Agricultural and Forest Meteorology 345 (2024) 109858

Contents lists available at ScienceDirect

Agricultural and Forest Meteorology

journal homepage: www.elsevier.com/locate/agrformet

Effects of water management and grassland renewal on the greenhouse gas emissions from intensively used grassland on bog peat

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A R T I C L E I N F O

Keywords:
Peatland
Greenhouse gases
Water management
Chamber method
Nitrogen
Phosphorus

A B S T R A C T

Artificial drainage is prerequisite for conventional agricultural use of peatlands, but causes high emissions of greenhouse gases (GHG), mainly carbon dioxide (CO_2). Furthermore, grassland renewal is regularly practiced to maintain high fodder quality, but might cause high emissions of nitrous oxide (N_2O). Raising water levels is necessary to reduce CO_2 emissions. Water management by subsurface irrigation (SI) and ditch blocking (DB) is thus discussed as potential compromise between maintaining intensive grassland use and reducing GHG emissions. Here, we present results of a four year study on the effects of SI and DB in combination with grassland renewal on GHG emissions from an intensively used grassland on bog peat in North-Western Germany.

The water management itself was successful and lead to average mean annual water levels of -0.33 m at the parcels with SI. This was 0.38 m higher than at the control parcels. Ditch blocking also raised the mean water levels to -0.33 m, but the parcel was dryer in summer and wetter in spring than those with SI. Despite clear effects on water levels, CO_2 and total GHG emissions were much (38 % and 31 %) higher from SI parcels than from the control parcels. CO_2 and GHG emissions of the DB parcel were similar to those of the control. Shallow ploughing increased N_2O emissions for around 1.5 years, but there was no clear effect of direct sowing. Methane emission from all parcels were low. The surprising results regarding CO_2 might be explained by an interaction of increased soil moisture in the topsoil and improved nutrient retention during periods of high soil temperatures facilitated by SI and, concurrently, by limitations of microbial activity due to dry conditions at the control parcels. Thus, results of this study do not support subsurface irrigation as a GHG mitigation measure at intensively used bog peatlands.

1. Introduction

Drained peatlands and other organic soils emit large amounts of carbon dioxide (CO_2) and nitrous oxide (N_2O) (Evans et al., 2021; IPCC, 2014; Tiemeyer et al., 2020). Thus, they cause large parts of the greenhouse gas (GHG) emissions from agriculture and agriculturally used soils in peat-rich countries such as the United Kingdom, Indonesia, the Netherlands or Germany. In Western Europe, drained peatlands are mainly used as grasslands. For example, in Germany and in the Netherlands, a major land use is fodder production for high intensity dairy farming. This often takes place in regions with a high share of peatlands, and thus, these dairy farms operate mainly on deeply drained peatlands and have in many cases little opportunity to move fodder production to mineral soils.

Rewetting, i.e. raising the groundwater level ideally to the ground surface with the goal of initiating the development of typical peatland vegetation, generally strongly decreases GHG emissions or may even initiate net carbon uptake (Wilson et al., 2016). Previous studies have shown that higher water levels in grasslands decreased CO_2 emissions (Renou-Wilson et al., 2014; Tiemeyer et al., 2016), but the investigated sites were low intensity grasslands with only one or two cuts per year and (nearly) no fertilisation. Consequently, the fodder quality from such

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https://doi.org/10.1016/j.agrformet.2023.109858
Received 29 May 2023; Received in revised form 23 September 2023; Accepted 6 December 2023
Available online 14 December 2023

sites would not be sufficient for high intensity dairy farming, as grasses such as *Juncus effusus, Holcus lanatus* or different *Carex* species have low nutritional value for dairy cows.

Conventionally, raising water levels is often done by blocking ditches (Price et al., 2003). The effect on the groundwater level strongly depends on, besides the hydraulic conductivity of the peat and the precipitation in summer, the groundwater inflow and might be therefore less effective at bog peat sites where ditches are not replenished by groundwater in summer (Kalinski et al., 2021). However, if ditches water level can be kept high in summer, there is an effect on field water levels, but they might be spatially heterogenous and too high in spring to allow trafficability. Thus, to counter soil subsidence, to reduce CO_2 emissions and to continue intensive grassland use at the same time, subsurface irrigation (SI) has long been suggested as an effective measure (Brüne, 1929; Harris et al., 1962; van den Akker et al., 2012; 2017). Subsurface irrigation systems (also termed "submerged drains") comprise shallow drainage pipes, which have their outlet below the water level of the drainage ditches. This allows both retention of precipitation and infiltration of ditch water into the peat, resulting in relatively high water levels in summer, but also enhancing drainage in winter and spring. Until recently, subsidence measurements were used to evaluate the effects of this management practice, claiming that SI may half CO_2 emissions (e.g. van den Akker et al., 2012). Recently, the first actual flux measurement study on bog peat showed that there were no differences in CO_2 emissions between grasslands with and without SI (Weideveld et al., 2021). However, in their study, mean summer water levels could only be slightly raised by, on average, 0.11 m, while mean annual water levels were partially even lower in the parcels with SI due to the drainage effect in winter. In contrast, Offermanns et al. (2023) showed a strong increase in water level and a decrease in CO_2 emissions by SI, but their one-year study was strongly influenced by the effect of grassland renewal. Therefore, the question still remains whether CO_2 emissions could be reduced when raising the water level as high as field topography, ditch geometry and trafficability allow. Approximately, trafficability with conventional agricultural machinery is not given at water levels above around 0.30 m below ground, but the exact limits strongly depend on soil moisture and quality of the sward (Blankenburg, 2015).

Besides moisture, the nutrient status might be important for CO_2 emissions. Field studies frequently showed lower CO_2 emissions from grassland on nutrient-poor than from nutrient-rich peat (IPCC, 2014; Renou-Wilson et al., 2014; Tiemeyer et al., 2016). However, both the study of Weideveld et al. (2021) as well as laboratory studies (Säurich et al., 2019a, 2019b) demonstrated high CO_2 emissions or respiration rates for bog peat. In the case of intensively used bog peatlands, the naturally low nutrient content and the low pH value of bog peat might be overridden by long-term fertilisation and secondary pedogenetic transformation of the peat. Besides nitrogen (N), especially phosphorus (P) might be a critical nutrient, as *Sphagnum* peat is poor in P and offers fewer binding sites than some fen peat or mineral soils. Indeed, P has been shown to be decisive for respiration rates in laboratory experiments (Brake et al., 1999; Säurich et al., 2019a). Overall, field data is missing on the hitherto understudied combination of moist conditions and ample nutrient supply facilitated by intensive grassland use.

Besides high CO_2 emissions, agriculturally used peatlands may also show high N_2O emissions both due to fertilization and nitrogen mineralization caused by drainage (IPCC, 2014; van Beek et al., 2011, 2010; Regina et al., 2004). N_2O is produced by a large number of processes (Butterbach-Bahl et al., 2013), but in peatlands, different denitrification pathways seem to dominate (Buchen et al., 2016; Petersen et al., 2020). Soil moisture is a crucial parameter for denitrification. It is generally thought that high N_2O emissions occur at high soil moisture (e.g. > 80 % water filled pore space, Butterbach-Bahl et al., 2013), but not under fully saturated conditions. Due to this strong dependency of N_2O formation on soil moisture on the one hand and the intensive grassland use on the other hand, raising the water level might increase N_2O emissions if the soil moisture reaches critical levels especially at fertilisation events.

Both because of an irregular soil surface caused by subsidence and the succession of unwanted species, grasslands (not only on organic soils) are regularly renovated (Kayser et al., 2018). Grassland renewal (or renovation) involves, especially when surface gradation is required, ploughing or other tillage methods. This practice might cause mineralisation of organic N and, subsequently, increased N_2O emissions (Ammann et al., 2020; Velthof et al., 2010). Buchen et al. (2017) carried out a study at a histic Gleysol and found only short-term effects. In contrast, Offermanns et al. (2023) measured extremely high N_2O emissions after renewal involving milling of the upper 10–15 cm of a grassland on bog peat. They explained the high fluxes by a combination of water levels raised by subsurface irrigation and an extreme excess of N in the soil caused by enhanced mineralisation of the peat, fertilisation and very poor growth of the newly seeded grass. Besides the strongly contrasting results of Buchen et al. (2017) and Offermanns et al. (2023), data on the effects of grassland renewal at organic soils is missing (Kayser et al., 2018).

Finally, emissions of methane (CH_4) also strongly depend on oxygen availability, which can be approximated by soil moisture or water level. Therefore, raising the water level will increase CH_4 emissions. However, several studies found that CH_4 emission strongly increase only at water levels of approximately −0.20 to −0.10 m below ground surface, i.e. at levels usually beyond trafficability for agricultural machinery (Evans et al., 2021; Levy et al., 2012; Tiemeyer et al., 2016).

With this study, we wanted to clarify the impact of water table management and grassland renewal on GHG emissions from intensively used grassland. We measured all three GHGs for four years using manually operated chambers at an experimental bog peat site under intensive grassland use in North-Western Germany. Control sites were drained by ditches and drainage pipes, while water levels were increased by either subsurface irrigation or blocked ditches. Grassland renewal encompassed shallow ploughing (GHG measurements at the control and the subsurface irrigation treatment only) and direct sowing (GHG measurements at the control treatment only). We hypothesized that

- N_2O emissions will be increased by grassland renewal, and that the effect will be stronger for shallow ploughing than for direct sowing,
- N_2O emissions will be increased by raised water levels, and that the effect of grassland renewal will be stronger for sites with increased water levels,
- CH_4 emissions of the sites with increased water levels will also increase, but not to levels relevant for the overall GHG balance, and that,
- CO_2 emissions will be reduced by raising water levels both by subsurface irrigation and blocked ditches, and that SI will be more effective than ditch blocking.

2. Material and methods

2.1. Study area

Our field site "Ipweger Moor" is situated in the "Wesermarsch", a lowland area near the river Weser (53.214° North, 8.309° East) in the federal state of Lower Saxony (Germany). While fen peat is found in the vicinity of the river, a large bog complex developed further inland. Most of the peatlands belonging to this bog complex are deeply drained and mainly used for intensive dairy farming. Intensive drainage works started in the 1960s. The climate is oceanic with long-term annual precipitation and temperature of 834 mm and 10.3 °C, respectively (meteorological stations Rastede and Bremen of the German Climate Service (DWD), 1991 to 2020).

The soil at our experimental site shows the typical horizonation of drained deep bog peat in North-Western Germany. All profiles can be classified as Ombric Drainic Fibric Histosols (Hyperorganic) (IUSS Working Group WRB, 2022) or "Normerdhochmoor" (German classification, Anonymous, 2005). Strongly decomposed peat at the peat base is overlain by weakly decomposed peat (Table 1), which started to develop during the Subatlantic, i.e. about 2600 BP (Rydin and Jeglum, 2006). Total peat depths still range from 2.00 to 2.55 m. Typically for drained bog peat, only the upper 0.10 to 0.20 m were strongly decomposed, showing a crumbly to fine-grained structure with only a very few recognizable vegetation remains. Accordingly, this horizon is very densely rooted, but some grass roots also protrude down to around 0.30 m into the weakly decomposed peat. As all measurements were carried out at the same grassland parcel, peat properties of all other measurement sites are very similar to those shown in Table 1 for the measurement site with blocked ditches (details in Table S1).

2.2. Experimental setup and agricultural management

Fig. 1 shows the layout of the study site combining the three water management approaches with the three grassland renewal treatments. All treatments were located on the same grassland parcel and thus showed an identical land use and water management history. Each stripe shown in Fig. 1 was approximately 7.5 m wide and 60 m long. Greenhouse gas (GHG) emissions were measured at six of the nine treatments: The water management control without grassland renewal (**o**riginal sward, CON_O), control with **d**irect sowing (CON_D), control with shallow **p**loughing (CON_P), subsurface irrigation without grassland renewal (SI_O), subsurface irrigation with shallow ploughing (SI_P) and ditch blocking without grassland renewal (DB_O). Each of these GHG measurement *sites* consisted of three individual *plots* (details Section 2.4).

Ideally, GHG emissions would have been meausured at all nine treatments. As this was not possible for budgetary and organisational reasons, we prioritised as follows: First, we chose the treatments without grassland renewal for all types of water management as "no renewal" is the regular situation in most years. Second, all three grassland renewal treatments at the control site were considered as this is of direct practical relevance: changes in grassland renewal could be easier adapted than changes in water management. Third, we chose the grassland renewal method which we hypothesed to have the stronger impact (shallow ploughing) in combination with the presumably more effective water management treatment (subsurface irrigation).

2.2.1. Water management

Before the onset of the experiment, the whole field was conventionally drained by PVC drainage pipes (drainage depth around 0.80 m, drainage distance 12 m) discharging into the northern and the southern ditch (Fig. 1). This drainage system remained unchanged for the control parcel, which discharges into the southern ditch. Both the ditch and the drainage channel are part of a larger water management area, which is regulated by a pumping station further downstream. Therefore, outside the experimental area, the water level in the system is kept low to ensure free drainage.

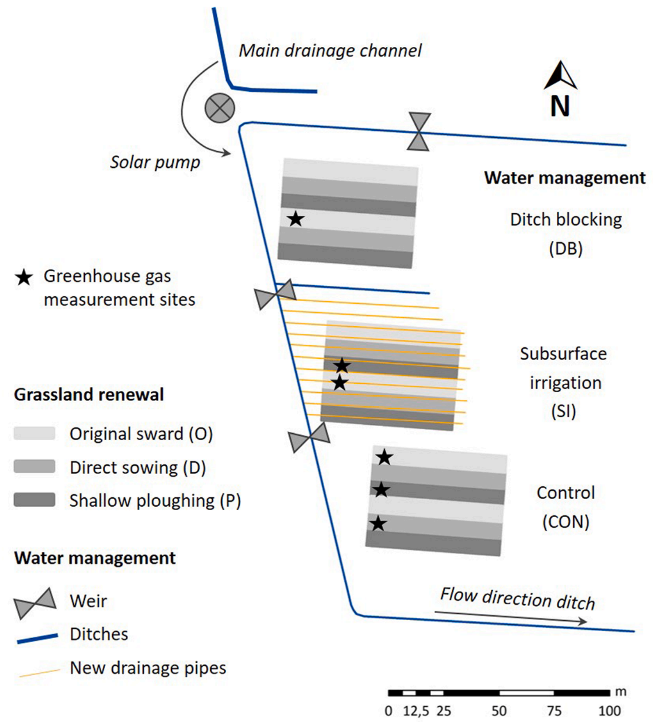

Fig. 1. Experimental setup. Drainage pipes at the control parcel are not shown.

To control the water level in the ditch bordering the parts of the field with subsurface irrigation and with ditch blocking, weirs were installed on 30.11.2016 to prevent runoff (Fig. 1). Water was pumped with a solar pump from the main drainage channel to the ditch system of the experimental site to facilitate constantly high water levels in those ditches. The pump had to be removed in winter (approximately from November to April) to avoid frost damage. Pumping started in May 2017, and during the first year of the experiment, the water management was optimized as initial problems with a blocked pump inlet, leaky weirs, clogged drainage inlets etc. precluded both the ditch and peat water levels to reach the planned levels. Such issues occurred only

Table 1
Basic soil properties of the measurement site with blocked ditches. Von Post: Degree of decomposition according to von Post (1924), BD: bulk density (mean ± standard error, $n = 6$), SOC: soil organic carbon, N_t: total nitrogen, n.d. = not determined, * $n = 18$.

Lower depth (m)	Peat type / texture	Von Post	Colour	BD (g cm^3)	SOC (g kg^{-1})	N_t (g kg^{-1})	C:N ratio	pH
−0.14	Amorphous bog peat	H10	Black	0.264 ± 0.011*	471	20	23.2	3.9
−0.30	*Sphagnum* spp. with some dwarf shrubs	H2	Yellowish brown	0.088 ± 0.001	487	7	67.7	3.6
−0.58	*Sphagnum* spp. with some *Eriophorum vaginatum* and some dwarf shrubs	H2	Dark reddish brown	0.082 ± 0.002	487	6	86.4	3.4
−0.72	*Sphagnum* spp. with *E. vaginatum* and some dwarf shrubs	H4	Black	0.083 ± 0.002	535	10	54.4	3.3
−1.25	*Sphagnum* spp. with some dwarf shrubs	H2	Rust-coloured to orange	0.066 ± 0.001	483	5	104.6	3.3
−1.60	*Sphagnum* spp. with some *Scheuchzeria palustris*	H5	Reddish brown	n.d.	n.d.	n.d.	n.d.	n.d.
−2.55	*Sphagnum* spp.	H7	Dark brown	n.d.	n.d.	n.d.	n.d.	n.d.
< −2.55	Slightly silty sand	–	dark grey	n.d.	n.d.	n.d.	n.d.	n.d.

occasionally during the following years. In the beginning of 2019, the weir levels were adjusted to allow for higher, i.e. bankfull, water levels in the ditches.

After the original drainage pipes had been taken out of operation by bending the outlets upwards, PVC drains for subsurface irrigation were installed on 20.09.2016 at a drainage distance of 5.00 m and at depth of 0.66 m below ground. To separate the treatments with ditch blocking and with subsurface irrigation, an additional ditch was dug (Fig. 1), which also cut the existing drainage pipes between the two treatments. It is important to note that the ditch blocking approach in this study strongly differs from typical ditch blocking as additional water from the main drainage channel is used to keep the water level in the blocked part of the ditch high, i.e. usually bankfull. If that had not been done, ditch water levels would have fallen to very low levels during summer due to missing groundwater inflow at this bog peat site (see Section 3.1).

2.2.2. *Agricultural management of the field and the measurement plots*

The field continued to be used for fodder production for dairy farming. As in the years before the experiment, the was no grazing. Therefore, depending on meteorological conditions, grassland management comprised 3 to 5 cuts per year and corresponding fertilisation consisting of both cattle slurry and mineral fertiliser (mainly compound fertilizer and calcium ammonium nitrate). The regular amount of nitrogen (N) and phosphorus (P) fertilisation ranged from 236 to 357 kg kg N N $ha^{-1}yr^{-1}$ and from 10 to 79 kg P ha^{-1} yr^{-1}, respectively (Table S2). This amount of fertilizer application complies to the German Fertiliser Application Ordinance. It was intended to treat all parcels equally, and thus generally all treatments received the same amount of fertilisation. However, because of the destruction of the vegetation by mice, the control treatments without renewal and with shallow ploughing received less fertilizer in 2020 (see Section 2.2.3). Furthermore, in 2019, the treatment with blocked ditches was too wet for the first application of cattle slurry in early spring. Details can be found in supplementary Table S2.

The GHG measurement plots were managed as the whole field. During field application of the fertiliser by the farmer, the plots were covered with plastic sheets. The plots were afterwards manually fertilized to control the fertilizer input. Cattle slurry was analysed for dry matter content (DIN e.V., 2001), total organic C (VDLUFA, 2011), total N (DIN e.V. 1997) and P_2O_5 (DIN e.V., 2009) by the service laboratory of the Lower Saxony Chamber of Agriculture. At each CO_2 measurement date, the vegetation height was measured and photographs of the measurement plots were taken. The plots were harvested as closely as possible to the harvest date of the field. The biomass was dried, weighed and homogenized. An aliquot was taken for C and N analysis (LECO TruMac CN, LECO Corporation, St. Joseph, USA).

2.2.3. *Grassland renewal and handling of a mice infestation*

The grassland renewal was conducted on 26.09.2016 by shallow ploughing (0.20 m) and direct sowing after mulching the upper 2 cm with a flail mulcher (Fig. S1). Afterwards, the corresponding stripes (Fig. 1) were re-sown with a grass mixture containing *Lolium perenne* (around 53%), *Festuca pratensis* (20%), *Phleum pratense* (17%) and *Poa pratensis* (10%). As grass growth was patchy due to dry conditions in autumn 2016, some repeated re-sowing was conducted in spring 2017.

In 2019, there were strong mice calamities in North-Western Germany, leading to severe losses in grassland yields at around 150,000 ha (Strotmann, 2020). This affected also the control sites, while the treatments with raised water levels were only damaged when nearly no vegetation was left at the control treatments and the surrounding grasslands. Mice damage was documented by plot photos during CO_2 measurement campaigns (Fig. 2).

As at the end of 2019 nearly no vegetation was left at the control treatments, the original grassland renewal treatments could obviously not be continued. Therefore, the treatment "no renewal" (CON_O) was left for vegetation succession, while the former shallow ploughing

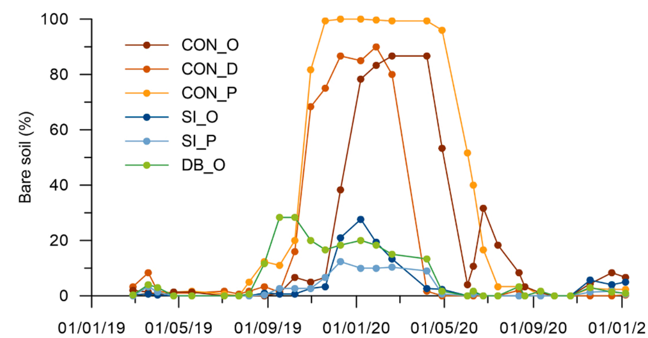

Fig. 2. Mice damage in 2019 and 2020 estimated from plot photographs. CON_O: water management control without grassland renewal (original sward), CON_D: control with direct sowing (sward transplanted from undamaged area in 2020), CON_P: control with shallow ploughing, SI_O: subsurface irrigation without grassland renewal, SI_P: subsurface irrigation with shallow ploughing and DB_O: ditch blocking without grassland renewal.

treatment (CON_P) was harrowed and re-sown as the remainder of the field. To enable GHG measurements at a freely drained control treatment with existing vegetation, the sward and the topsoil (0–0.15 m) were transplanted from the area with blocked ditches. This was done by cutting pieces fitting into the GHG measurement frames (0.75 m x 0.75 m) with serrated knives and a flexible saw and transporting each whole piece with a tarpaulin to its new location (former direct sowing treatment CON_D). To ensure comparable conditions, vegetated soil was also transplanted to the vicinity of the soil temperature sensor.

2.3. *Environmental and soil parameters*

2.3.1. *Hydro-meteorological data*

Long-term and study period data (1991 to 2020) on daily precipitation (P) and air temperature (T) data from Rastede (9 km away) and Bremen (35 km away) were provided by the German Climate Service (DWD). Soil temperature was measured at each GHG measurement site in 2 cm depth. The photosynthetically active radiation (PAR) was measured half-hourly directly at the field site (Quantumsensor Typ 6.5, F&C GmbH, Gülzow, Germany). In the case of gaps, data from a field site 15 km away was used. All PAR sensors were calibrated against a reference sensor (LI-190R Quantum Sensor, Licor, Lincoln, NE, USA).

At each site, the water level in the peat was measured in slotted PVC dip wells with a diameter of 5 cm and a Nylon mesh screening using Mini-Divers in combination with Baro-Divers for atmospheric pressure correction (Schlumberger Water Services, Delft, Netherlands). Dipwells were installed 1.25 m away from the drainage pipes, i.e. halfway between the drainage pipe and the middle between two drainage pipes. A water level (WL) below ground surface was defined as negative and ponding as positive.

2.3.2. *Soil properties*

At each site, the soil type, the horizons, the von Post value of decomposition (von Post, 1924) and the peat depths were determined at soil pits and by augering to the underlying sand. From each horizon within the first 0.3 to 1.2 m, six sample rings (244.29 cm^3) were taken for soil physical analysis. Additional grab samples also from deeper horizons were used for chemical analysis. As soil organic carbon (SOC) and nutrient stocks strongly depend on the correct and robust determination of the bulk density, 18 additional topsoil samples were taken distributed over the treatment stripes with the same sample rings from each treatment and dried at 105 °C.

Dried, sieved (<2 mm) and grinded grab samples were analysed for contents of SOC and total nitrogen (N_t) (LECO TruMac, LECO Corporation, St. Joseph, USA). The pH values were measured with a glass electrode after an extraction with a 0.01 mol L^{-1} $CaCl_2$ solution.

At each N_2O measurement date, soil samples (0–30 cm) were taken in the vicinity of the frames (mixed from 6 subsamples) for the analysis of extractable mineral nitrogen (N_{min}, VDLUFA, 2002). These samples were extracted with 0.0125 M $CaCl_2$ solution, shaken for one hour, filtered and analysed for nitrate (NO_3^-) and ammonium (NH_4^+) using a Continuous Flow Analyzer (Skalar, Breda, The Netherlands). N_{min} is the sum of extractable NO_3-N and NH_4-N expressed in kg per hectare.

From April to October 2020, topsoil samples (0–15 cm) from each treatment were taken monthly for P analysis. No samples were taken from CON_D as the transplanted measurement plots strongly differed from the surrounding from which samples could have been taken. Samples were homogenized with a knife mill (Grindomix GM 300, Retsch, Haan, Germany). Plant available phosphorus (P_{CAL}) was determined by calcium acetate lactate extraction (Schüller, 1969; VDLUFA, 2012) from field-moist samples. Extraction with a bicarbonate buffered dithionite solution was used as a proxy for reductant-soluble P (P_{BD}, Zak et al., 2008). P concentration in the extracts were measured by ICP-OES (Inductively-Coupled Plasma-Optical Emission Spectrometry, iCAP 7400 Thermo Fisher Scientific, Waltham, MA, USA).

Further, water samples from the main drainage channel have been taken on 8 occasions in 2020. Concentrations of nitrate and ammonium were determined by ion chromatography (Metrom, Filderstadt, Germany), while total P concentrations were also measured by ICP-OES.

2.4. Greenhouse gas measurements and flux calculation

Manual measurements were carried out using transparent (acrylic glass) and white opaque (PVC) non-steady state chambers (Livingston and Hutchinson, 1995) with a height of 0.5 m and an area of 0.61 m^2. Chambers were ventilated by fans and vented for pressure equilibration. When occasionally necessary in summer, transparent chambers were cooled with icepacks to limit the increase in temperature. To ensure air-tightness, chambers had rubber joints, were placed for the flux measurement on permanent frames inserted into the soil and fixed with clamps. In the case of subsurface irrigation, the center of the frames was located halfway between drainage pipes and the mid between two drainage pipes. Boardwalks were installed to prevent ebullition events or other artefacts caused by the measurement procedure. Details on the measurement protocol and the flux calculation approaches can be found in Oestmann et al. (2022).

2.4.1. Measurement of nitrous oxide and methane fluxes

To capture the immediate effects of grassland renewal, CH_4 and N_2O flux measurements started earlier than the CO_2 measurements, i.e. shortly before grassland renewals (first measurement campaign: 27.09.2016). Using the opaque chambers, CH_4 and N_2O flux measurements were conducted every two weeks and, additionally, on the 1st, 3rd and 7th day after fertilization. The chambers were closed for approximately 80 min, and five samples were collected in headspace vials after around 0, 20, 40, 60 and 80 minutes. Temperature inside the chambers was measured at each sampling occasion. Gas samples were analyzed by gas chromatography (Shimadzu, Kyoto, Japan) in the laboratory coupled with flame ionization detector (FID) for CH_4 and electron capture detector (ECD) for N_2O and CO_2.

2.4.2. Calculation of N_2O and CH_4 fluxes and annual balances

All flux calculation steps and further data analyses were carried out with R (R Core Team, 2020).

Methane and N_2O fluxes were calculated by either robust linear (RL) regression (Huber, 1981) or Hutchinson-Mosier Regression (HMR, Pedersen et al., 2010) using the R-package *gasfluxes* (Fuß, 2019) and quality criteria described in Oestmann et al. (2022). The decision between RL and HMR was based on the criterion of Hüppi et al. (2018), i.e. on trade-off between bias and uncertainty based on the accuracy of the gas chromatic measurements. CO_2 concentrations which were 10 ppm lower than the preceding value were interpreted as a sign for leakage or faulty field records. One of these two measurements in question was discarded based on the R^2 of the linear regression of CO_2. If more than one of the five concentration values were discarded, no flux was calculated. Furthermore, CH_4 and N_2O fluxes were discarded when the corresponding CO_2 flux was less than 30 % of the maximum CO_2 flux of that site and date. This was done as we could not conceive any biological reason for a strong and only episodic spatial heterogeneity of CO_2 fluxes at these microbially very active sites. Overall, 96 % of both the N_2O and the CH_4 fluxes passed all quality checks.

For the calculation of annual CH_4 and N_2O fluxes and their uncertainty, we adopted an approach similar to Günther et al. (2014): First, we derived 2000 series of flux data for each measurement year by randomly choosing one of the three fluxes for each measurement date. For each of the 2000 time series, we performed a jack-knife ("leave one out") resampling procedure, resulting in n-2 flux estimates for each annual budget, where n is the number of measurement dates. For each measurement site and year, the annual flux estimate is given by the mean of the jack-knife means and the error by the mean of all jack-knife errors.

2.4.3. Measurement of carbon dioxide exchange

CO_2 measurements were conducted for four years (first campaign 13.12.2016, last campaign 06.01.2021). Ecosystem respiration (R_{eco}) was measured with the opaque chambers and net ecosystem exchange (NEE) with the transparent chambers. Measurement campaigns took place in intervals of around three weeks during the vegetation period and four weeks in winter, respectively. Campaigns covered a diurnal cycle and started before sunrise and continued until the maximum soil temperature was reached. Depending on the season, each plot was visited at least four times with both opaque and transparent chambers. CO_2 concentrations during the chamber closure time of 90–180 s were recorded at a measurement interval of 1 s in the field with a portable infrared gas analyzer (LI-820, Licor, Lincoln, NE, USA). Additionally, air temperature and water vapor concentrations (Rotronic GmbH, Ettingen, Germany) were measured inside the chamber at the same resolution to monitor temperature development and to enable density correction of the CO_2 concentrations (Webb et al., 1980).

2.4.4. Calculation of CO_2 fluxes and balances

All flux calculation steps and further data analyses were carried out with R (R Core Team, 2020).

First, CO_2 fluxes were calculated for each single measurement by linear regression over optimal time periods, which were identified by applying a moving window between 40 and 60 s (depending on season) and choosing the linear regression equation with the highest R^2. The first 5% of each measurement were discarded. Fluxes were only accepted when R^2 was > 0.75, the temperature within the chamber changed less than 1.5 °C and PAR varied less than 10% (for transparent chambers only). 96% of all flux measurements passed all quality checks.

We adopted a campaign-based interpolation strategy for R_{eco} and gross primary production (GPP) as frequent and abrupt changes in biomass due to harvest affect fluxes, which prohibits pooling data from different campaigns during the vegetation period. This approach has frequently been chosen for grassland sites with multiple harvests (e.g. Beetz et al., 2013; Leiber-Sauheitl et al., 2014; Eickenscheidt et al., 2015). Thus, as a second step, the temperature response function of Lloyd & Taylor (1994) for R_{eco} was fitted to soil temperature in 2 cm depth and CO_2 fluxes determined with opaque chambers (Eq. (1)). Data from all three plots was pooled. In winter, the daily soil temperature amplitude was often small, and thus all data from measurement campaigns between November and February was pooled for each site, while Eq. (1) was fitted to data from individual campaigns for the remainder of the year.

$$R_{eco}(T) = R_{ref} \times \exp\left[E_0 \times \left(\frac{1}{T_{ref} - T_0} - \frac{1}{T - T_0}\right)\right] \tag{1}$$

The two fitting parameters are R_{ref} [mg CO_2-C m^{-22} h^{-1}] – respiration at the reference temperature T_{ref} (283.15 K) – and the activation-like parameter E_0 [K]. T_0 is the temperature constant for the start of biological processes (227.13 K).

In a third step, time series of R_{eco} were interpolated. For any half-hourly time step between two campaigns, two sets of R_{eco} were calculated using the parameters of the response functions of these two campaigns and half-hourly temperature data as input. At each time step, the final flux value was calculated as a weighted average of the two fluxes. As exemplarily shown for R_{eco} at time step t_i, weighting was based on the temporal distance to the campaigns:

$$R_{eco,\ i} = \frac{t_i - t_1}{t_2 - t_1} \times R_{eco,\ 2} + \frac{t_2 - t_i}{t_2 - t_1} \times R_{eco,\ 1} \tag{2}$$

where t_1 and t_2 are the date and time of the first and second campaign and $R_{eco,1}$ and $R_{eco,2}$ the ecosystem respiration estimated with the parameters of the first and the second campaign, respectively.

To calculate GPP, interpolated R_{eco} values were subtracted from fluxes determined with transparent chambers (NEE). A Michaelis-Menten type light response function depending on PAR (Eq. (3)) was then fitted to GPPP and PAR data of each measurement campaign (Michaelis and Menten, 1913; Falge et al., 2001). PAR data were corrected for the light transmittance of the transparent chambers (95%).

$$GPP(PAR) = \frac{GPP2000\ \times \alpha \times\ \text{PAR}}{GPP2000 + \alpha\ \times\ \text{PAR} - \frac{GPP2000}{2000\ \mu mol\ m^{-2} s^{-1}}\ \times\ \text{PAR}} \tag{3}$$

Here, the two fitting parameters are GPP_{2000}, i.e. the rate of carbon fixation at a PAR of 2000 µmol m^{-2} s^{-1} [mg CO_2-C m^{-2} h^{-1}], und α, i.e. the initial slope of the light response curve [mg CO_2-C m^{-2} h^{-1}/µmol m^{-2} s^{-1}].

Time series of GPP were interpolated in the same way as those of R_{eco}. In case of harvest, the GPP function of the previous campaign was used up to the harvest date and the response functions of the campaign after the harvest date from harvest onwards. Harvest was accounted for by setting GPP to zero, and thus GPP for the period between the harvest and the subsequent campaign is a weighted average of zero and the GPP calculated from the response function of that campaign. NEE was finally calculated as the sum of R_{eco} and GPP and summed up to annual values.

Uncertainty of the annual values of NEE, R_{eco} and GPP was assessed by bootstrapping Eqs. (1) and (2) (2000 model runs) and thus recalculating the annual sums 2000 times. The resulting standard error of NEE is then used when calculating the uncertainty of the annual GHG budgets (Section 2.4.5). This approach can only asses the uncertainty caused by an imperfect fit of the R_{eco} and GPP functions, but not by the interpolation between measurement campaigns.

2.4.5. Greenhouse gas balances

The net ecosystem carbon budget (NECB, all terms in [t C ha^{-1} yr^{-1}]) is given by Eq. (4):

$$\text{NECB}\left[\text{t Cha}^{-1}\text{yr}^{-1}\right] = \text{NEE} - \text{C}_{\text{in}} + \text{C}_{\text{ex}} + \text{CH}_4 - \text{C} \tag{4}$$

C_{in} and C_{ex} are the import of C by organic fertilizer and the export via harvest. As we aim to quantify the change in SOC, the harvested biomass is assumed to be released as CO_2 elsewhere in the same year and is thus a loss from the system. This approach has also been applied for deriving default emission factors (IPCC, 2014). We follow the atmospheric sign convention, i.e. positive fluxes and budgets indicate a loss of carbon and GHGs from the soil to the atmosphere. Our NECB does not account for losses of dissolved organic carbon (DOC), which are assumed to make a minor contribution to the total C budget of our site. Annual greenhouse gas budgets are calculated by Eq. (5):

$$\text{GHG budget}\left[\text{t CO}_2\text{-eq. ha}^{-1}\text{yr}^{-1}\right] = \text{NEE} - \text{C}_{\text{in}} + \text{C}_{\text{ex}} + 28\text{CH}_4 + 265\text{N}_2\text{O} \tag{5}$$

In Eq. (5), NEE, C_{in} and C_{ex} are given in [t CO_2 ha^{-1} yr^{-1}]. We used global warming potentials (GWP) of 28 for CH_4 and 265 for N_2O (100 years, Myhre et al., 2013). The uncertainties of the NECB and the GHG budget were determined by error propagation.

2.5. Data analysis

2.5.1. Estimation of the heterotrophic respiration (R_h)

Ecosystem respiration (R_{eco}) is composed of heterotrophic respiration (R_h) and autotrophic respiration (R_a). Thus, any differences in R_{eco} or in the temperature dependency of R_{eco} includes effects of differences in biomass. For analysing e.g. the effects of the water level on SOC losses, the R_h is thus more informative than R_{eco}. The autotrophic respiration of the plants strongly correlates with gross primary production (GPP) and can be approximated by GPP/2 (Gilmanov et al., 2007; Luyssaert et al., 2009). Accordingly, R_h can be estimated as:

$$\text{R}_\text{h} = \text{R}_\text{eco} - 0.5 * \text{GPP} \tag{6}$$

To be able to analyse possible differences between treatments, R_h was calculated only for the dates of the measurement campaigns, which minimizes interpolation errors. Then, the Lloyd-Taylor model (Eq. (1)) was fitted separately to the data of each year and treatment.

2.5.2. Comparison of CO_2 emissions to previous studies

To compare the CO_2 emission reported here to previous studies on grasslands on bog peat (Offermanns et al., 2023; Renou-Wilson et al., 2014, 2016; Tiemeyer et al., 2016; Weideveld et al., 2021), we fitted the Gompertz equation (Eq. (7)) which was also used by Tiemeyer et al. (2020) to all data from the above-mentioned references excluding treatments with subsurface irrigation (Offermanns et al., 2023; Weideveld et al., 2021).

$$CO_2\ (WL) = \ CO_{2_{min}} +\ CO_{2_{diff}}\ e^{-a\ e^{b\ WL}} \tag{7}$$

If there were multiple years of the data, we used mean values of both annual CO_2 emissions and mean annual water levels (WL). In Eq. (7), $CO_{2\ min}$ is the lower asymptote, $CO2_{diff}$ the difference between upper and lower asymptote, while a and b are fitting parameters related to the displacement along the x-axis and the growth rate, respectively. Due to the limited data set, $CO_{2\ min}$ was set to -3.4 t CO_2 ha^{-1} yr^{-1} (Tiemeyer et al., 2020). Uncertainties of the function fit have been estimated by bootstrapping ($n = 2000$) using the package *nlstools* (Baty et al., 2015). Using this approach, we are able evaluate whether our data lies within the expected range of emissions given certain mean annual water level.

2.5.3. Statistical analysis of GHG fluxes and budgets

2.5.3.1. Effect of grassland renewal on nitrous oxide fluxes. While annual sums of N_2O emissions (1st January to 31st of December) were required to calculate GHG balances (Section 2.4.5), cumulative N_2O emissions were used to evaluate the effect of grassland renewal. This was done as we were interested in how long potential effects of grassland renewal lasted and as CO_2 measurements started only in the beginning of 2016. Cumulative N_2O emissions were calculated starting at the first measurement date, i.e. directly after grassland renewal. For each of the 2000 series of flux data, daily means of the according jack-knife realizations were calculated. These 2000 time series were cumulated, and the mean as well as the 2.5 and 97.5 percentiles of the cumulated N_2O fluxes were calculated for the whole time series (26.09.2016 – 31.12.2020). As N_2O data was not available for all combinations of grassland renewal and water level management, we could compare the following treatments: Control without grassland renewal vs. control with direct sowing, control without grassland renewal vs. control with shallow ploughing and subsurface irrigation without grassland renewal vs. subsurface irrigation with shallow ploughing.

2.5.3.2. Effect of water management on CO_2 and GHG budgets. To summarize the effect of the two tested water management approaches on CO_2 and GHG budgets, we first calculated for each year the difference for corresponding grassland renewal treatments (control without grassland renewal vs. subsurface irrigation without grassland renewal, control without grassland renewal vs. ditch blocking without grassland renewal and control with shallow ploughing vs. subsurface irrigation with shallow ploughing). Secondly, to make use of all data, we compared annual differences of the means of all grassland renewal treatments. Mean values and standard errors of the differences are reported.

For an overall statistical evaluation of the water management effects, the unbalanced study design had to be taken into account. Therefore, we used a generalized least squares (*gls*) model with the varIdent variance structure of the R package "*nlme*" (Pinheiro et al., 2020). This variance structure allows for handling specific variances of sites, grassland renewal treatments and measurement years. Using maximum likelihood (ML) estimation, we tested the appropriateness of different variance structures by comparing the AIC (Akaike Information Criterion) and additionally visually checked the residuals. Furthermore, residuals were checked for autocorrelation both visually and by adding a first-order autocorrelation structure (Zuur et al., 2009). Probably due to the short duration of the data set (4 years), no autocorrelation was found. Again following Zuur et al. (2009), the models were refit using REML (restricted maximum likelihood). *P*-values were computed with the Tukey's honest significant differences test ($p = 0.05$) and adjusted with the Bonferroni correction using the R package "*multcomp*" (Hothorn et al., 2008).

3. Results

3.1. Weather conditions and effects of water management on water levels

During the first experimental year (2017), pumping only started in May, and the water management infrastructure was still being optimized. Therefore – and not because of the comparatively high precipitation – differences in mean annual water level (WL) between the treatments with free drainage and especially with subsurface irrigation remained rather small (Fig. 3, Table 2).

After a wet spring, the summer of 2018 was characterised by a strong drought over Western Europe. As a consequence, summer precipitation was only 55% of the long-term average of 439 mm. Despite this, WLs could be strongly raised by water management (Fig. 3). As the drought persisted into late autumn, the WL at the control treatments did not reach the ground surface in winter 2018/19 and neither in winter 2019/20.

2019 and 2020 both saw average precipitation sums. As the weir levels were increased in early 2019, the WLs at the subsurface irrigation treatment could be raised even higher than during the preceding years (Table 2). Overall, average mean annual WL at the treatments with subsurface irrigation were the same as the parcels with blocked ditches (-0.33 m), but the intra-annual dynamics differed: At the parcel with blocked ditches, WLs in spring tended to be higher, but fell to lower values during summer (Fig. 3).

3.2. Biomass development

Biomass development and dry yield from the experimental plots were similar in 2017 and 2018 despite the drought in 2018 (Fig. 4). Regarding grassland renewal, the direct sowing resulted in slightly higher yields than the shallow ploughing, which was on a similar level as the original sward.

In 2019, a massive infestation with mice ultimately destroyed the sward of the control treatments, while both parcels with subsurface irrigation and blocked ditches were clearly less affected (Fig. 2). In combination with the lasting impact of the drought in 2018, the yields of all treatments were the lowest of all four measurement years, and extremely low for the control treatment. The yields of the control treatments in 2020 were still influences by the destruction of the sward. The re-sown treatment (CON_P) developed only slowly in the beginning of the year, resulting in a neglectable yield at the first cut. The following cuts, however, showed similar yields as the water management treatments. The "succession treatment" (CON_O) showed very low yields all year. The transplantation of the sward (CON_D) worked very well and resulted in similar yields as at DB_O from where the sward was transplanted. During all years, yields from the treatments with subsurface irrigation were higher than from the other treatments.

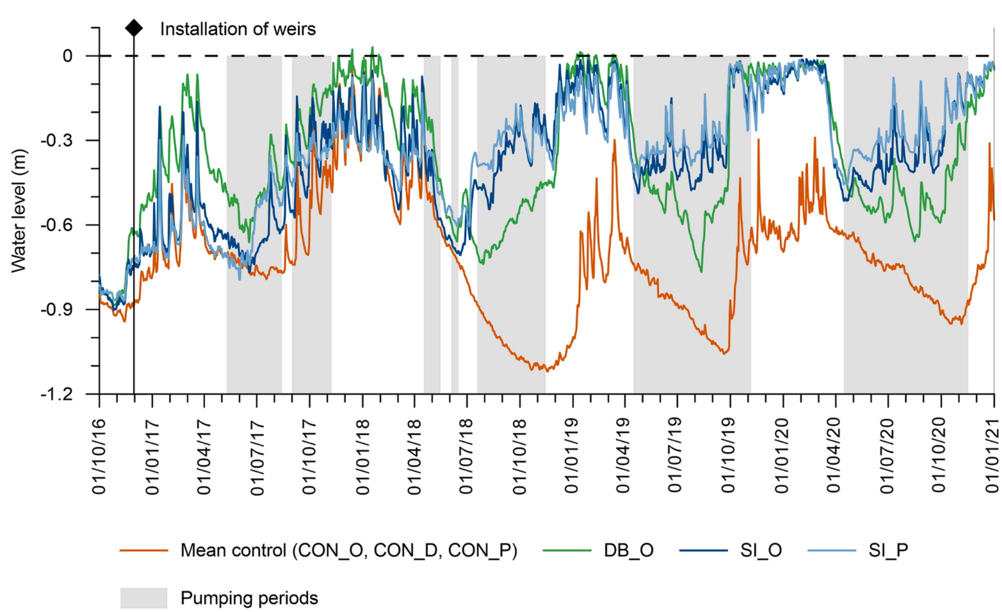

Fig. 3. Water management and daily water levels. Mean control: Mean value of the three treatments with free drainage, DB_O: ditch blocking with the original sward, SI_O: subsurface irrigation with the original sward and SI_P: subsurface irrigation d with ploughing.

Table 2

Annual and summer precipitation (P, DWD station Rastede), mean annual and temperature (T, DWD station Bremen), mean annual and summer (1st May to 31st October) water level (WL). Increase in WL: difference to the mean of the three control treatments with free drainage. CON_O: water management control without grassland renewal (original sward), CON_D: control with direct sowing, CON_P: control with shallow ploughing, SI_O: subsurface irrigation without grassland renewal, SI_P: subsurface irrigation with shallow ploughing and DB_O: ditch blocking without grassland renewal. Mean YEAR and mean SUMMER: Mean and standard deviation of the four study years.

	P	T	Water level (WL) (m)						Increase in WL (m)		
	(mm)	(°C)	CON_O	CON_D	CON_P	SI_O	SI_P	DB_O	SI_O	SI_P	DB_O
2017 year	961	10.2	−0.64	−0.51	−0.62	−0.49	−0.50	−0.31	0.10	0.09	0.28
2017 summer	516	13.6	−0.70	−0.64	−0.69	−0.54	−0.51	−0.39	0.13	0.17	0.28
2018 year	578	11.0	−0.77	−0.73	−0.75	−0.37	−0.35	−0.39	0.38	0.40	0.36
2018 summer	242	14.6	−0.89	−0.88	−0.89	−0.46	−0.39	−0.58	0.43	0.49	0.31
2019 year	854	10.9	−0.79	−0.81	−0.74	−0.24	−0.22	−0.27	0.54	0.56	0.51
2019 summer	465	14.0	−0.88	−0.91	−0.86	−0.32	−0.27	−0.46	0.57	0.62	0.43
2020 year	845	11.1	−0.74	−0.70	−0.72	−0.25	−0.21	−0.33	0.47	0.51	0.38
2020 summer	404	13.7	−0.80	−0.80	−0.80	−0.35	−0.28	−0.52	0.45	0.52	0.28
Mean YEAR	810 ± 163	10.8 ± 0.4	−0.73 ± 0.07	−0.69 ± 0.12	−0.71 ± 0.06	−0.34 ± 0.12	−0.32 ± 0.14	−0.33 ± 0.05	0.37 ± 0.19	0.39 ± 0.32	0.38 ± 0.09
Mean SUMMER	407 ± 119	14.0 ± 0.5	−0.82 ± 0.09	−0.81 ± 0.12	−0.81 ± 0.09	−0.42 ± 0.10	−0.36 ± 0.11	−0.49 ± 0.08	0.39 ± 0.18	0.45 ± 0.20	0.32 ± 0.07

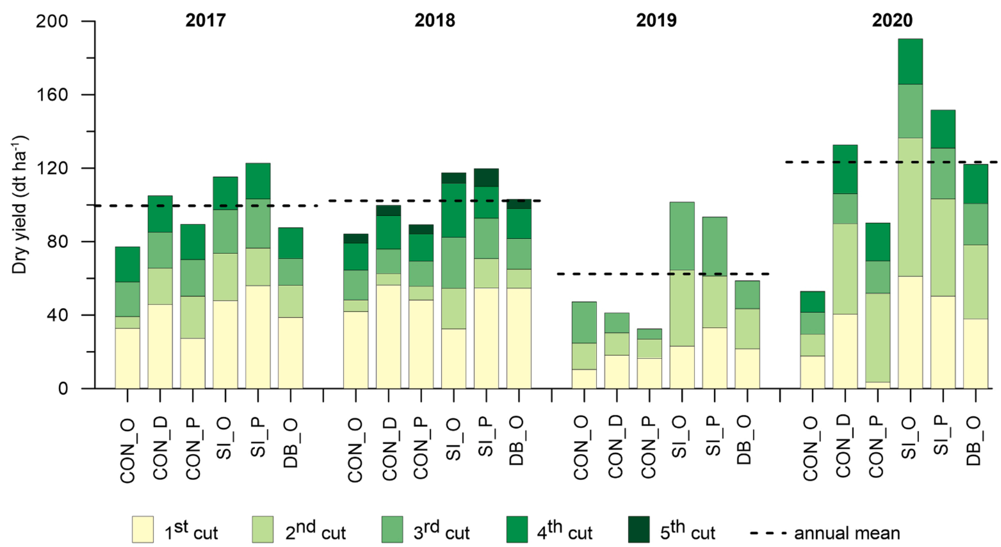

Fig. 4. Grass yield (dry matter) at the greenhouse gas measurement plots. CON_O: water management control without grassland renewal (original sward), CON_D: control with direct sowing, CON_P: control with shallow ploughing, DB_O: ditch blocking without grassland renewal, SI_O: subsurface irrigation without grassland renewal and SI_P: subsurface irrigation with shallow ploughing.

3.3. Greenhouse gas exchange

3.3.1. Nitrous oxide and methane exchange

Table 3 shows the annual nitrous oxide (N_2O) and methane emissions (CH_4) of all measurement sites.

Nitrous oxide fluxes were generally high, but varied strongly between treatments and years. On average, all three sites with water management showed lower N_2O emissions than the control (Table 3, average over four years: CON: 15.8 kg N_2O-N ha^{-1} yr^{-1}, SI: 9.4 kg N_2O-N ha^{-1} yr^{-1}, DB: 6.6 kg N_2O-N ha^{-1} yr^{-1}). Also, when comparing annual values of the same grassland renewal treatments, sites with water management showed lower N_2O emissions with the exception of 2019. During this year, the N_2O emission of SI_O and DB_O exceeded those of CON_O, and those of SI_P exceeded those of CON_P. Overall, annual N_2O emissions correlated well with mean annual N_{min} stocks; especially for the control and SI treatments (Fig. S2). N_{min} stocks tended to be higher at the control site, and the same N_{min} stock resulted in much higher N_2O fluxes at the control than at the water management treatments.

Methane fluxes were generally low. Control sites showed, on average, a minor CH_4 uptake of -0.2 kg CH_4 ha^{-1} yr^{-1}, while slightly elevated CH_4 fluxes occurred at the sites with raised water levels (SI: 38.83 kg CH_4 ha^{-1} yr^{-1}, DB: 23.3 kg CH_4 ha^{-1} yr^{-1}). Only during the last two years, higher CH_4 fluxes occurred at the sites with raised water levels (Fig. S3), i.e. at DB_O in 2020 at a mean annual WL of -0.33 m, and at SI_O and SI_P in 2019 and 2020 (mean annual water levels between -0.21 and -0.25 m).

Fig. 5 shows the cumulative N_2O emissions starting from the date of the grassland renewal (26.09.2016) by shallow ploughing. The autumn of 2016 was dry, and thus emissions only started after the water level rose. This was especially evident for the shallow ploughing treatments. Over the first year after grassland renewal, N_2O emissions of the treatments with shallow ploughing were 187 % (control without water management) and 221 % (SI) of those without renewal. In the case of direct sowing, cumulative N_2O emissions amounted to only 39 % of the control (not shown). While N_2O emission during autumn and winter were similar, the higher emissions of the site CON_O were caused by the reaction to the last two fertiliser applications in 2017.

After two years, the effects of grassland renewal had declined. Still, N_2O emissions of the treatments with shallow ploughing were 117 % (control without water management) and 145 % (SI) of those without renewal. However, considering the large uncertainty of the cumulative fluxes, these differences were not significant. N_2O emissions from the

Table 3
Annual balances (1st January to 31st December) of ecosystem respiration (R_{eco}), gross primary production (GPP), net ecosystem exchange (NEE), carbon input by cattle slurry (C_{in}), carbon export by harvest (C_{ex}), nitrous oxide (N_2O), methane (CH_4) as well as the net ecosystem carbon balance (NECB) and the greenhouse gas balance (GHG) using global warming potential (GWP100) of 265 for N_2O and 28 for CH_4, respectively. CON_O: water management control without grassland renewal (original sward), CON_D: control with direct sowing, CON_P: control with shallow ploughing, SI_O: subsurface irrigation without grassland renewal, SI_P: subsurface irrigation with shallow ploughing and DB_O: ditch blocking without grassland renewal.

Treatment	Year	R_{eco} (t CO_2-C ha^{-1} yr^{-1})	GPP (t CO_2-C ha^{-1} yr^{-1})	NEE (t CO_2-C ha^{-1} yr^{-1})	C_{in} (t CO_2-C ha^{-1} yr^{-1})	C_{ex} (t CO_2-C ha^{-1} yr^{-1})	CO_2 (t CO_2 ha^{-1} yr^{-1})	N_2O (kg N_2O-N ha^{-1} yr^{-1})	CH_4 (kg CH_4 ha^{-1} yr^{-1})	NECB (t C ha^{-1} yr^{-1})	GHG (t CO_2-Eq. ha^{-1} yr^{-1})
CON_O	2017	23.6 ± 0.3	−15.9 ± 0.9	7.7 ± 0.9	−0.4	3.4 ± 0.3	39.1 ± 3.6	22.2 ± 15.4	0.1 ± 1.1	10.7 ± 1.0	48.4 ± 7.4
CON_O	2018	25.1 ± 0.8	−16.2 ± 0.5	8.9 ± 0.7	−1.5	3.9 ± 0.4	41.7 ± 3.0	33.2 ± 16.1	2.0 ± 2.7	11.4 ± 0.8	55.5 ± 7.4
CON_O	2019	25.1 ± 0.5	−14.9 ± 0.4	10.2 ± 0.5	−1.1	2.3 ± 0.1	41.5 ± 1.9	4.4 ± 1.6	−1.3 ± 0.6	11.3 ± 0.5	43.3 ± 2.0
CON_O	2020	22 ± 0.4	−12.2 ± 0.5	9.7 ± 0.5	−0.7	2.4 ± 0.3	42.2 ± 2.3	7.9 ± 1.5	−3.2 ± 0.4	11.5 ± 0.6	45.4 ± 2.4
CON_D	2017	27.3 ± 0.4	−19.6 ± 0.8	7.7 ± 0.9	−0.4	4.6 ± 0.3	43.6 ± 3.4	9.5 ± 6.6	−0.6 ± 0.7	11.9 ± 0.9	47.6 ± 4.4
CON_D	2018	25.5 ± 0.9	−17.8 ± 0.4	7.7 ± 0.7	−1.5	4.7 ± 0.3	39.8 ± 3.0	15.2 ± 10.6	2.2 ± 4.0	10.8 ± 0.8	46.2 ± 5.3
CON_D	2019	25.4 ± 0.6	−14.4 ± 0.4	11.0 ± 0.7	−1.1	2.0 ± 0.1	43.4 ± 2.5	7.9 ± 2.1	−1.4 ± 0.4	11.8 ± 0.7	46.7 ± 2.6
CON_D	2020	29.5 ± 0.5	−20.3 ± 0.5	9.2 ± 0.6	−1.2	6.1 ± 0.5	51.6 ± 2.7	12.2 ± 3.5	−3.0 ± 0.5	14.1 ± 0.7	56.6 ± 3.1
CON_P	2017	28.2 ± 0.5	−18.7 ± 1.1	9.5 ± 1.1	−0.4	3.9 ± 0.2	47.5 ± 4.1	45.8 ± 15.6	0.1 ± 0.9	12.9 ± 1.1	66.6 ± 7.7
CON_P	2018	24.8 ± 0.5	−15.8 ± 0.5	9.0 ± 0.6	−1.5	4.1 ± 0.3	42.7 ± 2.5	19.2 ± 17.2	2.9 ± 4.1	11.7 ± 0.7	50.8 ± 7.6
CON_P	2019	23.1 ± 0.5	−13.7 ± 0.4	9.4 ± 0.7	−1.1	1.5 ± 0.0	36.2 ± 2.4	2.7 ± 0.5	0.1 ± 0.5	9.9 ± 0.7	37.3 ± 2.4
CON_P	2020	22.6 ± 0.4	−15.8 ± 0.4	6.8 ± 0.4	−0.7	4.2 ± 0.4	37.6 ± 2.1	9.7 ± 1.8	−0.4 ± 0.3	10.3 ± 0.6	41.6 ± 2.2
SI_O	2017	30.5 ± 0.5	−18.8 ± 0.5	11.6 ± 0.6	−0.4	5.0 ± 0.1	59.3 ± 2.2	8.0 ± 5.8	−1.8 ± 0.5	16.2 ± 0.6	62.6 ± 3.3
SI_O	2018	26.4 ± 0.5	−19.4 ± 0.6	7.0 ± 0.8	−1.6	6.5 ± 0.8	43.8 ± 4.2	13.2 ± 12.3	4.4 ± 4.0	12.0 ± 1.1	49.4 ± 6.6
SI_O	2019	32.5 ± 1.4	−21.6 ± 0.8	10.9 ± 1.2	−1.1	4.7 ± 0.1	53.1 ± 4.5	8.0 ± 2.6	68.5 ± 26.6	14.5 ± 1.2	58.4 ± 4.6
SI_O	2020	33.6 ± 0.5	−22.9 ± 0.5	10.7 ± 0.5	−1.2	8.7 ± 0.2	66.8 ± 2.1	6.6 ± 2.8	148.3 ± 72.1	18.3 ± 0.6	73.7 ± 2.4
SI_P	2017	28.3 ± 0.5	−17.3 ± 0.3	11.0 ± 0.5	−0.4	5.3 ± 0.5	58.2 ± 2.5	16.7 ± 2.7	2.2 ± 1.4	15.9 ± 0.7	65.3 ± 2.8
SI_P	2018	28.3 ± 0.5	−20.6 ± 0.4	7.7 ± 0.5	−1.6	5.5 ± 0.2	42.9 ± 2.1	8.8 ± 6.8	1.4 ± 2.3	11.7 ± 0.6	46.6 ± 3.5
SI_P	2019	33.9 ± 0.5	−19.4 ± 0.7	14.5 ± 0.8	−1.1	4.3 ± 0.1	65.1 ± 2.8	10.1 ± 4.1	46.9 ± 33.6	17.8 ± 0.8	70.6 ± 3.3
SI_P	2020	33.1 ± 0.4	−20.7 ± 0.5	12.4 ± 0.5	−1.2	6.9 ± 0.2	66.6 ± 2.0	3.5 ± 1.5	40.4 ± 18.7	18.2 ± 0.6	69.2 ± 2.1
DB_O	2017	27.7 ± 0.4	−17.4 ± 0.4	10.3 ± 0.5	−0.4	3.8 ± 0.2	50.2 ± 2.0	8.9 ± 8.2	4.2 ± 2.4	13.7 ± 0.5	54.1 ± 3.9
DB_O	2018	24.2 ± 0.4	−20.2 ± 0.4	4.0 ± 0.4	−1.6	4.8 ± 0.5	26.5 ± 2.3	2.4 ± 1.4	8.9 ± 4.9	7.2 ± 0.6	27.8 ± 2.3
DB_O	2019	29.3 ± 0.8	−17.3 ± 0.5	12.0 ± 0.9	−0.6	2.8 ± 0.3	52.1 ± 3.4	11.7 ± 7.5	6.3 ± 3.9	14.2 ± 0.9	57.2 ± 4.6
DB_O	2020	29 ± 0.5	−21.1 ± 0.5	7.9 ± 0.7	−1.2	5.6 ± 0.2	45.2 ± 2.6	3.1 ± 1.3	73.7 ± 53.7	12.4 ± 0.7	48.6 ± 2.7

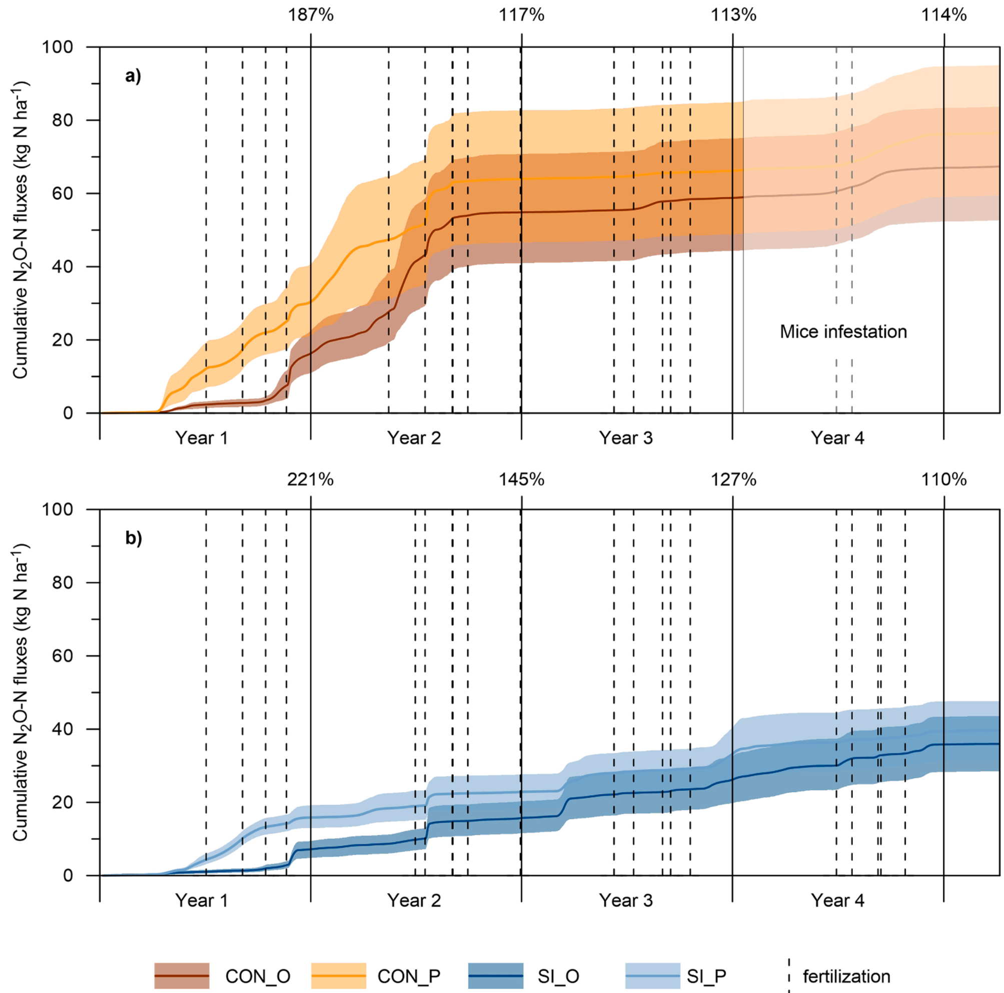

Fig. 5. Cumulative nitrous oxide (N_2O) emissions starting from the date of grassland renewal, a) water management control without grassland renewal (original sward) (CON_O) and with shallow ploughing (CON_P), b) subsurface irrigation without grassland renewal (original sward) (SI_O) and with shallow ploughing (SI_P). Percentages given at the top of the figures: Total N_2O emissions of the renewal treatments relative to the original swards. Uncertainty bands: 2.5 and 97.5 percentiles of 2000 bootstrap runs for cumulative fluxes.

direct sowing treatment remained much lower than those of the control. Due to the severe damage by mice, and the subsequent handling of the field site (Section 2.2.3), any differences or similarities in N_2O fluxes of the control sites without water management cannot be assigned to the initial renewal treatments from the end of 2019 onwards.

3.3.2. *Exchange of carbon dioxide*

The CO_2 emissions of the control treatments remained rather constant from year to year despite drought and mice damage (Table 2) as shown by standard deviations of the annual mean of only 1.3 to 5.2 t CO_2 ha^{-1} yr^{-1}. Even the transplanted topsoil of CON_D showed in 2020 higher, but still similar CO_2 emissions (51.6 t CO_2 ha^{-1} yr^{-1}) as during the previous years (39.8 to 43.6 t CO_2 ha^{-1} yr^{-1}).

In this study, we did not measure losses of dissolved organic carbon (DOC) or other fluvial C losses, which constitute further losses of C from the soil. The standard IPCC emission factor for CO_2 emissions from DOC is 0.31 t C ha^{-1} yr^{-1} (IPCC, 2014), while Frank (2016) determined losses of 0.32 t C ha^{-1} yr^{-1} from an intensively used grassland on bog peat similar to our site. These values would correspond to 2–3 % of the NECB determined for the control sites. Ditch blocking and SI might both change DOC concentrations and water balance components, but so far, data is lacking. However, due to the very high CO_2 emissions (Table 3), a high share of DOC to the NECB seems unlikely at this site even if raising the water level would increase DOC losses.

The water management was still being optimized in 2017 and thus the water level did not yet reach the envisaged values. Still, CO_2 emissions from all water management treatments exceeded those of the control treatments (Table 3). In 2018, the water level at the SI parcels

Table 4
Mean value ± standard error of the differences between the annual CO_2 and GHG emissions from the water management treatments (subsurface irrigation: SI and ditch blocking: DB) and the control treatments.

Basis for comparison	CO_2 emissions (t CO_2 ha^{-1} yr^{-1}) SI	DB	GHG emissions (t $CO_{2\text{-Eq}}$ ha^{-1} yr^{-1}) SI	DB
Mean value of all control treatments per year	+35 ± 11 %	+3 ± 14 %	+28 ± 14 %	−2 ± 16 %
Control treatment with the same grassland renewal approach	+40 ± 11 %	+6 ± 15 %	+33 ± 13 %	0 ± 18 %
Mean	**+38 ± 11 %**	**+5 ± 14 %**	**+31 ± 14 %**	**−1 ± 17 %**

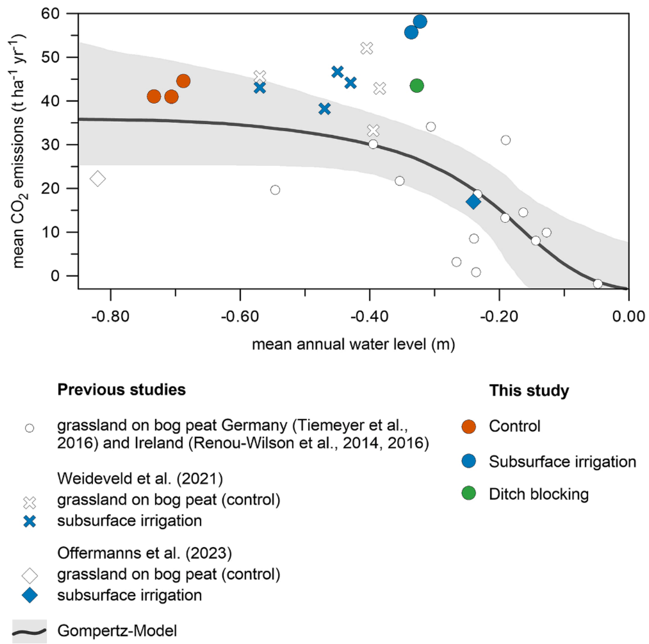

Fig. 6. Mean annual water levels and CO_2 emissions of this study in comparison to previous data on grassland on bog peat in Germany (Tiemeyer et al., 2016; Offermanns et al., 2023), in Ireland (Renou-Wilson et al., 2014; 2016) and in the Netherlands (Weideveld et al., 2021). The Gompertz model was only fitted to previous data excluding sites with subsurface irrigation. Uncertainty bands were derived by 2000 bootstrap runs.

could be raised stronger both in absolute terms and compared to the control (Table 2). During this year, CO_2 emissions of the parcel with blocked ditches (26.5 t CO_2 ha^{-1} yr^{-1}) were lower than the mean of the control treatments (41.4 t CO_2 ha^{-1} yr^{-1}). CO_2 emissions from the SI, however, were clearly lower than in the preceding year, but still higher than those from the controls (means 2017 and 2018: 58.8 and 43.4 t CO_2 ha^{-1} yr^{-1}, respectively). A further adjustment of the weir levels at the SI parcels led to mean summer and annual water levels of −0.30 and −0.23 m, respectively, both in 2019 and 2020. Despite this, CO_2 emissions from SI treatments were 2019 again at the same level as 2017 or even higher. Similarly, CO_2 emissions from the treatment with blocked ditches in 2019 were again as high as in 2017. These results were confirmed in 2020.

Overall, mean (± standard deviation of the site means) CO_2 emissions of the treatments with SI (57.0 ± 1.7 t CO_2 ha^{-1} yr^{-1}) were both significantly higher than those of the control (42.2 ± 2.1 t CO_2 ha^{-1} yr^{-1}, $p < 0.001$) and the ditch blocking (43.5 t CO_2 ha^{-1} yr^{-1}, $p < 0.01$). The slight difference between control and DB was only weakly significant ($p < 0.05$). The increase of the CO_2 emissions by SI was also both evident when comparing all data and when comparing only parcels with the same grassland renewal treatment, i.e. SI_O with CON_O on the one hand and SI_P with CON_P on the other hand (Table 4). Overall, CO_2 emissions from the treatments with subsurface irrigation were 38 ± 11% higher than those from the control treatment (Table 4). In the case of the ditch blocking treatment, there was no clear overall effect on the CO_2 emissions (increase by 5 ± 14%, Table 4).

Fig. 6 compares our results to previous studies on grassland on bog peat in Germany (Tiemeyer et al., 2020), Ireland (Renou-Wilson et al., 2014; 2016) and in the Netherlands (van Weideveld et al., 2021). So far, there is only data for two intensively used grasslands on bog peat in Germany, which both show lower CO_2 emissions than control treatments at our site (Beetz et al., 2013; Offermanns et al., 2023; Tiemeyer et al., 2020). However, data from the Netherlands (Weideveld et al., 2021) were at least as high as our results, which are thus within the expected range. All nutrient-poor Irish sites showed comparatively low CO_2 emissions and were, in contrast, used for low-intensity grazing (Renou-Wilson et al., 2014; 2016).

CO_2 emissions from the parcels with subsurface irrigation were higher than those from all previous studies. Both in the case of 4-year averages (Fig. 6) and individual years, there were higher CO_2 emissions at higher WLs both on the basis of the whole year and on summer data. In contrast, Weideveld et al. (2021) could not find any clear difference in CO_2 emissions, while Offermanns et al. (2023) meausured lower values at their site.

The dependence of the estimated heterotrophic respiration (R_h) on soil temperature differed between years and treatments (Fig. S4). While

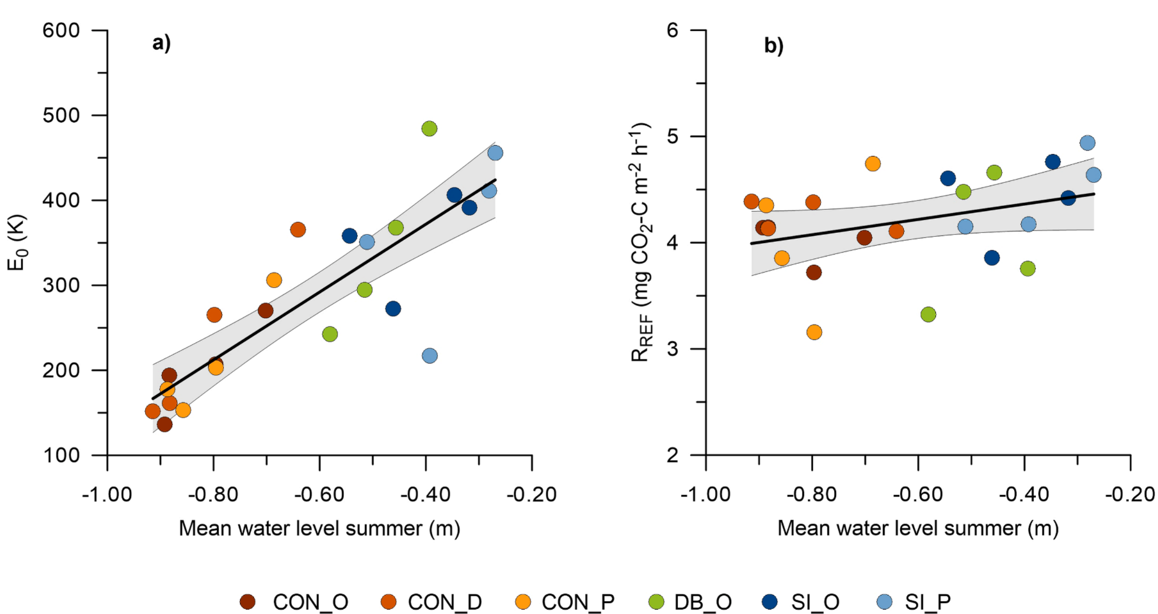

Fig. 7. Relationship between mean water level in summer and the parameters of the Lloyd-Taylor model fitted to soil temperature and estimated heterotrophic respiration data of every measurement year: a) activation-like parameter E_0 and b) respiration at the reference temperature (R_{REF}). CON_O: water management control without grassland renewal (original sward), CON_D: control with direct sowing, CON_P: control with shallow ploughing, DB_O: ditch blocking without grassland renewal, SI_O: subsurface irrigation without grassland renewal and SI_P: subsurface irrigation with shallow ploughing.

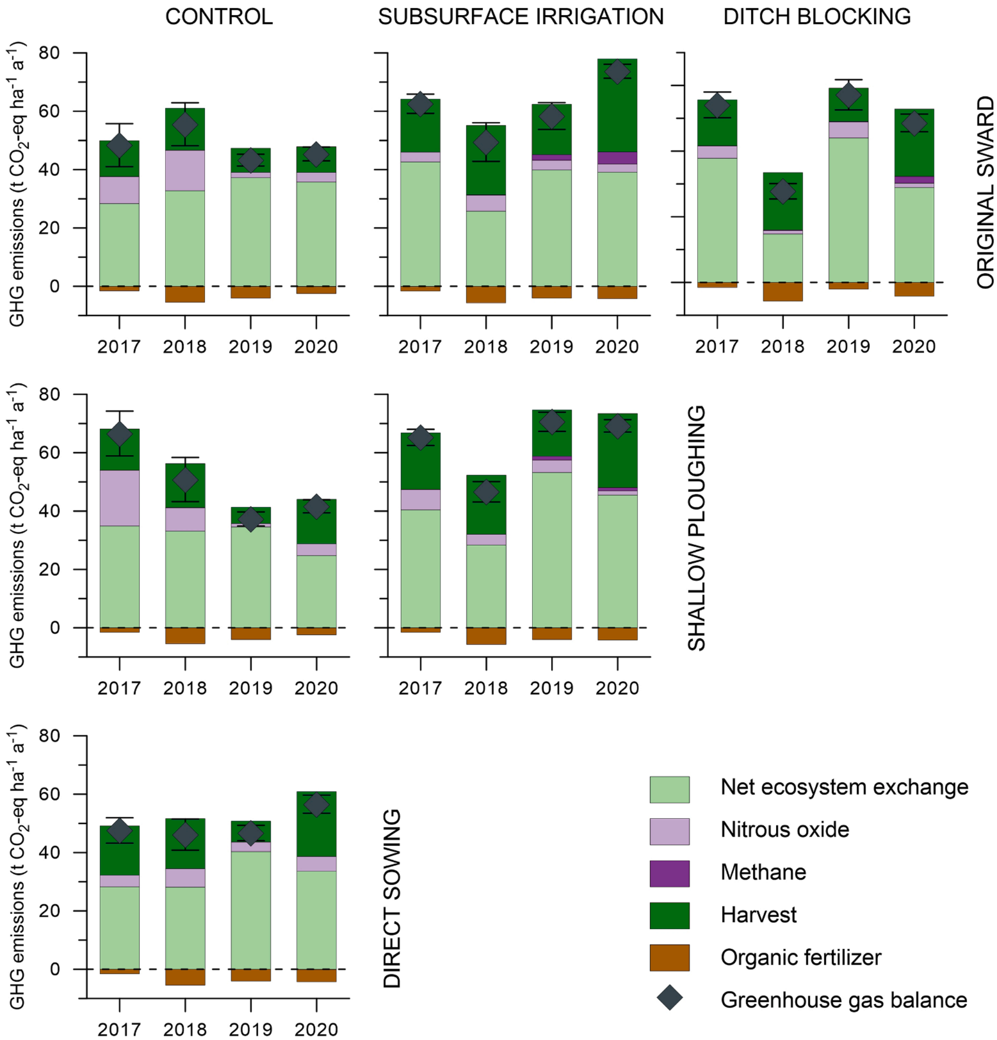

Fig. 8. Annual greenhouse gas (GHG) balances of all treatments. Uncertainty bars show the standard error of the GHG balance. Note that in 2020, plots of the control treatments were still located at the same spots as before but do not represent the original grassland renewal treatment anymore due to the mice damage in 2019, but were handled as follows: original sward: free succession, shallow ploughing: re-sown, direct sowing: transplanted sward.

the reaction of R_h to an increase in soil temperature was similar for all treatments in 2017 and 2018, there were strong differences in 2019 and 2020: Despite lower maximum soil temperatures, the increase in R_h was much stronger for the treatments with water management, particularly with subsurface irrigation. This resulted in the activation-like parameter E_0 strongly increasing with WL (Fig. 7a, $R^2 = 0.72$), while any correlation between R_{ref} and mean annual or mean summer water level was weak (Fig. 7b, $R^2 = 0.13$). There was no correlation between Lloyd-Taylor parameters shown in Fig. 7 and biomass export, pointing to a plausible partitioning between R_h and autotrophic respiration.

3.3.3. *Greenhouse gas balances*

Fig. 8 shows all components of the GHG balance by treatment and year. Due to both high CO_2 and N_2O emissions, the total GHG emissions from the control treatments (mean and standard deviation of the site means: 48.8 ± 0.6 t CO_2-eq. ha^{-1} $^{yr-1}$) were by far higher than previously reported for grassland on bog peat in Germany (15.2 ± 9.2 t CO_2-eq. ha^{-1} yr^{-1}, Tiemeyer et al., 2016). The high variability between treatments and years is mainly caused by the N_2O emissions. Depending on treatment and year, N_2O emissions were an important component of the GHG balances (3 to 29%, mean 13%), while CH_4 was not relevant for the GHG balances (maximum <0.2%).

Due to the strongly elevated CO_2 emissions, the total GHG emissions of the treatments with subsurface irrigation (62.0 ± 1.3 t CO_2-eq. ha^{-1} yr^{-1}) were significantly ($p < 0.01$) higher than those of the control treatments (48.8 ± 0.6 t CO_2-eq. ha^{-1} yr^{-1}). This corresponds to an increase of 31 ± 14 % (Table 4). N_2O emissions from the SI treatments were both in absolute terms (Table 3) and relatively (2–11%, mean 7%) less important for the total GHG balance than in the case of the control treatments. Despite slightly elevated CH_4 emissions, they contributed only 0 to 6% (mean 2%) to the total GHG balance. In the case of ditch blocking there were no overall effects on the GHG balance compared to the control treatments as CO_2 emissions were slightly higher, but N_2O emissions considerably lower than from the control treatments (Table 3). Consequently, mean total GHG emissions (46.9 t CO_2-eq. ha^{-1} yr^{-1}) were also significantly lower ($p < 0.05$) than those of the SI treatments.

4. Discussion

4.1. *Successful water management*

The first study on CO_2 fluxes from grassland with subsurface irrigation (Weideveld et al., 2021) was conducted at field sites where the water level could only be raised marginally in summer by subsurface irrigation, while some mean annual water levels were even slightly lower than those of the controls (Fig. 6). In contrast, Offermanns et al. (2023) showed that water levels could be effectively raised by subsurface irrigation using a pressurized system. As in their study, our water management was highly effective and raised the annual and summer water levels by 0.38 and 0.42 m, respectively (Table 2). When not including the first transition year, the resulting mean annual water levels (-0.27 m) were so high that they exceeded the threshold of -0.30 m proposed by IPCC (2014) for "shallow drained grassland" with lower default CO_2 emission factors at least for nutrient-rich organic soils. Reasons for the differences to Weideveld et al. (2021) could be found in our very high ditch water levels and in the well preserved peat at the depths of the subsurface irrigation drains (von Post degree of decomposition H2 to H4, Table 1). The peat at our sites has thus presumably a higher saturated hydraulic conductivity than the peat and clayey peat found at the Dutch sites of Weideveld et al. (2021).

On average, the blocked ditches raised the mean water levels as much as the subsurface irrigation (-0.33 m, Table 2), but there was stronger intra-annual variation (Fig. 3). Here, it is important to stress that the high water level in the parcels with blocked ditches and also in the ditch feeding the subsurface irrigation drains depended on additional water pumped from an adjacent major drainage channel (Fig. 1). Without this water, it would not have been possible to maintain high ditch and thus field water levels during summer (Kalinski et al., 2021).

The higher water levels in the parcels with subsurface irrigation and blocked ditches were clearly an advantage when a strong infestation with mice caused regional damage to grasslands (Strotmann, 2020). This resulted in higher yields particularly during 2019 and 2020. Furthermore, during the drought year 2018, slightly higher grass yields were found at the plots with water management. Even in the comparatively wet year 2017, there was no disadvantage of raising the water levels by subsurface irrigation. Although we cannot conclude on extremely wet years, the water management was also agronomically successful, also as there was no succession to unwanted species.

4.2. High level of nitrous oxide emissions

Overall, N_2O emissions especially of the deep-drained control sites were very high (Table 3, 4.4 to 45.8 kg N_2O-N ha^{-1} yr^{-1}) and strongly correlate with extractable N_{min} stocks in the upper 30 cm (Fig. S2). Accordingly, N_2O emissions were relevant for the total GHG balance (Fig. 8). For grassland on bog peat in Germany, Tiemeyer et al. (2016) found N_2O emissions of 1.8 ± 2.4 kg N_2O-N ha^{-1} yr^{-1}, while the default emission factor of the IPCC Wetland Supplement (IPCC, 2014) for nutrient poor grasslands is slightly higher (4.3 kg N_2O-N ha^{-1} yr^{-1}). However, underlying data for the German emission factor comprises only one site with intensively used grassland on bog peat, while the remaining data originates from rather wet, unfertilized grasslands under nature protection (details in Leiber-Sauheitl et al., 2014 and Förster, 2016). Still, more recent results of a deeply drained and intensively used grassland on bog peat also showed rather low N_2O emissions (3.9 kg N_2O-N ha^{-1} yr^{-1}, Offermanns et al., 2023). However, similarly high N_2O emissions as in our study have been measured at croplands in Finland (5–37 kg N_2O-N ha^{-1} yr^{-1}, Regina et al., 2004) and grassland on fen peat in the Netherlands with a low pH (<5,0), intensive use and a similar fertilization regime as our site (13 to 32 kg N_2O-N ha^{-1} yr^{-1}, van Beek et al. 2010, 2011).

N_2O emissions are known to correlate negatively to pH values (e.g. Wang et al., 2017; Weslien et al., 2009) as the final reduction step of N_2O to N_2 is limited by low pH values. In combination with the high N fertilisation rates (Table S2), the partially high soil moisture during or after fertilisation and the partially slow biomass development in 2017 after grassland renewal might explain the high N_2O emissions. Thus, current emission factors for intensively used grasslands especially with low pH values might need to be reconsidered.

Sufficient soil moisture, i.e. a water filled pore space of 80% (Butterbach-Bahl et al., 2013) or even higher (Säurich et al., 2019a), is prerequisite for high N_2O emission. The lowest N_2O emissions from the control treatments were measured in 2019, which was still impacted by the drought in 2018. Overall, 2018 was the dryer year, but in contrast to 2019 water levels were still relatively high in early spring (Fig. 2) when the first fertilisation event took place. This first fertilisation event generally tended to cause the strongest N_2O peak, possibly not only due to the higher soil moisture, but also to freeze-thaw events, which are known to trigger high N_2O emissions (e.g. Koponen and Martikainen, 2004). Therefore, avoiding fertilisation in February and March seems to be an opportunity to reduce N_2O emissions.

4.3. Effects of water management on nitrous oxide emissions

Contrary to our hypothesis, all three sites with water management showed, on average, lower N_2O emissions than the control (Table 3). The mostly lower yields of control treatments (Fig. 4) suggest that N_2O emissions might reflect the N surplus. However, no correlation at the level of annual balances could be found, as differences between N_2O emissions from the water management and control treatments were strongest when difference in yield were comparatively minor. While there was a correlation between extractable N_{min} stocks and N_2O fluxes (Fig. S2), the N_2O fluxes from the water management treatments were much lower than those from the control parcels at the same N_{min} values. This points to a more efficient complete denitrification at the water management treatments. In view of the better plant growth and thus the higher N uptake it also seems surprising that 2019 was the only year where N_2O emissions of the water management treatments exceeded those of the control sites. However, in contrast to previous years, the control sites remained relatively dry during winter, which might have prevented high N_2O fluxes during spring. This interpretation is supported by the results in 2020, when the control sites were again wetter in spring, and N_2O emissions again exceeded those of the water management treatments.

4.4. Effects of grassland renewal on nitrous oxide emissions

As grassland renewal has already taken place on 26.09.2016 and N_2O flux measurements also started in September 2016, the endpoints of the cumulative N_2O curves (Fig. 5) do not equal the sums of the annual values given in Table 3. Overall, the effects of grassland renewal were variable. As the autumn of 2016 was rather dry, and the water management system not yet functioning, N_2O emissions after grassland renewal remained low until first frost-thaw events and slightly higher water levels occurred. Starting in summer 2017, there were stronger N_2O peaks after fertilisation. In line with our hypothesis, the first year after renewal showed approximately twice as high N_2O emissions from both treatments with shallow ploughing compared to the original swards. Contrary to our hypothesis that grassland renewal generally increases N_2O fluxes, emissions from the direct sowing treatment were lower than those from the original sward, possible because of the higher uptake by biomass (Fig. 3).

As hypothesised, the effect of grassland renewal by shallow ploughing was stronger for the SI than for the control treatment. Further, the cumulative N_2O emissions of the SI treatments increased rather continuously over the four years, while control treatments strongly reacted to fertilisation during the wet spring in 2018. Afterwards, the sites were probably too dry for very high N_2O fluxes. Therefore, the parcels with subsurface irrigation might be more sensitive to any management intervention (grassland renewal, fertilisation) as sufficient soil moisture for denitrification is frequently given, while at the control treatments, dry conditions might have prevented even stronger effects of the grassland renewals. This is also supported by the study of Offermanns et al. (2023) who meausured extremely high N_2O fluxes after grassland renewal at a site with subsurface irrigation.

Overall, the effects of grassland renewal declined with time, but not as quickly as in the study of Buchen et al. (2017), who only observed a short peak, but no significant differences in annual N_2O emissions at a histic Gleysol. In contrast, enhanced N_2O emissions over 2–3 years were found at an intensively used grassland on mineral soil (Ammann et al., 2020).

4.5. Effects of water management on methane emissions

Methane contributed at maximum 4.2 t CO_2-eq ha^{-1} yr^{-1} or 6% to the annual GHG balances, confirming our initial hypothesis on its minor importance for our study site. Due to low water levels, CH_4 emissions from the control treatments were very low, while emissions from the water management treatments started to show elevated values in 2019 and even more so in 2020 (Table 2, Fig. S3). Possibly, the higher emissions in 2020 than in 2019 despite similar water levels might be explained by a slow (Hahn-Schöfl et al., 2011), but improved adaption of the microbial community to wetter conditions. CH_4 emissions from the control treatments were within the expected or even in the lower range of nutrient-poor grasslands on organic soils (IPCC default values: 0.7–2.9 kg CH_4 ha^{-1} yr^{-1} (IPCC 2014); German data: 24 ± 35 kg CH_4 ha^{-1} yr^{-1} (Tiemeyer et al., 2016)), while emissions from the water management treatments were clearly higher in 2019 and 2020.

4.6. High CO_2 emissions of the control treatments

In comparison to other grassland on bog peat in Germany and Ireland (Renou-Wilson et al., 2014, 2016; Tiemeyer et al., 2016), CO_2 emissions of our study site were very high (Fig. 6). As in the case of N_2O it should be noted that previous measurements in Germany comprised only two study sites with intensively used grassland (Beetz et al., 2013; Offermanns et al., 2023), while the remaining data originates from rather wet, unfertilized grasslands under nature protection which partially even featured typical peatland species (Leiber-Sauheitl et al., 2014 and Förster, 2016). The same applies for the Irish data on pastures with low-intensity grazing (Renou-Wilson et al., 2014; 2016). However, the possibility of high CO_2 emissions from intensively used grasslands on bog peat has already been shown by recent results from the Netherlands (Weideveld et al., 2021). Therefore, it might be that the peat type is not as decisive for CO_2 emissions from drained peatlands as hydrological conditions or management intensity.

The possibly so far underestimated (IPCC, 2014; Tiemeyer et al., 2016) high CO_2 emission potential of bog peat is also supported by laboratory studies which showed that the microbial and soil chemical properties of bog and fen peat become more similar after undergoing degradation and secondary pedogenetic transformation (Urbanová and Bárta, 2016; Säurich et al., 2021), and that the respiration potential of bog peat is as high as of fen peat under these conditions (Säurich et al., 2019b). Laboratory incubation of our study site's peat at different water contents also confirmed the high emission potential (Säurich et al., 2019a).

4.7. Even higher CO_2 emissions of treatments with subsurface irrigation, no change due to blocked ditches

Contrary to our hypothesis, CO_2 emissions from the SI treatments were 38% higher than those of the control treatments, and this increase was particularly clear in the two years with the highest water levels (2019, 2020). These results reject previous assumptions based on subsidence measurements that subsurface irrigation would halve the CO_2 emissions from agriculturally used peatlands (van den Akker 2012, 2017).

Despite a similarly clear raise of the water level by blocked ditches, the CO_2 emissions were similar to those of the control treatments. To our further surprise, there was neither a linear nor non-linear negative dependence of the measured CO_2 emissions on water level. In the following, we will discuss a number of explanations for our results:

- Artefacts cause by the infestation of mice
- Transition period after implementation of hydrological change
- Intra-annual interaction between soil moisture and soil temperature
- Water limitation vs. soil moisture optimum for microbial activity
- Dependence of the temperature sensitivity of respiration on soil moisture
- Improved nutrient supply by the retention of nutrient-rich water and release of redox-sensitive P

The infestation with mice had no effect on the C balance of the control treatments: While the NEE of all control treatments was highest in 2019, this was counteracted by the lower C export by harvest (Fig. 8). Further, the ecosystem respiration (R_{eco}) of the water management treatments exceeded those of the controls already in spring 2019, i.e. long before mice damage occurred. When discussing the total CO_2 emissions of damaged sites, the uncertainty caused by the additional "export" of biomass by mice needs to be considered. This cannot be directly quantified, but assuming that the ratio of harvest and GPP would have been the same as in the two previous years (23 %), the amount of biomass eaten by mice can be estimated from GPP. This corresponds to around 5 t CO_2 ha^{-1} yr^{-1} not accounted for in Fig. 6, Fig. 8 and Table 3. However, this is probably overestimating the error as we cannot make any estimate on the amount of C brought back to the system by mice faeces. These uncertainties cannot explain the difference between the control and the water management treatments, and would generally be relevant for 2019 only. Further, it should be noted, that mice damages occurred at the control sites at which CO_2 emissions behaved as expected as both CO_2 emissions and groundwater levels were similar to previous years. In the case of the SI, the high emissions measured in 2019 were confirmed by the results of the last measurement year. Finally, mice within or below the chambers could have caused high CO_2 fluxes, but there were no conspicuous fluxes. Again, this would have rather affected the control treatments, as the mice largely avoided the wetter sites. Overall, there is no evidence that the mice infestation is a reason for the higher CO_2 fluxes of the SI treatments.

Due to the strong changes in soil hydrology, there might be a transition period needed for the microbial community to adapt to the new conditions. There are indications that this might be the case for methane (see above). In the case of CO_2, however, emissions from the treatments with subsurface irrigation rather increased during the course of the study and were in 2020 at the same level as in 2019 (SI_P) or even higher (SI_O). In comparison to the re-wetting of peat extraction areas, which do not have a fertilized topsoil, there is, overall, a lack of studies on rewetted bog peatlands previously under intensive agricultural use. A study at a bog site previously used for forestry showed a strong reduction of CO_2 emissions or even CO_2 uptake at the level of near-natural sites within the first year after re-wetting (Komulainen et al., 1999). Similarly, a rewetted bog peatland previously used as low-intensity grassland showed CO_2 uptake 2–6 years after full re-wetting (Förster 2016). In addition, *Sphagnum* paludicultures may be a CO_2 sink even in their initial phase (Günther et al., 2017). Therefore, a fast adaption of the microbial community to new hydrological conditions might be expected. Furthermore, for productive grassland use, fertilization will have to continue, and at some point, the drainage depth would have to be lowered again, leading to a renewed transition period. Still, four years is not a long period in terms of ecosystem development, and future changes can of course not be excluded.

Laboratory experiments with different peat types frequently showed that highest CO_2 fluxes occurred at an optimum value of the water content i.e. that there is a parabolic dependency on either water filled pore space or soil moisture (Byun et al., 2021; Säurich et al., 2019a; Taggart et al., 2012; van Lent et al., 2018). For example, the highest CO_2 fluxes from topsoils of our study site occurred at a water filled pore space (WFPS) of 83 % (Säurich et al., 2019a). These high optimum values of soil moisture are not an exception; e.g. Byun et al. (2021) found optimum values of WFPS of 60–96 % for a range of temperate peat soils. Due to such high optimum values of WFPS it can be assumed that microorganisms at the control treatments were limited by the lack of water for large parts of the growing season, and that this limitation is lifted by raising the water levels. Of course, these observations should not invite the interpretation that deeper drainage would by a reasonable mitigation measure, as in the long term, more peat would be aerated, and vegetation growth would suffer even more than during the study period.

In this context, it is important to note that there is usually a strong interaction between temperature and soil moisture/water level during the course of the year, i.e. wet phases are usually cool and vice versa. Subsurface irrigation strongly changes the distribution of water levels, and therefore optimum soil moisture can also occur at higher temperatures. Whereas the relationship between groundwater level and soil temperature at the control treatments was similar in all four years, the patterns for the water management treatments changed over the study period (Fig. S5). While DB and SI behaved similarly in 2017, they differed in the following years. In 2019 and 2020 the temperature-water level relationship of the treatments with SI was different especially from the CON sites, which might contribute to the understanding of why those two years showed the highest CO_2 emissions. This interpretation is supported by results of Tiemeyer et al. (2016) who found in a synthesis study on 48 grasslands on organic soils that those sites – given a similar mean annual water level – which do *not* become very dry in summer showed higher CO_2 emissions than those which do.

Of course, respiration rates do not only depend on a suitable soil moisture range, but are generally strongly dependent on temperature. There was only slightly higher (estimated) heterotrophic respiration (R_h) from the treatments with raised water levels at the same temperature in the first two years. In 2019 and 2020, the temperature sensitivity of R_h at these sites was clearly higher than that of the control treatments (Fig. S4). This effects even outweighs the lower soil temperatures caused by the raise of the water level: In summer, subsurface irrigation dampened the amplitude of the soil temperature in 2 cm depth by up to 10 °C and led to generally lower maximum temperatures (Fig. S4).

These results strengthen the interpretation that respiration at the control treatments was water-limited and that water levels between −0.25 and −0.35 m in summer provide optimum conditions for respiration, at least for the given peat properties and nutrient supply. Similarly, Mäkiranta et al. (2009) found in a field study on forested peatlands in Finland that the CO_2 fluxes from moist plots showed a stronger temperature sensitivity than those from dry plots. As in our case (Fig. 7), they found a linear relationship between E_0 and mean annual water level with highest E_0 values at −0.35 to −0.40 m. They also explained their results by a water deficit in the topsoil or, alternatively, by a different composition and a lower biomass of the microbial community at the dry plots. Taggart et al. (2012) also observed in a laboratory study with warm climate Histosols a stronger temperature sensitivity at higher soil moisture. Similar results were reported by Byun et al. (2021), who found the strongest temperature sensitivity of respiration for peat from natural temperate peatlands at WFPS between 78 and 100% (with one exception of 53%). To our best knowledge, similar data is missing so far for drained temperate peatlands, but temperature sensitivity of ecosystem respiration decreased with decreasing soil moisture at a grassland site on a SOC-rich mineral soil in northern temperate Canada (Flanagan and Johnson, 2005).

Processes discussed so far can only explain why CO_2 emissions at the dry control treatments were lower than those of water management treatments, particularly with subsurface irrigation, but only insufficiently why the CO_2 emission during the years with highest water levels were particularly high. Besides the relationship between water level and temperature and the ensuing increased temperature sensitivity, the nutrient supply and retention and the specific properties of *Sphagnum* peat can offer an interpretation. Incubation studies of Brake et al. (1999) and Säurich et al. (2019a, b) have shown that (plant-available) phosphorus has a strong impact on respiration rates, while Amador & Jones (1993) found that P fertilisation increased respiration of peat with low and intermediate P content. All treatments generally received the same fertilisation, but due to missing binding sites, P is generally weakly retained (Nieminen and Jarva, 1996) and easily leached from agriculturally used bog peat (Kuntze and Scheffer, 1979). However, both subsurface irrigation and blocked ditches keep nutrient-rich water within the peat and even cause the infiltration of ditch water. Nutrient concentrations in the water of the main drainage channel supplying the experiment could only be measured occasionally in 2020. Mean concentrations (± standard deviation) were 1.7 ± 0.2 mg l^{-1} (total phosphorus) 1.6 ± 4.4 mg l^{-1} (nitrate, only one value > 0.2 mg l^{-1}) and 0.5 ± 0.2 mg l^{-1} (ammonium). The hydrological conditions imposed by subsurface irrigation and ditch blocking cause a hydrologically and hydro-chemically untypical situation even for agriculturally used bog peatlands as there is usually no inflow of potentially nutrient-rich groundwater and surplus water is quickly transferred to the drains. In this context, it is important to stress that while there is additional nutrient input via the surface water containing especially P, the major source of nutrients is autochthonous, i.e. fertilisation and decomposition of the peat. This can be seen from the high values of extractable nitrogen stocks (Fig. S2) dominated by extractable nitrate (72–95 %, data not shown) despite low nitrate concentrations in the main drainage channel. Thus, especially the retention of water and thus nutrients is relevant for biochemical processes in the peat.

Despite these hydrological differences, the topsoil density of plant available phosphorus (P_{CAL}) was in 2020 similar in all treatments although there were strong differences in yield (Fig. S6). In other words, there must have been a high P supply to allow for the higher yields. High water levels cause a reduction of the redox potential and thus possibly the release of redox-sensitive phosphorus (Zak et al., 2008). We used dithionite-extractable phosphorus (P_{BD}) as a proxy for reductant-soluble P (Zak et al., 2008), and indeed both this P fraction and the difference between P_{BD} and P_{CAL} were clearly higher in the control treatments (Fig. S6), suggesting that it has already been released in the treatments with elevated water levels. So far, there is limited data on P fractions in different bog peats, but our results confirm that such sites might react similarly to iron-poor fen peat sites with a high risk for P release (Zak et al., 2010). The release of redox-sensitive P might not only have contributed to the higher respiration rates, but also to the higher yields of the treatments with raised water levels.

The combination of comparatively high water levels in summer and improved P supply could thus explain the high CO_2 emission in 2019 and 2020: During these years, conditions especially in the topsoil (amorphous earthified peat) might have been ideal. Several laboratory studies have shown that respiration rates of degraded topsoil peat enriched with nutrients are much higher than those of subsoils with more or less undisturbed peat (Brake et al., 1999; Säurich et al., 2019a, 2019b). In contrast to drained fen peat, the earthification and further secondary pedogenetic processes in bog peat are restricted to the upper 10 to 20 cm despite deeper drainage. This is also the case at our study site, where there is a sharp boundary between the black, amorphous topsoil and the directly underlying weakly decomposed peat (H2 to H3 on the von Post scale, Table 1). Therefore, it can be assumed that microbial decomposition and transformation of the peat is largely limited to the topsoil, and that changes of the hydrological conditions in the subsoil are of minor importance for the microbial processes in the subsoil. Further, the strong differences in the soil properties within the profile can also explain the differences to a study by Boonman et al. (2022) who found both in the field (fen peat) and in modelled scenarios that raising the water levels was particularly effective in dry years: They assumed homogenous maximum respiration rates for the whole profile. Overall, our results also strongly reject previous assumptions that a strong reduction of CO_2 emissions could be achieved by raising the mean lowest water levels with subsurface irrigation a few decimetres in the subsoil (Brouns et al., 2015; van den Akker et al., 2012) in the case of bog peat.

When comparing our results to previous work on the relationship between groundwater level and CO_2 emissions it needs to be considered that there was nearly always an inherent correlation between groundwater level and land use intensity (Fig. 6, Evans et al., 2021; Tiemeyer et al., 2020). While there is some data on relative dry grasslands on peat with low-intensity use (i.e. 1–2 cuts, no fertilisation), this is the first multi-year study to investigate the combination of very moist conditions and intensive grassland use on bog peat. Thus, functions calculating CO_2 emissions from water levels published so far may not be transferred to peatlands with subsurface irrigation and intensive agricultural management. Finally, the question remains unanswered whether a combination of water by subsurface irrigation and nutrient management (i.e. lower fertilisation rates and a reduced number of cuts) might lead to a reduction of CO_2 emissions, or whether the effects of optimum soil moisture in summer would prevail.

5. Conclusions

On average, mean annual groundwater levels at the parcels with subsurface irrigation and ditch blocking were −0.33 m and thus much higher than at the control parcels (−0.71 m). This raised water level proved to be favourable both for minimizing mice damage and safeguarding grass yields, especially in dry years. Thus, the experiment was successful in hydrological and agronomical terms.

Both CO_2 and N_2O emissions of the control treatments were high, revoking previous assumptions of lower emissions of both gases from grasslands on bog peat than from those on fen peat. Indeed, CO_2 emissions were on average 38% higher at parcels with subsurface irrigation than at the control parcels. Despite lower N_2O emissions, GHG emissions were still clearly (31%) increased by subsurface irrigation. CO_2 and GHG emissions of the treatment with blocked ditches were similar to those of the controls. These surprising results regarding CO_2 might be explained by an interaction of increased soil moisture in the topsoil and improved nutrient retention during phases of high soil temperatures facilitated by subsurface irrigation and, at the same time, by limitations of microbial activity due to low soil moisture at the control parcels.

Due to the increased CO_2 and GHG emissions despite clearly raised water level, results of this study do not allow recommending subsurface irrigation as a mitigation measure for intensively used grassland on bog peat. However, it is important to stress that our results do not question the positive effects of raised water levels in the context of wet grasslands (Förster, 2016) or *Sphagnum* farming (Günther et al., 2017; Oestmann et al., 2022) which do not receive any fertilisation and are, in the case of *Sphagnum* farming, based on the cultivation of typical bog plants. Thus, water management preferably at the catchment scale remains a necessity for mitigation measures in peatlands.

CRediT authorship contribution statement

Bärbel Tiemeyer: Conceptualization, Writing – original draft, Visualization, Formal analysis, Supervision. **Sebastian Heller:** Investigation, Formal analysis, Writing – review & editing. **Willi Oehmke:** Investigation. **Peter Gatersleben:** Investigation. **Melanie Bräuer:** Investigation, Software, Writing – review & editing. **Ullrich Dettmann:** Software, Formal analysis, Writing – review & editing.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

Data will be made available on request.

Acknowledgement

We are grateful for the financial support of the project "SWAMPS" by the European Regional Development Fund, the Lower Saxony Ministry of Food, Agriculture and Consumer Protection and the Lower Saxony Ministry of the Environment, Energy and Climate Protection (ZW-85003757). Our gas chromatography laboratory (Roland Fuß, Daniel Ziehe, Andrea Oehns-Rittgerodt and Kerstin Gilke) analysed the gas samples, while the central laboratory (Nicole Altwein, Ines Backwinkel, Ute Tambor) and the laboratory for soil monitoring (Arne Heidkamp) were invaluable for soil chemical and biomass analyses. We also want to express our thanks to Thomas Viohl for soil physical analyses. The extensive field work in this project would not have been possible without the support of Dirk Lempio, Frank Hegewald, Marc Jantz, Thomas Viohl, Lena Barczyk, Katja Braun, Annalisa del Ben, Johannes Eltzsch, Jessica Hügen and Julia Zenner as well as the Peatland Research Group of the Thünen Institute of Climate-Smart Agriculture. We also express our thanks to our project partners. Further, two anonymous reviewers helped to improve the manuscript by their insightful comments. Finally, the project would not have been possible without the farmer who granted us access to their field and provided us with all the necessary information and support to conduct our study.

Supplementary materials

Supplementary material associated with this article can be found, in the online version, at doi:10.1016/j.agrformet.2023.109858.

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