models: $R^2 > 0.82$, $RMSE = 0.02$–5.65). The models were then used to generate hourly fluxes as follows: the balances and associated uncertainties were estimated by modelling hourly fluxes for all 49 measurement days, each time leaving out one measurement day (the 'delete-1 Jack-knife' method). 49 annual balances were then calculated by summing up modelled hourly fluxes. The mean and standard error of the 49 balances were then the reported balances and uncertainties for each measurement site.

CO_2 from harvest export
Harvesting at the measurement points and the three additional locations was carried out some days after their surroundings were mowed by farmers. Körziner Wiesen was harvested once in July 2014 and once in August 2015, while Krielower Wiesen was harvested once in July 2014 and twice in 2015 (July and August/September). The biomass within the frames was cut with a set of electric shears (Gardena Classic Cut) at ~10 cm height, weighed, dried for three days at 60 °C, then weighed again. Afterwards the biomass was shredded (Hege 44, Wintersteiger) and then ground to powder in mills (Retsch SM and MM 301). Finally, each sample was separated into two subsamples for calculating CO_2-C export: one subsample was used for determining the C content by means of a CN analyser (vario Max, Elementar Analysensysteme) and the second sub-sample was dried at 105 °C until constant mass was achieved to derive dry matter yield.

DOC export from ditches
DOC was sampled bimonthly from June 2015 to June 2016 (seven sampling days in total), in ditchwater at the inflow and outflow points of both study areas. For this we used a steel rod and 100 ml low density polyethylene (LDPE) vessels to sample against the

flow direction, from the middles of the ditches ~15 cm below the water surface. The ditchwater originates from sandy ditches and rivers in landscapes with mineral soils. Before starting the sampling, depths and widths of the ditches were measured to calculate their cross-sectional areas. After every sampling, water flow velocity was measured with an electromagnetic flow sensor (Nautilus C 2000, Ott Hydromet GmbH). Water samples were treated in situ with HCl (pH < 2) and stored at -18 °C. After filtration with a 0.45 µm membrane filter (Minisart RC 25 by VWR), the water samples were analysed for DOC concentration by high temperature oxidation using Liqui-TOC (Elementar Analysensysteme). Finally, the DOC export was calculated as the difference between the discharge of the inflow and outflow sampling points.

Net GHG balances

The net GHG balances of the measurement sites were calculated by converting all fluxes to CO_2 equivalents (CO_2-eq) using a 100-year time horizon (CO_2 = 1, CH_4 = 27, N_2O = 273; Forster *et al*. 2021) or the factor 44/12 for C in harvested plants and DOC exports. CH_4 emissions from the ditches were not measured. Therefore, the IPCC CH_4 emission factor for temperate shallow-drained grassland (IPCC 2014) and a ditch-area proportion of 1.3 % (Tiemeyer *et al*. 2017) were used to calculate CH_4 emissions from ditches. The errors in the net GHG balances were calculated by integrating the individual errors (in CO_2-eq), following the law of error propagation (Günther *et al*. 2015).

RESULTS

Environmental conditions

During the 25-month study period, the mean daily air temperature ranged from -8.8 to 27.9 °C (Figure 2, upper panel) and the mean daily soil temperature at 10 cm depth from 1.7 to 23.3 °C (not shown). The

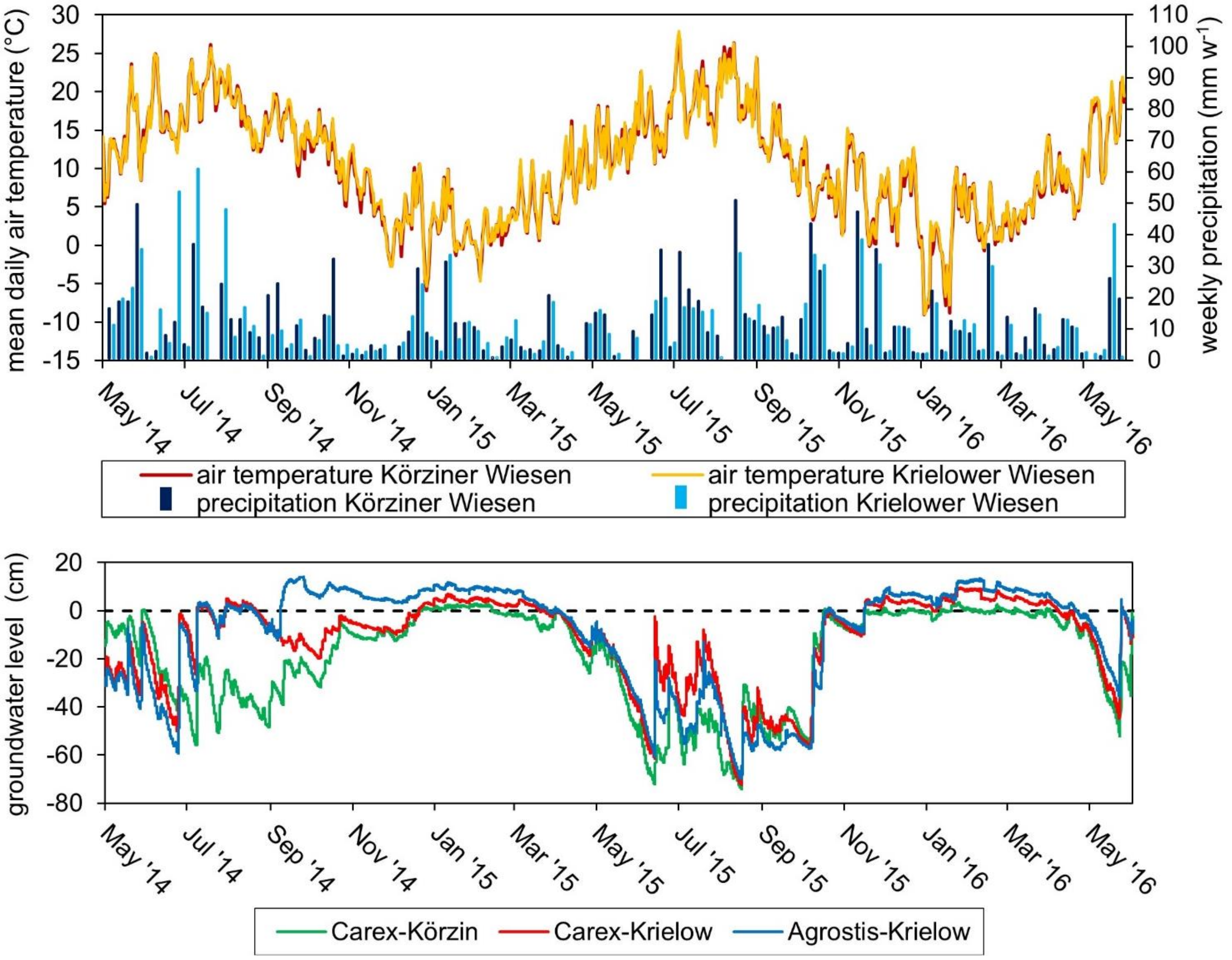

Figure 2. Environmental conditions recorded on the measurement sites. Upper panel: mean daily air temperature (°C) and total weekly precipitation (mm w^{-1}); lower panel: groundwater level dynamics (cm).

annual precipitation varied from 538 to 638 mm yr^{-1}, which is around the 30-year average for the weather station at Potsdam (586 mm yr^{-1}, climate period 1980–2010; German Meteorological Service (DWD) 2018). However, July 2014 was exceptionally wet with a total precipitation of 122 mm (50 % higher than the long-term mean), whereas May 2015 was extremely dry with 22 mm (65 % lower). 60 to 70 % of the annual precipitation fell during the growing season (April to October). Snow cover occurred on only a few days in the last week of January 2015 and in the first week of January 2016. Groundwater levels (GWLs) varied as a function of precipitation, evapotranspiration, water retention by weirs and beaver activity, from 72 cm below ground level in summer to 14.1 cm above the ground surface in winter (Figure 2, lower panel). While drought and heavy rain events caused GWLs to fluctuate between the surface and 72 cm depth during the growing season, from October to April they remained almost constant (from near the surface to 13.4 cm above it). A beaver dam that had been built and removed twice in the 'V008' ditch at the northern end of Krielower Wiesen caused flooding during the growing season from August 2014. Agrostis-Krielow was closer to this dam, such that the GWL rose to 13 cm above the surface; while Carex-Krielow was farther away, resulting in a GWL of 3 cm above the surface at maximum. Consequently, at all sites, WFPS at 10 cm depth varied between 96 and 100 % from autumn to spring and decreased to 87.6 % at deepest GWL in summer. During extraordinarily heavy rainfall from the end of June to the middle of August 2017 (total precipitation 303 mm, not shown), single daily precipitation events from 13.1 to 73.5 mm led to permanently flooded conditions with GWLs 5–10 cm above the ground surface over large parts of Körziner Wiesen. Soil characteristics of the three sites are shown in Table 1, exhibiting similar values for C_{org}, N_t, S_t and pH ($CaCl_2$) but differing in mean GWL and SO_4^{2-} content. High DOC concentrations from 100 to 470 mg L^{-1} were observed in the soil pore water from 15 to 50 cm depth at Carex-Krielow and Agrostis-Krielow throughout the study period. Carex-Körzin had lower DOC concentrations of less than 100 mg L^{-1} (Table 1). Ammonia was the main form of N in pore water, with concentrations of up to 3.5 mg L^{-1} compared to nitrate-N concentrations of up to 0.3 mg L^{-1}. SO_4^{2-} concentrations in soil pore water were high at all depths at Carex-Körzin and Agrostis-Krielow (337–1308 mg L^{-1}) compared to much lower SO_4^{2-} concentrations at Carex-Krielow (0.4–251 mg L^{-1}). Whenever GWL increased, SO_4^{2-} concentrations decreased (sulphate reduction),

Table 1. Selected soil characteristics at the three measurement sites, based on samples (n = 4) collected at 15 cm depth from 29th July to 3rd August 2016. Values are means with standard deviation, except for DOC, sulphate, iron, electrical conductivity and groundwater level (GWL).

Site		Carex-Körzin I	Carex-Krielow	Agrostis-Krielow
Dominant taxa		*Carex acuta*	*Carex acuta*	*Agrostis stolonifera, Eleocharis palustris*
C_{org} (mg g^{-1})		345 ± 8	403 ± 10	388 ± 14
N_t (mg g^{-1})		33 ± 0.7	36 ± 0.7	33 ± 0.4
S_t (mg g^{-1})		12 ± 4	9 ± 1	12 ± 4
DOC (mg L^{-1})[a]		40	142.8	267.7
SO_4^{2-} (mg L^{-1})[a]		1306	5.5	1150
Fe_t (mg L^{-1})[a]		4.2	11.9	18.7
Dry bulk density (mg cm^{-3})		279 ± 6	262 ± 15	328 ± 19
pH ($CaCl_2$)		5.8	5.8	5.9
Electrical conductivity (mS cm^{-1})[b]		1.44–2.45	0.43–1.81	0.3–3.2
Mean GWL (cm)	2014–2015	-15.2	-7.2	-2.8
	2015–2016	-22.6	-16.2	-16.8

[a] One total value for mixed samples of soil porewater taken from 3 mini-piezometers per measurement site on 20–21 June 2016.

[b] Minimum–maximum values at 1 m depth recorded during the greenhouse gas measurements from May 2014 to May 2016.

whereas SO_4^{2-} increased when GWL decreased (sulphide oxidation). Furthermore, extremely high H_2S concentrations of up to 500 ppm were detected in soil pore water at 30–70 cm depth at Carex-Körzin on 80 % of all measurement days, in contrast to the lower H_2S concentrations (< 2 ppm) at 50–70 cm depth at Agrostis-Krielow, and no detectable H_2S release in Carex-Krielow at all depths. pH values of soil pore water varied from 6.8 to 7.6 at 15 cm depth during the period May 2015 to June 2016, with lower values in summer and higher values in late winter.

CO_2 exchange

The CO_2 fluxes of the three measurement sites showed clear seasonal patterns (expressed in CO_2-C) depending on environmental conditions and plant growth (Figure 3). Low net ecosystem exchange (NEE) up to -137 mg m^{-2} h^{-1} and near-zero ecosystem respiration (R_{eco}) from 0.02 to 50 mg m^{-2} h^{-1} were observed from late autumn to early spring at low PAR and low temperatures. The highest fluxes, with NEE up to -795 mg m^{-2} h^{-1} and R_{eco} up to 602 mg m^{-2} h^{-1}, were measured in summer with mature vegetation, PAR up to 2350 µmol m^{-2} s^{-1} and high microbial activity due to high soil temperatures. Dead vegetation and low cover of living vegetation at Agrostis-Krielow early in the growing season of 2015 led to low NEE fluxes (lower than -42 mg m^{-2} h^{-1} in May 2015 vs. -427 mg m^{-2} h^{-1} in May 2014). After harvest, R_{eco} outweighed NEE for at least one week. Validation of gap-filled CO_2 showed very good 1:1 agreement, with adjusted R^2 between 91 and 95 % for all runs, sites and years. If we disregard CO_2 release from the biomass after harvest, Carex-Körzin and Carex-Krielow had similar mean annual net CO_2 balances of 418 and 563 g m^{-2} yr^{-1}. Agrostis-Krielow was a stronger mean annual source of CO_2 at 1129 g m^{-2} yr^{-1}.

CH_4 fluxes

CH_4-C fluxes were relatively low throughout the study period apart from for the flooding event at the Krielow sites starting in late summer 2014 (Figure 4, upper panel), which was accompanied by higher

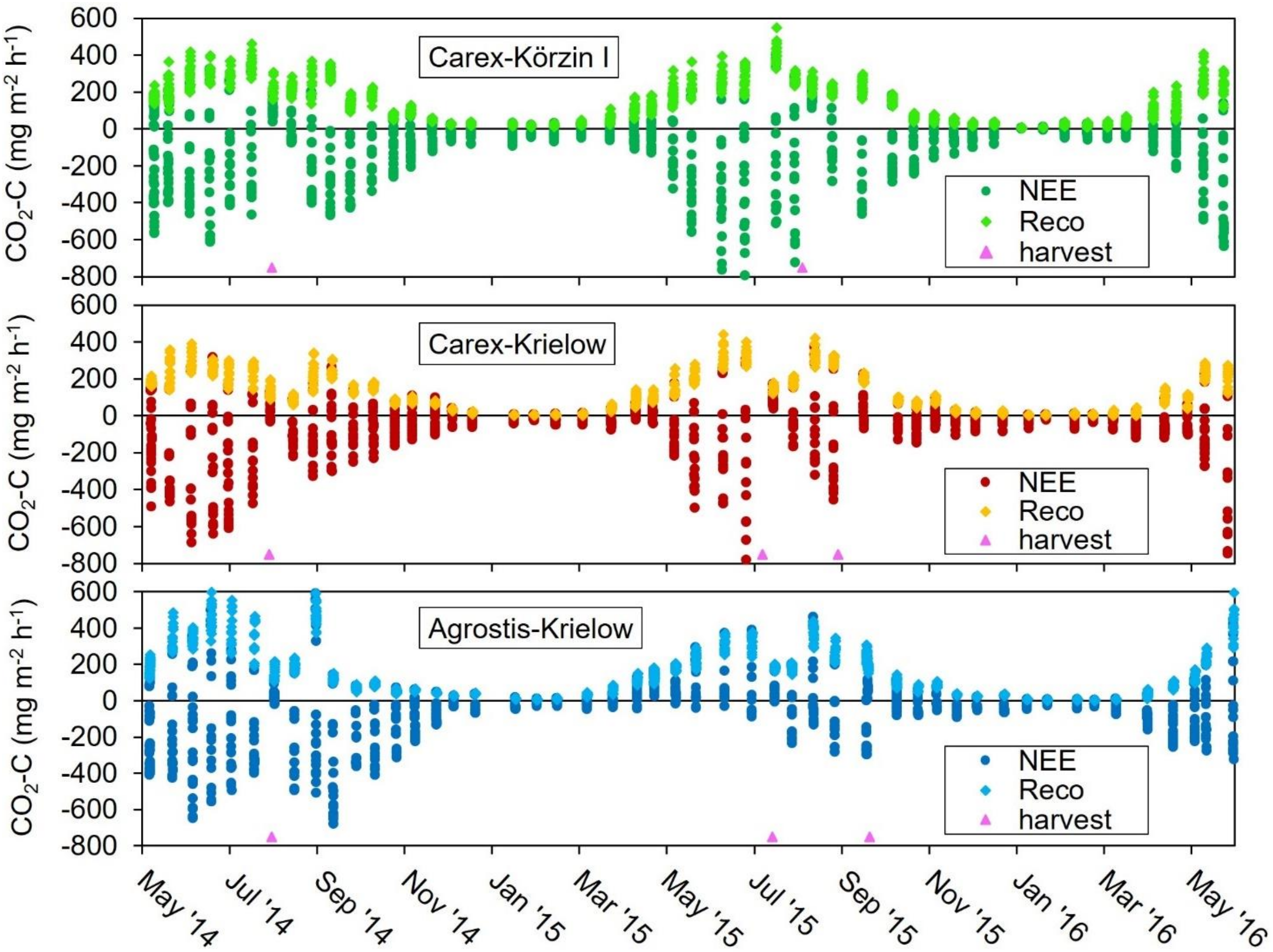

Figure 3. Measured values of NEE (darker circles) and R_{eco} (lighter squares) with harvest dates (magenta triangles) at the sites Carex-Körzin I, Carex-Krielow and Agrostis-Krielow.

DOC concentrations (up to 452 mg L^{-1}) and higher soil temperatures (CH_4-C emissions up to 8.46 mg m^{-2} h^{-1} at Carex-Krielow and up to 3.71 mg m^{-2} h^{-1} at Agrostis-Krielow). During the extraordinarily heavy rainfall and freshwater flooding of the Körzin site from the end of June to the middle of August 2017, CH_4-C emissions remained below 0.64 mg m^{-2} h^{-1} at the measurement site Carex-Körzin II (data not shown). After harvest at the end of July 2014, CH_4-C fluxes increased at Carex-Krielow but decreased at Agrostis-Krielow. After removal of the beaver dam on the northern part of the quarter ditch at Krielow in August and a drop in GWL from 14.1 cm above to 19.5 cm below the ground surface, CH_4-C fluxes decreased but remained substantial until the middle of November 2014. Thereafter, CH_4-C fluxes were lower than 0.69 mg m^{-2} h^{-1} with a flux magnitude order of: Carex-Körzin I + II < Agrostis-Krielow < Carex-Krielow. CH_4-C uptake rates of up to -0.11 mg m^{-2} h^{-1} were measured only from June to September 2015 at Carex-Körzin and Agrostis-Krielow, starting at GWLs deeper than 50 cm depth. At all sites, CH_4-C fluxes correlated most strongly to GWL (r_S up to 0.78) followed by WFPS, DOC and soil temperature at 10 cm depth. Annual accumulated CH_4-C balances ranged from 0.68 kg ha^{-1} yr^{-1} at Carex-Körzin I in year 2 to 104.94 kg ha^{-1} yr^{-1}at Carex-Krielow in year 1. Regarding the year 1 CH_4-C balances at the Krielow sites, the heavy rainfall of late summer 2014 accounted for up to 71 % and the subsequent beaver dam for up to 30 % of the annual balances.

N_2O fluxes

N_2O-N fluxes were similar across all sites (Figure 4, lower panel). The highest fluxes of up to 71.76 µg m^{-2} h^{-1} were observed when GWL was below 52 cm depth and WFPS at 10 cm depth was below 87.6 % (typically in summer). Lower N_2O-N fluxes or uptake (between 52.26 and -47.03 µg m^{-2} h^{-1}) occurred at GWLs above 30 cm depth (Figures 2 and 4) and WFPS at 10 cm depth between 96 and 100 %. At all sites, N_2O-N fluxes correlated most strongly to GWL (r_S = -0.83 to -0.91) followed by WFPS (r_S = -0.67 to -0.91). The third strongest driver was soil temperature at 10 cm depth with r_S = 0.54–0.69. No influence of nitrate in the soil pore water on N_2O-N fluxes could be detected. Annual N_2O-N balances varied from -0.23 kg ha^{-1} yr^{-1} at Carex-Krielow in year 1 to 0.65 kg ha^{-1} yr^{-1} at Agrostis-Krielow in year 2.

Dry matter yield and CO_2 from biomass export

In 2014, dry matter yields were similar between the three measurement sites, at around 3 t ha^{-1} yr^{-1}

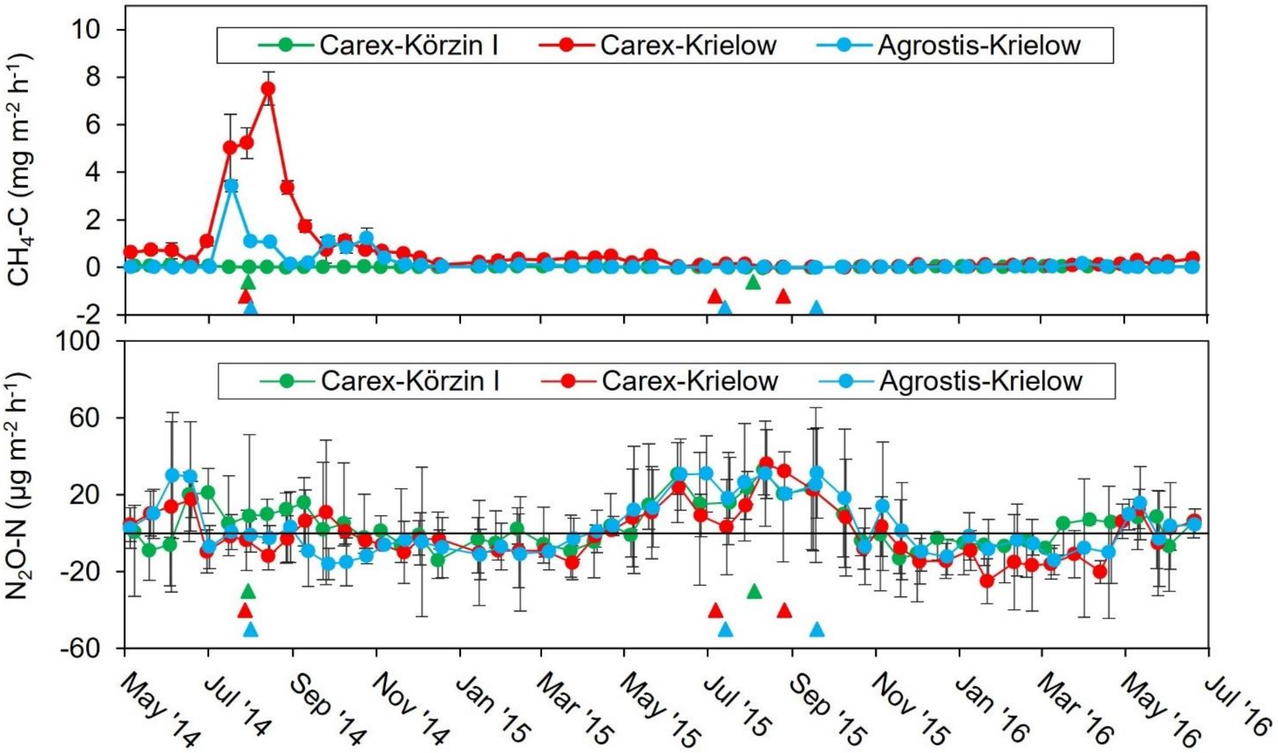

Figure 4. CH_4-C and N_2O-N fluxes (dots) at the sites Carex-Körzin I, Carex-Krielow and Agrostis-Krielow with n = 3 and standard errors (bars). Triangles indicate the dates of harvests: Carex-Körzin I **Δ**, Carex-Krielow **Δ** and Agrostis-Krielow **Δ**.

(Table 2). Dry matter yields at Carex-Körzin (one harvest) and Carex-Krielow (two harvests) were ~1 t higher in 2015 than in 2014, whereas dry matter yields at Agrostis-Krielow were ~1 t lower in 2015 than in 2014. With a measured C content of 45.3 % in leaves of *Agrostis stolonifera* and 47.7 % in leaves of *Carex acuta*, the exported biomass resulted in accountable CO_2 emissions from 3.1 t ha^{-1} yr^{-1} for Agrostis-Krielow to 7.2 t ha^{-1} yr^{-1} for Carex-Krielow (Table 3).

DOC export from ditches

From June 2015 to June 2016, alternating surplus and deficiency of DOC was observed in the ditches 'Königsgraben' and 'Graben V008' at the outflows of the two study areas (Table 4). Weak DOC deficiency occurred on the majority of normal sampling days without ditch disturbance. DOC surplus arose only after disturbances such as ditch clearing by excavator and crossing of ditches by animals. Therefore, DOC exports from ditches were assessed as contributing zero kg ha^{-1} yr^{-1} of CO_2 equivalents to the GHG balance for all measurement sites.

Net GHG balance

The net GHG balances of the sites ranged from 7.7 to 18.5 t ha^{-1} yr^{-1}, with Carex-Körzin emitting the least and Agrostis-Krielow the most (Table 3). Overall, CO_2 dominated the GHG balances at all sites and in both years, with proportional contributions ranging from 67.5 % for Carex-Krielow in 2014 to 97.3 % for Carex-Körzin in 2014. At the Carex sites, the CO_2-eq export by harvesting activities was higher than CO_2 emissions from plants and soil overall, although this alternated between the two years (Table 3). CH_4 emissions up to 3.8 t ha^{-1} yr^{-1} (in CO_2-eq) contributed substantially to the overall GHG balances of year 1 at Carex-Krielow and Agrostis-Krielow following the late-summer flooding of 2014. N_2O emissions contributed only up to 0.3 t ha^{-1} yr^{-1} (in CO_2-eq) to the total GHG balances of the sites (Table 3).

DISCUSSION

The robust validation provided by our fortnightly measurement campaigns over almost the entire study period indicates that this was an adequate frequency for the manual GHG measurements. The net CO_2 balances in this study are in the same range as for other extensive moist to wet meadows with sedges (Drösler *et al*. 2013, Günther *et al.* 2015, Huth *et al*. 2018), for flood lawns (Kandel *et al*. 2017, Reichelt & Lechtape 2018), or for shallow-drained, nutrient-rich grasslands (IPCC 2014). Although average GWLs were higher at the Krielow sites, as a result of heavy precipitation and the construction of beaver dams in 2014, NEE was lowest (i.e., closest to net zero) at Carex-Körzin. This counter-intuitive pattern is due to flooding of the Krielow sites in late summer 2014, which led to a partial dying-off of vegetation (especially at Agrostis-Krielow) that could not be compensated for by new plant growth within the study period. NEE almost doubled from year 1 to year 2 at Agrostis-Krielow, suggesting a high component of heterotrophic respiration in 2015 following vegetation die-off in late 2014. *Carex*-dominated meadows are more resilient to fluctuating water tables than flood lawns featuring *Agrostis* or other Poaceae species because their aerenchymatous rooting system is better able to resist inundation and prolonged anoxic periods (Huth *et al.* 2018) than *Agrostis*, which tends to form mats that are able to grow rapidly across shallow water but, once rooted, are often prone to inundation. Our study shows that short-term weather events can override emission patterns, at least for a certain time (Huth *et al*. 2018, Koebsch *et al*. 2020).

'CO_2 emissions through harvest' is the second most important component of the GHG balance after NEE in our study (Table 3). A high GWL during the growing season limits soil respiration resulting in an NEE mostly dominated by net primary production of plants. The NEE balance of the meadows is then dominated by CO_2 arising from harvest activities, as

Table 2. Total annual dry matter (105 °C) yields at the measurement sites Carex-Körzin I, Carex-Krielow and Agrostis-Krielow in 2014 and 2015 (n = 6). Values are means with standard deviation. Different letters indicate significant differences among the sites in each year.

Site	Carex-Körzin I	Carex-Krielow	Agrostis-Krielow
Dry matter 2014 (t ha^{-1} yr^{-1})	3.06 ± 0.42 a	2.83 ± 0.13 a	2.86 ± 0.31 a
Dry matter 2015 (t ha^{-1} yr^{-1})	3.62 ± 0.45 a	4.12 ± 0.20 b	1.85 ± 0.37 c

Table 3. GHG balances (in CO_2-eq) of the three measurement sites Carex-Körzin, Carex-Krielow and Agrostis-Krielow for individual years and in total, from 01 May 2014 to 31 May 2016 (with standard errors in brackets), and their associated annual mean GWL values. CO_2-eq emissions of CH_4 and N_2O are calculated using conversion factors of 27 and 273, respectively, for a 100-year time horizon (GWP100) (Forster *et al*. 2021).

Year	Measurement site	NEE (CO_2) (soil + plants) ($t\ ha^{-1}\ yr^{-1}$) (n = 3)	CO_2 (harvest) ($t\ ha^{-1}\ yr^{-1}$) (n = 6)	CH_4 (soil) ($kg\ ha^{-1}\ yr^{-1}$) (n = 3)	N_2O (soil) ($kg\ ha^{-1}\ yr^{-1}$) (n = 3)	DOC export (ditches) ($t\ ha^{-1}\ yr^{-1}$)	CH_4 emission (ditches) ($kg\ ha^{-1}\ yr^{-1}$)	Mean GWL (cm)	Total GHG balance ($t\ ha^{-1}\ yr^{-1}$)
2014 ↓ 2015	Carex-Körzin I	7.3 (0.2)	5.4 (0.3)	79.8 (1.6)	104.5 (0.9)	0	185 (43)	-15.2	13.1 (1.4)
	Carex-Krielow	3.0 (0.2)	4.9 (0.1)	3777.7 (49.0)	-99.6 (0.6)	0	185 (43)	-7.2	11.7 (2.1)
	Agrostis-Krielow	7.8 (0.6)	4.9 (0.2)	1260.2 (4.7)	-41.4 (0.5)	0	185 (43)	-2.8	14.1 (1.5)
2015 ↓ 2016	Carex-Körzin I	1.0 (0.2)	6.2 (0.3)	24.6 (1.0)	255.6 (1.0)	0	185 (43)	-22.6	7.7 (1.4)
	Carex-Krielow	8.3 (0.2)	7.2 (0.1)	252.3 (4.9)	140.1 (0.6)	0	185 (43)	-16.2	16.1 (1.4)
	Agrostis-Krielow	14.8 (0.3)	3.1 (0.3)	60.9 (1.7)	279.5 (0.7)	0	185 (43)	-16.8	18.5 (1.4)
Ø 2014 ↓ 2016	Carex-Körzin I	4.2 (0.2)	5.8 (0.3)	52.2 (1.0)	180.1 (0.9)	0	185 (43)	**-18.9**	**10.4 (1.4)**
	Carex-Krielow	5.6 (0.2)	6.0 (0.5)	2015.0 (24.7)	20.2 (0.6)	0	185 (43)	**-11.7**	**13.9 (1.7)**
	Agrostis-Krielow	11.3 (0.5)	4.0 (0.5)	660.5 (2.6)	119.1 (0.6)	0	185 (43)	**-9.8**	**16.3 (1.5)**

Table 4. DOC inflow and outflow rates ($mg\ s^{-1}$) in the ditches 'Königsgraben' and 'V008' in the study areas Körzin and Kielow, respectively, from June 2015 to July 2016. * = ditch clearing by excavator; ** = ditch crossing by animals.

Site	Location	04/05 Jun 2015	04/05 Aug 2015	02/05 Oct 2015	02/12 Dec 2015	05/16 Feb 2016	22/28 Apr 2016	28 Jun / 01 Jul 2016
Körzin	inflow	21.6	33.8	22.3	93.6	156.3	108.0	30.1
	outflow	18.9	29.7	54.0*	94.2	135.3	126.0	29.8
Krielow	inflow	425.1	665.6	234.9	660.1	550.4	432.3	229.3
	outflow	291.9	305.5	50.1	1436.4*	507.0	461.9**	140.5

at Carex-Krielow in 2014. In addition, favourable conditions for plant growth (e.g., optimal water supply) may also induce a high below-ground net primary productivity that could compensate for respiratory losses of the peat soil as, for example, at Carex-Körzin I in 2015. At Agrostis-Krielow the biomass harvest in 2015 was less than 60 % of the harvest in 2014, which was caused by later, fewer and lower sprouting as a result of vegetation die-off in late 2014. In sedge reeds, dry matter yields of 2–12 t ha^{-1} yr^{-1} are possible (BfN 2021). The dry matter yields achieved at our study sites lie at the lower limit of the aforementioned range of sedge reeds and are characteristic of extensive wet meadows and *Carex acuta* fens with low nutrient supply (Busch 2001, Grzelak *et al*. 2011, Wallor *et al*. 2014, Huth *et al*. 2018). The main drivers of these low dry matter yields are the high GWL (Harter & Luthardt 1996, Käding *et al*. 2005), the low nitrate and potassium supply, the absence of manuring (Käding *et al*. 2005, farmers' advice), the low harvest frequency (Wallor & Zeitz 2016), and the more-slender leaves and lower shoot density of *Carex acuta* (Busch 2001). Additionally, the dense moss layer impedes soil warming and sprouting of other plants, again resulting in lower dry matter yields (Klapp & Stählin 1936, Bernard *et al*. 1988).

All three measurement sites were weak CH_4 sources during the study period. The only substantial CH_4 emissions occurred during the late-summer inundation of 2014 at the Krielow sites. Outside the inundation period, CH_4 fluxes indicated both weak sources and sinks, with the latter mostly occurring when the GWL dropped below -50 cm, as happened from June to September 2015 at Carex-Körzin and Agrostis-Krielow. Hence, the measured CH_4 balances lie towards the lower end of the range reported for temperate rewetted rich fens and wet meadows (IPCC 2014, Günther *et al*. 2015, Kandel *et al*. 2017, Huth *et al*. 2018) as well as published data for pristine fens (Abdalla *et al*. 2016). Therefore, eight years after rewetting, our results do not indicate increased CH_4 emissions on formerly drained nutrient-rich fen peatlands as discussed by some authors (Augustin & Joosten 2007, Günther *et al*. 2020, Evans & Gauci 2023). The previously mentioned seasonal trend of lower CH_4 emissions from late autumn to spring at high GWL is caused by the high temperature sensitivity of the enzymes of methanogenesis, resulting in an increase in CH_4 emissions as temperatures rise in summer at high GWL and in a rapid decrease in CH_4 emissions at lower temperatures from late autumn to spring at high GWL (Schipper *et al*. 2014). Furthermore, the supply of root exudates and plant litter for methanogens is greater in summer, often leading to the highest CH_4 emissions occurring late in the growing season (Koebsch *et al*. 2013). However, except in 2014, CH_4 emissions remained low during the growing season, likely because periods of high GWL were simply too short for methanogenesis since other time-consuming reactions such as denitrification, iron and sulphate reduction had already run the redox potential sequence (Ottow 2011, Estop-Aragones *et al*. 2013). The reported trend of the CH_4 sources in the observed order of Carex-Körzin < Agrostis-Krielow < Carex-Krielow might be caused by different rooting depths and root biomass - in this case, *Carex* with higher root biomass rooting deeper than *Agrostis* and *Eleocharis* - as well as site-specific factors such as inundation of the Krielow sites in summer 2014 due to beaver dam construction. In addition, sulphate reduction may have hindered CH_4 production at Carex-Körzin (Pester *et al*. 2012, Segarra *et al*. 2015), where the highest sulphate and H_2S concentrations in soil pore water at 30 cm depth were measured, especially during the period of extraordinarily heavy rainfall from the end of June to the middle of August 2017 with permanently flooded conditions over several weeks.

The observed N_2O fluxes were negligible at all three sites and similarly low when compared to published studies focusing on extensive pristine, moist to wet and rewetted fens (Nykänen *et al*. 1995, Augustin *et al*. 1998, Kratz & Pfadenhauer 2001, Höper 2007, Leppelt *et al*. 2014, Günther *et al*. 2015, Kandel *et al*. 2017, Huth *et al*. 2018). The only consistent N_2O emissions occurred on all three sites during the 2015 growing season, when GWL dropped below -20 cm. GWL and WFPS were the main drivers of N_2O fluxes, with GWL alternating between -20 and -50 cm or a WFPS higher than 70 % typically being associated with N_2O production and emissions (Ruser *et al*. 2006, Goldberg *et al*. 2009). In addition, mineral N supply is low at all sites, since manuring and fertilisation ceased several years (at Krielow) or even decades (at Körzin) before this study. Together with the removal of biomass by harvest activity, this has likely led to substantial nutrient degradation and N competition between plants and microorganisms, which hampers strong N_2O formation (van Duren & Pegtel 2000). N_2O uptake has also been noticed in other studies (Höper 2007, Audet *et al*. 2014). Due to WFPS being > 80 % throughout the study period, all N_2O originated solely from denitrification (Säurich *et al*. 2019). Our results show that, despite heavily degraded peat soils and low C/N quotients, nature conservation management at high GWLs can omit most N_2O emissions related to former drainage-based peatland use.

A wide range of DOC export values from 0.8 to 46 g m^{-2} yr^{-1} can be found in the literature for near-natural and rewetted fens (Schwalm & Zeitz 2011). No significant DOC surplus from the ditches could be detected over the DOC measurement period in either study area. Therefore, DOC export was not taken into account for the measurement sites. However, with only bimonthly measurements, we might have missed peak outflow events. On the other hand, rewetting has been shown to reduce DOC pollution to natural levels within a decade (Frank *et al*. 2014), while sedimentation of DOC in the ditches due to a low velocity of water flow is also possible (Faust *et al*. 2018). Low DOC pollution in the ditches is further indicated by growth of *Stratiotes aloides* and beaver activity, which both depend on the presence of clean ditchwater (Zantout *et al*. 2011, Peterson & Schulte 2016).

CONCLUSIONS

All sites have quite similar GWLs over the entire study period, with mean annual water levels within less than 10 cm of one another. GWLs are too low in summer to be considered peat forming, and this is reflected by the net average CO_2-eq emissions of 10–17 t ha^{-1} yr^{-1}, dominated by CO_2. This is congruent with the GEST (‘Greenhouse-gas-Emission-Site-Types’) emission factor of 11 t ha^{-1} yr^{-1} for very moist meadows containing *Carex acuta* (Couwenberg *et al*. 2008) and slightly greater than the emissions of 9.6 t ha^{-1} yr^{-1} reported for a very moist meadow dominated by *Carex acuta* by Huth *et al*. (2018). However, GHG emissions are similar to other groundwater influenced peat grassland sites in a subcontinental climate with low-frequency cutting and GWL 10–20 cm below surface (e.g. Drösler *et al*. 2013, Leiber-Sauheitl *et al*. 2014, Tiemeyer *et al*. 2016, 2020). Compared to the emissions for drained grasslands (31.7 t ha^{-1} yr^{-1}; Tiemeyer *et al*. 2020), our study suggests that nature conservation management of wet meadows can mitigate between 15–20 t ha^{-1} yr^{-1} - in other words, up to two-thirds of the emissions. However, in our study the effect of summer freshwater flooding due to heavy rainfall has led to a year-to-year difference in net CO_2-eq emissions of around 3 t ha^{-1} yr^{-1} in a mown *Carex* stand, with more than half of the climate effect caused by large CH_4 emissions. Conservatively assuming CO_2-eq emissions of ~1 t ha^{-1} yr^{-1} related to CH_4 and another 2 t ha^{-1} yr^{-1} due to hampered vegetation regrowth in the year following flooding, the average net GHG emissions of the Krielow sites may actually be even lower over the longer term. Paludicultures are expected to emit less than 8 t ha^{-1} yr^{-1} (Heinrich-Böll-Stiftung 2023). However, studies focusing on net GHG emissions from wet peatland use that is paludiculture are scarce, and those that do exist usually show CO_2-eq emissions above 8 t ha^{-1} yr^{-1} (Günther *et al*. 2015, Huth *et al*. 2018, Daun *et al*. 2023), and a mitigation potential compared to drained grassland similar to the net GHG balances of our study sites (i.e., 15–20 t ha^{-1} yr^{-1}). Therefore, nature conservation management of wet fen meadows could be considered under paludiculture funding schemes and their overall climate effect will likely become even more favourable when accounting for the overall life cycle of the product (Lahtinen *et al*. 2021). In addition, nature conservation management of wet fen meadows is a well-established form of wet peatland use in Germany, extending to about 15,000 ha (Nordt & Tanneberger 2021). Therefore, it has great potential for transforming the way peatlands are used in Germany because it can be pragmatically implemented as an intermediate ‘moist’ step (Tanneberger *et al*. 2021) and, if adapted to higher GWLs (close to the ground surface) in summer, can be expected to help preserve peat even in later stages of the transformation pathway.

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AUTHOR CONTRIBUTIONS

RW set up the sites, measured the GHG fluxes and their control factors, calculated the GHG balances with VH, wrote the first draft manuscript and is the lead author. Revisions were done by VH and FJ.

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