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Groundwater level effects on greenhouse gas emissions from undisturbed peat cores

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ABSTRACT

Peat soils store a large part of the global soil carbon stock, which can potentially be lost when they are drained and taken into cultivation, resulting in CO_2 emission and land subsidence. Groundwater level (GWL) management has been proposed to mitigate peat oxidation, but may lead to increased emissions of nitrous oxide (N_2O) and methane (CH_4).

The aim of this experiment was to study trade-offs between greenhouse gas emissions from peat soils as a function of GWL. We incubated 1 m deep, 24 cm diameter undisturbed bare soil cores, after removal of the grass layer, from three contrasting Dutch grassland peat sites for 370 days at 16 °C. The cores were subjected to drying-wetting cycles, with the GWL varying between near the soil surface to 160 cm below the surface. We measured gas fluxes of CO_2, N_2O and CH_4 from the soil surface, extracted pore water for DOC and mineral nitrogen analysis, and measured soil hydraulic and shrinkage characteristics.

Emissions of CO_2 increased after lowering the GWL, but showed different GWL-response curves during rewetting of the soil. On average, highest CO_2 emissions of 1.5 g C·m^{-2} day^{-1} were found at a GWL of 80 cm below the surface. However, the 0 cm GWL was the only treatment with significantly lower CO_2 emissions than other GWLs. Cumulative CO_2 emissions differed significantly between sampling sites. Emissions of N_2O showed a different response, peaking at GWL heights above −20 cm, particularly after a recent GWL rise. Though not significantly different, the highest N_2O emissions were measured at the 0 cm GWL treatment. We confirmed this pattern for N_2O in un-replicated soil cores with grass sward, although emission values were lower in these cores due to the root uptake of mineral nitrogen. CH_4 emissions or −uptake remained low under any GWL. We conclude that raising the GWL is a successful strategy to reduce CO_2 emissions from peat oxidation. However, raising the GWL close to the soil surface could lead to N_2O emissions that negate any gains in terms of global warming potential. Our results suggest that raising the GWL in peat grasslands to −20 cm creates such a risk. A constant GWL at the surface (0 cm) would be preferential for mitigating both CO_2 and N_2O emissions, although such conditions don't allow for agricultural grass production (mowing or grazing).

1. Introduction

Peatlands are formed under wet, anaerobic conditions and are characterized by a high soil organic matter (SOM) content resulting from the accumulation of dead plant material. Globally, peat soils are estimated to have accumulated over 600 Gt carbon (C) during their formation (Yu et al., 2010), making up about 25 % of the current estimated C pool stored in soils worldwide (Jobbagy & Jackson, 2000). Across the

Abbreviations: GHG, greenhouse gas; GWL, groundwater level; GWP, global warming potential; SOM, soil organic matter; VWC, volumetric water content; WFPS, water filled pore space.

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world, many peat soils experience C loss due to oxidation of their SOM, resulting in emissions of carbon dioxide (CO_2). This can be the result of active drainage for agricultural practices or groundwater extraction resulting in aeration of the soil and accelerating oxidation reactions (Hiraishi et al., 2014). Greenhouse gas (GHG) emissions from peat soils (of which by definition at least 40 of the upper 80 cm of the soil profile is organic soil) are especially important in the Netherlands, as many peat soils are drained for dairy farming (RIVM, 2022). Drainage of organic soils (peat and peaty soils, of which the latter contain an organic layer making up less than 40 cm in the upper 80 cm) is the largest source of CO_2 emissions from land use and land use change in the Netherlands, accounting for a substantial part of the Dutch national GHG emissions (RIVM, 2022). In addition to increasing GHG emissions, drainage causes subsidence of peat soils at rates ranging from 2 to 25 mm yr^{-1}, through SOM oxidation as well as shrinkage and consolidation processes (Erkens et al., 2016; Van den Akker et al., 2008). Van den Akker et al. (2008) estimated that a subsidence of 1 mm yr^{-1} is associated with an emission of 2.2 Mg CO_2 $ha^{-1}yr^{-1}$. Soil subsidence can cause damage to buildings and infrastructure, affects the water regulation in adjacent nature reserves, reduces the water storage capacity of an area and can cause upwards seepage of salt water (Van den Born et al., 2016). Moreover, it increases the flood risk of the Dutch low coastal peat land, an area already situated below the mean sea level.

In order to reduce CO_2 emissions from these soils, as well as to prevent further subsidence, it has been suggested to raise the groundwater level (GWL) through the raising of ditchwater levels and improving infiltration via drains or trenches. Particularly the prevention of relatively deep GWLs during dry summer periods may be effective in reducing CO_2 emissions and subsidence (Hoving et al., 2022). Based on several past and ongoing studies to explore the potential of subsoil irrigation systems for better GWL manipulation (Boonman et al., 2022; Van Asselen et al., 2023), the Dutch government has expressed its intent to raise the GWL in the low lying peat meadow areas to −40 or even −20 cm with respect to the soil surface (z-coordinate positive upwards). This should eventually result in an emission reduction from peat soils of 1 Tg CO_2-eq yr^{-1} (Ministerie van Landbouw, 2019).

However, raising the GWL might create conditions beneficial to the production of nitrous oxide (N_2O) and methane (CH_4), two GHGs with a strong warming potential that possibly could negate positive effects of CO_2 reduction. Production of N_2O in soils mainly occurs through microbial nitrification and denitrification (Butterbach-Bahl et al., 2013; Canadell et al., 2021). The rate of emission is the result of several processes, each affected by soil parameters, soil–plant interactions and external conditions such as nitrogen (N) input through fertilizer application or precipitation and temperature (Butterbach-Bahl et al., 2013; Firestone & Davidson, 1989). In cultivated peat soils, denitrification is the dominant N_2O producing process, due to the high SOM content, frequent anaerobic conditions, and high mineral-N availability, resulting from organic N mineralization and fertilizer input (Van Beek et al., 2004). Annual N_2O emissions due to mineralization of organic N in organic soils (grasslands and croplands) were estimated at 0.7 Tg CO_2-eq in 2020, about 14 % of total N_2O emissions from agricultural soils in the Netherlands (RIVM, 2022). Although raising the GWL in peat soils may reduce N_2O resulting from peat mineralization (Van Beek et al., 2010), it may also introduce a risk of high N_2O production (Velthof et al., 1996a). The attempt to create a fixed high GWL close to (but below) the soil surface, may lead to strong fluctuations in moisture conditions in the topsoil during rainfall events. This could give rise to a combination of nitrate (NO_3^-) production during SOM oxidation and nitrification under aerobic conditions, followed by incomplete denitrification and N_2O diffusion to the atmosphere under anaerobic conditions (Goldberg et al., 2010). The presence of the anaerobic–aerobic interface close to the surface, may decrease the diffusion time to the atmosphere. Given the relative delayed activation response of N_2O reductase to anaerobic conditions (Baggs & Philippot, 2010), a higher N_2O/N_2 ratio may be expected for denitrification at shallow depth compared to deeper soil. Mineral N applied as fertilizer (particularly as NO_3^-) and manure to the soil surface, may also be denitrified under the partially anaerobic conditions created by a high GWL (Velthof et al., 1996b). The greenhouse gas CH_4, with a warming potential 27 times higher than CO_2 (global warming potential over 100 years, Forster et al., 2021), is commonly only produced under the strongly anaerobic conditions of wetland soils, and less so in managed peat soils (Van den Pol-van Dasselaar et al., 1997).

The mitigating effect on CO_2 emissions of reducing drainage levels in peat grasslands has been widely confirmed (Evans et al., 2021), although some have suggested an optimum in the relation between water table depth and CO_2 (Laiho, 2006; Tiemeyer et al., 2016). Optima have also been observed for the relationship with soil suction levels (Berglund & Berglund, 2011; Kechavarzi et al., 2010; Saurich et al., 2019). N_2O emissions are known to prevail during variable moisture conditions, often translated into hot moments and hotspots (Berendt et al., 2023). Emissions of CH_4, on the other hand, are highest under wet conditions in a restored peat grassland (Hendriks et al., 2007) or in inundated mesocosms (Van de Riet et al., 2013). Soil inundation did not always result in CH_4 emissions, however, potentially due to an insufficient duration of the treatments (Saurich et al., 2019; Taft et al., 2018). Field experiments are useful for their high degree of realism, incorporating not only the field scale, but also the soil depth dimension. While lab experiments have the benefit of a controlled environment, the size of experimental objects is often reduced to a representation of a single soil horizon, and the soil structure may be disturbed during sieving or drying. Potential reactions during upwards gas transport in the soil column are therefore not fully represented in such studies (Koops et al., 1997). Neither are the hydrological processes creating the environmental conditions in the soil matrix. Experiments on soil mesocosms (with lengths ranging between 25 and 80 cm) in field or lab settings have confirmed the mitigating effect of GWL rise on CO_2 emissions, in combination with a (slightly) increasing CH_4 flux (e.g., Karki et al., 2016; Moore & Dalva, 1993; Nielsen et al., 2023; Van de Riet et al., 2013). Constraints to these studies involved the use of disturbed soil in repacked columns (Moore & Dalva, 1993), a limited number of static GWL treatments (Karki et al., 2016; Nielsen et al., 2023; Van de Riet et al., 2013), or a relatively small column depth, limited to the topsoil (Van de Riet et al., 2013). The hysteresis in CH_4 emissions observed by Moore and Dalva (1993) after drying and rewetting to 50 cm indicates the importance of GWL dynamics. Emissions were strong during drying and near absent during rewetting, indicating production and storage along the depth gradient. A similar importance of dynamic water levels can be expected for N_2O emissions. Unlike CH_4 however, N_2O production may be most pronounced during rewetting of previously aerated layers (Goldberg et al., 2010). Regina et al. (2015) observed an increase in N_2O during wetting from 70 to 30 cm GWL depth, although shallower GWLs were not included. A mesocosm incubation treated with dynamic drainage levels (drying and rewetting) and including detailed monitoring of moisture content, matric potential and GHG emissions could provide better insight into the oxidation processes occurring in drained grassland peat soils. To our knowledge, no such an experiment has been conducted yet.

Therefore our aim in this experiment was to study the trade-offs between CO_2, N_2O and CH_4 emissions associated with peat oxidation, from incubated, non-disturbed soil cores from Dutch peat meadows as a function of a dynamically changing GWL. To be able to measure the effects of heterotrophic respiration, an experiment using bare soil cores was required. We hypothesized that (1) CO_2 emissions are negatively related to the GWL, i.e., higher emissions at a deeper GWL, but reach a maximum when the shallower soil layers have moved from oxygen limitation to moisture limitation; (2) N_2O emissions are highest at a shallow groundwater level, 30 to 10 cm below the soil surface, and occur strongest during rewetting of the soil; (3) CH_4 emissions are low or absent except during inundated conditions; and (4) CO_2 will be the main contributor to the total global warming potential (GWP).

2. Materials and methods

2.1. Locations and sampling

Three dairy farms in the Netherlands, cultivating grassland on peat soils, were selected as sampling locations: Zegveld (52°08′ N, 4°50′ E) and Vlist (51°58′ N, 4°49′ E) in the "Groene Hart" area in central-western Netherlands and Aldeboarn (53°03' N, 5°54' E) in the northern Friesland province (Fig. 1). In Zegveld, a clayey peat top layer, containing some anthropogenic debris, is found above a thick mesotrophic wood peat layer, with eutrophic sedge peat underneath, which reaches down to six meters. The soil on the Vlist location has a peaty clay top layer above mesotrophic wood peat with peat and clay layers reaching as deep as ten meters. In Aldeboarn, two meter deep oligotrophic peat, composed of Sphagnum, Eriophorum and heather, is covered by a thick peaty clay top layer (Van Asselen et al., 2023). Physical and chemical properties of the soil horizons included in this experiment are provided in Table 1. On all three farms the effects of subsoil irrigation systems compared to regular ditch drainage are studied as part of a Dutch research program on reducing greenhouse gas emissions from peat meadows (NOBV, 2021). The fields in Zegveld and Vlist are fertilized with a combination of slurry and artificial fertilizer, at a rate of 200–250 kg N ha^{-1} yr^{-1}. In Aldeboarn, the fields receive only slurry at 170 kg ha^{-1} yr^{-1}. In the summer of 2021, four undisturbed soil cores were sampled close to each other on the experimental field with subsoil irrigation of each location. The sampling plot was kept free from fertilizer application for at least three months before sampling. Three replicates were sampled after removing the upper 5 cm soil including the grass sward, while one replicate was taken without removal of the grass sward. Acrylate tubes with a length of 120 cm and inner diameter of 24 cm were pushed carefully 113 or 118 cm into the soil for the bare and grass cores respectively. To prevent the soil inside the tube from compressing, the tube was closed off on top and connected to a pump to establish a negative air pressure inside the tube while pushing it down.

Fig. 1. Locations of the three sampling sites in the Netherlands.

2.2. Setup experiment

The cores were transported to the laboratory where they were stored in a climate controlled room at 16 °C and 70 % relative humidity. The lower 10 cm of each peat core was removed and replaced with fine grained sand, chosen for its strong capillary suction, after which the tube was closed at the bottom with a watertight PVC cap. Tap water was used for all groundwater level manipulation in the climate room. The cores were kept saturated until the start of the experiment, except during the installation of drains and sensors, for which the cores were drained temporarily. All cores had been saturated for at least two weeks before day 1 of the experiment. Two drainage pipes were installed; one inside the sand layer and the second at the bottom of the peat core. A hanging water column was connected to the upper drain to regulate the water levels down to −100 cm or to the sand drain for suction levels down to −160 cm. To be able to apply suction levels lower than the soil cores bottoms, all cores were mounted on top of a 50 cm high bench (Fig. S1, see Supplemental Information). We used the horizon borders identified during sampling in the field (Table 1), to distinguish layers with different properties in the columns. In the middle of each horizon, a porous cup for pore water extraction, a volumetric water content (VWC) sensor and a tensiometer were installed horizontally into the soil. The thicker top horizon of the Zegveld cores received two rows of probes at equal distances from the horizon borders. To allow for the three highest rows of probes to move along with the vertical shrinkage movement of the soil matrix, they entered the column tube through vertical slots that were sealed water tight (Fig. S2). The cores received no fertilizer throughout the experiment.

2.3. Groundwater level treatments

Between January 2022 and February 2023, all cores were subjected simultaneously to two drying-wetting cycles. The first cycle was aimed at testing the drainage system down to a water table of 100 cm below the soil surface, in eleven one-week steps. These treatment steps included 0, −20, −40, −60, −80 and −100 cm with respect to the soil surface (z-coordinate positive upwards) and vice-versa. The second cycle included the same treatment steps as above, as well as suction induced steps of −140 and −160 cm with respect to the soil surface. All levels in cycle two were maintained for two (sometimes three) weeks, resulting in a duration of 33 weeks for this cycle.

2.4. Biochemical analyses

Gas emissions of CO_2, N_2O and CH_4 were measured using the static chamber method, at the start and the end of each GWL step. Opaque PVC chambers were closed over the cylinders for a period of 30 min, during which the concentration inside was measured four times with a Gasera One photoacoustic gas monitor (Gasera, Finland). Concentration change slopes were determined using linear regression between sampling points, after which flux values were calculated using the ideal gas law. Cumulative emissions over the period of the experiment were subsequently calculated using the 'gasfluxes' package in R (Fuss, 2023). N_2O and CH_4 emissions were converted to CO_2 equivalents using the global warming potential (GWP) values for a 100 years horizon reported in the sixth IPCC assessment report: 273 and 27 for N_2O and CH_4, respectively (Forster et al., 2021).

Water samples were extracted at the end of each GWL step from those rhizon suction cups (Rhizosphere Research Products, the Netherlands) that were located above and below the prevailing water table. At water tables below −100 cm, no water samples were extracted. The samples were analyzed for NO_3^-, NH_4^+ and inorganic and total carbon using segmented flow analysis. Dissolved organic carbon (DOC) was calculated as the difference between total and inorganic carbon.

Table 1
Properties of the soil horizons, at the time of sampling in Aldeboarn (Ald.), Vlist (Vli.) and Zegveld (Zeg.). SOM content is given as the percentage of the total dry mass. Clay particles (<2 μm) content is given as fraction of the remaining (mineral) mass. SOM, total carbon and nitrogen and C/N ratio were measured in the five cm interval in the middle of a horizon. Dry bulk density values are the mean of four replicate 100 cm^3 ring samples. The peat humification status is based on the soil classification performed in the field.

Location	Horizon depth (cm)	SOM (% of dry soil)	Clay (% mineral mass)	Tot-C (g kg^{-1} dry soil)	Tot-N (g kg^{-1} dry soil)	C/N	Dry BD (g cm^{-3})	Peat humification status
Ald.	0–12	28.9	40	128	11.4	11.2	0.6	–
	12–28	18.1	45	78.7	5.5	14.3	0.927	–
	28–45	70.5	20	367	10.6	34.5	0.233	Humified
	45–80	93.6	15	472	9.4	50	0.138	Lightly humified
	80–180	95.4	15	476	10.9	43.5	0.128	Reduced
Vli.	0–15	45	50	203	19.4	10.5	0.595	–
	15–37	17.5	60	60.2	7	8.6	0.835	–
	37–55	60.9	45	322	24.2	13.3	0.256	Humified
	55–75	77.8	40	383	27.6	13.9	0.204	Lightly humified
	75–130	71.4	40	350	23.3	15	0.166	Reduced
Zeg.	0–30	36.5	35	174	16.8	10.4	0.62	Humified
	30–45	69.7	55	334	24.2	13.8	0.248	Humified
	45–65	74.7	35	367	22.9	16	0.204	Lightly humified
	65–150	79.8	45	383	22.6	16.9	0.158	Reduced

2.5. Physical analyses

Volumetric water content and matric potential were monitored using EC-5 capacitance sensors (Metergroup, USA) and tensiometers (Rhizo Instruments, the Netherlands) respectively. All sensors were connected to dataloggers; the recording frequency was every ten minutes. Calibration curves for the relationship between VWC and the EC-5 signal were composed experimentally for each soil horizon present in the soil cores (the Vlist horizons 4 and 5 were merged in one calibration function as well as Aldeboarn horizons 1 and 2).

Vertical displacement of the soil surface was monitored during each gas flux measurement, to account for changing headspace volumes. To monitor vertical as well as horizontal shrinkage of the different soil horizons, small cylindrical markers (20 mm long, 6 mm diameter) were pushed horizontally into the soil through small holes, drilled for this purpose, such that the marker intersection was visible either through the transparent acrylate or through the hole. Four markers were placed at equal distances around the core perimeter, at the middle of each horizon as well as the horizon boundaries and at 2 cm below the core top. This resulted in 36 markers in the Zegveld columns and 40 markers in the Aldeboarn and Vlist columns. At the end of each GWL step, vertical displacements of the markers were determined with a ruler and horizontal displacements by pointing a caliper through the hole in front of the marker. When the marker had moved vertically beyond the level of the hole, the horizontal distance to the soil in front of the hole was measured instead.

2.6. Statistical analyses

Data analyses were performed using the R statistical software (v4.3.1; R Core Team, 2023). We used linear regression to study the relationship between GWL during the two drying and two rewetting tracks and CO_2 emissions for each sampling location and the three locations combined. The intermediate period between cycle 1 and 2 was excluded from these relationships. Cycle 1 ended on 29–03-2022, one week after raising the GWL to 0 cm. Cycle 2 started on 30–05-2022; this was three weeks after raising the GWL from the intermediate −20 cm level to 0 cm, at which point we assume a stable situation of saturation was prevalent again (rather than the response to rewetting). The best fit model was selected based on the models' adjusted R^2 value, Akaike information criterion, and p-values of the coefficients. The CO_2 fluxes and log-transformed N_2O and CH_4 fluxes at different GWL conditions (averaged over the soil types and cycles) were compared in linear mixed effects models using the *lme* function in the R package 'nlme' (Pinheiro et al., 2023). The individual objects- i.e., the soil cores – were added as random factor and the groundwater level treatment, sampling location or both (according to the best fit based on the Akaike information criterion) were added as fixed factors. The model was validated by checking the residuals for normality, homogeneity and independence graphically. Pairwise comparison was performed in case of a significant GWL effect, using a Tukey's honest significant difference test (alpha = 0.05). The same was done for the sum of CO_2 and N_2O, expressed as CO_2-equivalent emissions. CH_4 was excluded from the summed analysis, due to missing values in the CH_4 timeseries. An analysis of variance (ANOVA) was used to compare cumulative emissions of CO_2, N_2O and CH_4 from the three soil types.

3. Results

3.1. Soil moisture and GHG dynamics during drying-wetting cycles

Matric potential values moved along with GWL changes (Fig. S3). The pressure of the water column, recorded by the lowest tensiometer, installed at depths between 80 and 87.5 cm below the surface, generally corresponded with the imposed GWL. At these depths, VWC observations rarely deviated from the saturated VWC levels identified in the sensor calibrations (Fig. S4). In the shallow soil layers (−3.5 to −21 cm depth), tensiometer readings were often obscured by air intrusion into the ceramic cups. The VWC in the top horizon decreased steeply after lowering of the GWL in Aldeboarn and Vlist soils, while it showed a more gradual and less strong decrease in the Zegveld soils.

Cumulative CO_2 emissions, standardized as the yearly rate, ranged between 0.33and 0.64 kg C·m^{-2} yr^{-1}, with stronger emissions occurring in the first drying-wetting cycle (Table 2). Over the course of the whole experiment, cumulative emission was highest in the Zegveld and lowest in the Vlist soil. Lowering of the GWL resulted in CO_2 emission increases for all three soil types (Fig. 2a, 3a, 4a). On average the highest CO_2 emissions were measured at a GWL of −80 cm, but only a 0 cm GWL resulted in significantly lower emissions than other treatments (Fig. 5a). The response of CO_2 to GWL change differed according to the direction of the change (Fig. 6a, 6c, 6e). Lowering the GWL in cycle 1 resulted in bell-shaped CO_2 responses (Fig. 6a, 6c, 6e), best described by second or third order polynomial functions (Table 3). In the second drying track, this pattern was only observed in the Zegveld cores (Fig. 6e). No clear relationship was found in the Aldeboarn cores, while in the Vlist cores, CO_2 emissions even decreased in response to drying (Fig. 6a, 6c, Table 3). Rewetting, on the other hand, resulted in decreasing CO_2 emissions. Contrary to the drying tracks, highest emissions in the rewetting tracks were found at the deepest GWLs (Fig. 6a, 6c, 6e). The response of CO_2 emissions to a rising GWL was best described by linear

Table 2
Mean cumulative emissions from bare soil columns (n = 3, ±SE) of CO_2 (kg C·m^{-2} y^{-1}), N_2O (g N m^{-2} yr^{-1}) and CH_4 (g C $m^{-2}yr^{-1}$) and the corresponding CO_2 equivalent values over a 100 year horizon (kg CO_2 $m^{-2}yr^{-1}$). The cumulative emission values were calculated for cycle 1 and 2 separately, as well as together, using linear interpolation between individual measurements and converted to yearly values. Superscript letters indicate significant differences between sampling locations for cycle 1 and 2 separately, tested with an ANOVA ($p < 0.05$).

Cycle	Loca-tion	CO_2 emission		N_2O emission		CH_4 emission	
		kg C $m^{-2}yr^{-1}$	kg CO_2 $m^{-2}yr^{-1}$	g N $m^{-2}yr^{-1}$	kg CO_2-eq $m^{-2}yr^{-1}$	g C $m^{-2}yr^{-1}$	kg CO_2-eq $m^{-2}yr^{-1}$
1	Ald.	0.640 ± 0.06 [a]	2.35 ± 0.21	4.22 ± 1.66[a]	1.81 ± 0.71	4.47 ± 3.51[a]	0.16 ± 0.13
1	Vli.	0.501 ± 0.03[a]	1.84 ± 0.12	1.89 ± 0.10[a]	0.81 ± 0.04	−0.40 ± 0.09[a]	−0.01 ± 0.003
1	Zeg.	0.614 ± 0.03[a]	2.25 ± 0.10	1.35 ± 0.25[a]	0.58 ± 0.11	0.30 ± 0.17[a]	0.01 ± 0.006
2	Ald.	0.390 ± 0.05[ab]	1.43 ± 0.17	2.14 ± 0.24[ab]	0.92 ± 0.10	−0.31 ± 0.22[a]	−0.01 ± 0.008
2	Vli.	0.327 ± 0.03[a]	1.20 ± 0.10	1.27 ± 0.30[a]	0.54 ± 0.13	−0.68 ± 0.07[a]	−0.02 ± 0.002
2	Zeg.	0.516 ± 0.04[b]	1.89 ± 0.14	2.98 ± 0.31[b]	1.28 ± 0.13	−0.57 ± 0.05[a]	−0.02 ± 0.002
1 + 2	Ald.	0.44 ± 0.04[ab]	1.61 ± 0.13	2.78 ± 0.43[a]	1.19 ± 0.18	1.02 ± 0.76[a]	0.04 ± 0.03
1 + 2	Vli.	0.36 ± 0.02[a]	1.33 ± 0.09	1.54 ± 0.30[a]	0.66 ± 0.13	−0.59 ± 0.07[a]	−0.02 ± 0.003
1 + 2	Zeg.	0.52 ± 0.04[b]	1.91 ± 0.15	2.97 ± 0.41[a]	1.28 ± 0.18	−0.32 ± 0.01[a]	−0.01 ± 0.000

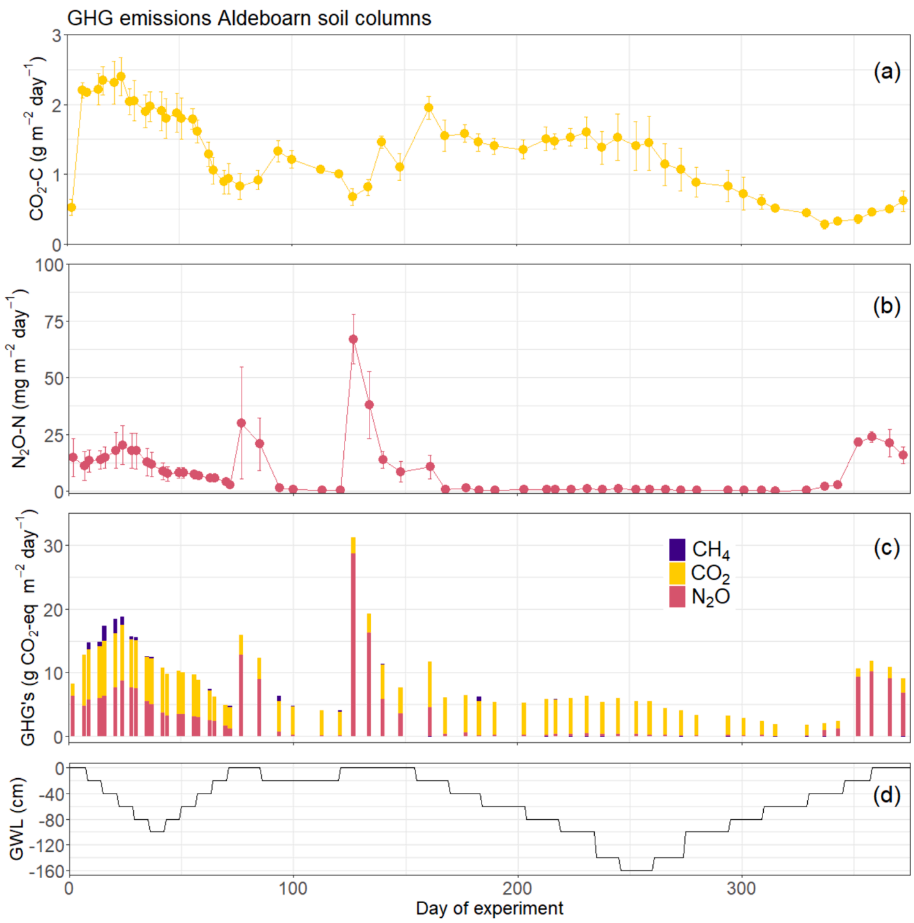

Fig. 2. Average CO_2 (a) and N_2O (b) emissions from Aldeboarn bare soil columns (n = 3), as well as the contributions of CO_2, N_2O and CH_4 to the total CO_2-equivalent emissions (c) during the corresponding GWL treatments (d). Error bars in (a) and (b) indicate the standard error of the mean.

equations for the three locations together. Separating the locations, however, resulted in quadratic or third order polynomial functions for Zegveld and the second rewetting track of the Aldeboarn columns (Table 3). In cycle 1, rewetting started by raising the GWL to −80 cm, at which point mean emissions of 1.8 g C·m^{-2} day^{-1} were observed. In cycle 2, the deepest rewetting level was associated with CO_2-C emissions of 1.2 g C·m^{-2} day^{-1} on average.

Cumulative N_2O emissions ranged between 1.27 and 4.22 g N m^{-2} yr^{-1} (Table 2). Over the whole experiment, the cumulative emissions were highest in the Zegveld and lowest in the Vlist soil cores. Peak emissions occurred particularly after a recent raise of the GWL to 0 cm for all of the locations (Fig. 2b, 3b, 4b). Averaged over all locations and both tracks and cycles, the highest emissions were measured at a GWL of 0 cm, though not significant (Fig. 5b). Strongest emission peaks occurred in response to raising the GWL to 0 cm after the intermediate period at −20 cm (up to 67 mg N m^{-2} day^{-1}). With the exception of the Aldeboarn soils during cycle 1, N_2O emissions were relatively low- below 2 mg N m^{-2} day^{-1} – at GWLs deeper than −20 cm. Mean cumulative emissions were very high in all soils, particularly for Zegveld (2.78 g N $m^{-2}yr^{-1}$) and Aldeboarn (2.97 g N $m^{-2}yr^{-1}$). The cumulative emissions in the Vlist soils were almost two times smaller than the other sites, on average 1.54 g N $m^{-2}yr^{-1}$ (Table 2). Tests with (un-replicated) vegetated soil

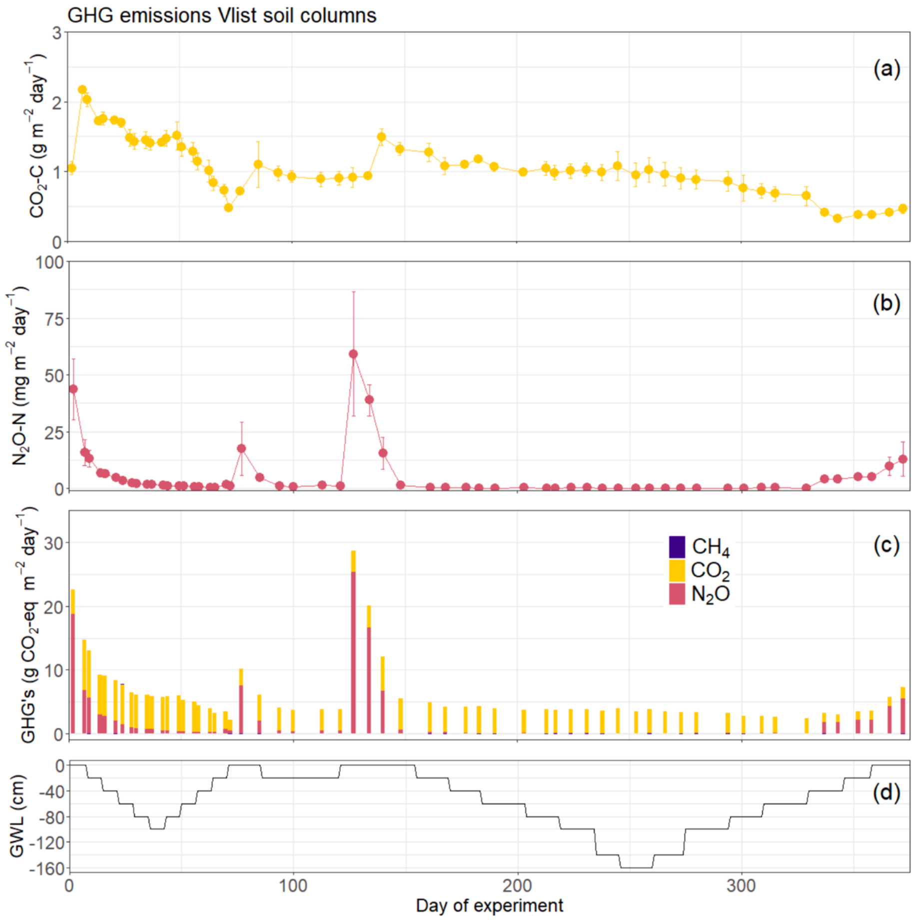

Fig. 3. Average CO_2 (a), N_2O (b) and total CO_2-equivalent emissions (c) from Vlist bare soil columns (n = 3) during the corresponding GWL treatments (d). Error bars in (a) and (b) indicate the standard error of the mean.

cores resulted in much lower emissions (Fig. S5), because mineral N was used for crop uptake and thus less N was left for N_2O production. The highest observed N_2O emission in the vegetated cores was 29.3 mg N m^{-2} day^{-1}, which occurred in the Aldeboarn grass core, after a GWL rise to 0 cm.

Emission or uptake of CH_4 was generally low (Fig. 2c, 3c, 4c), with a few strong peaks, resulting in emission values ranging between −4.7 and 67.2 mg C·m^{-2} day^{-1}. Similarly to the N_2O emissions, there was a period of strong CH_4 increase during the first drying track in the Aldeboarn cores (Fig. 2c). Although this pattern occurred in all three replicates, variation was very high. CH_4 emissions showed no relation with GWL conditions (Fig. 5c). Cumulative fluxes showed strong variation between drying-wetting cycles and ranged between −0.68 and 4.47 g C·m^{-2} yr^{-1} (Table 2).

Over the course of both drying-wetting cycles, the highest GWP (composed of the cumulative emissions of the three GHG's together) was found in the Zegveld soil cores- 3.18 versus 2.84 and 1.98 kg CO_2-eq $m^{-2}yr^{-1}$ for Aldeboarn and Vlist, respectively (Table 2). High mean N_2O and CH_4 cumulative emissions (with large standard errors) raised the total GWP of the Aldeboarn cores in cycle 1. Both in cycles 1 and 2 and for all locations, mean cumulative CO_2 emissions had a larger GWP than the sum of cumulative N_2O and cumulative CH_4 emissions together (Table 2). However, N_2O was the main contributor at shallow GWL depths, resulting in the highest CO_2 equivalent emissions occurring at the 0 cm GWL treatment (Fig. 5d).

3.2. Pore water chemistry

Dissolved organic carbon concentrations in the pore solution ranged between 39 and 645 mg C l^{-1} in cycle 1 and 22 and 712 mg C l^{-1} in cycle 2 (Fig. S7a). At any GWL condition, DOC concentrations were higher when the experiment was in a drying phase, than during the corresponding rewetting phase observation (Fig. S8a). There was no apparent effect of sampling depth or GWL on DOC concentrations. Mean NO_3^- concentrations in the ranges 0–123 mg NO_3-N l^{-1} and 0–220 mg NO_3-N l^{-1} were observed during cycles 1 and 2, respectively. The highest peaks were measured in pore water extracted from the upper 10 cm of the soil. More NO_3^- was available during the rewetting tracks than the drying tracks (Fig. 7b, S8b). Concentrations of NH_4^+ on the other hand, appeared to be higher under waterlogged conditions, although a strong relationship was not detected (Fig. 7c, S8c). Mean values ranged between 0 and 12 or 0 and 4.8 mg NH_4^+-N l^{-1} during cycles 1 and 2 respectively. Mean pH values in the pore water samples ranged between 4.9 and 6.4 in cycle 1 and 4.6 and 6.2 in cycle 2. The pH was higher during drying than during rewetting (Fig. S8d).

3.3. Carbon and soil volume losses

Location markers showed vertical as well as horizontal displacement in response to GWL lowering. This was most pronounced at the soil surface. At the end of drying-wetting cycle two, the columns had not risen back to their original volume again (Fig. 7, Table 4). Instead, the soil surface was located at a height of 0.87 to 1.32 cm below the original elevation 62 days after the final GWL rise up to the surface.

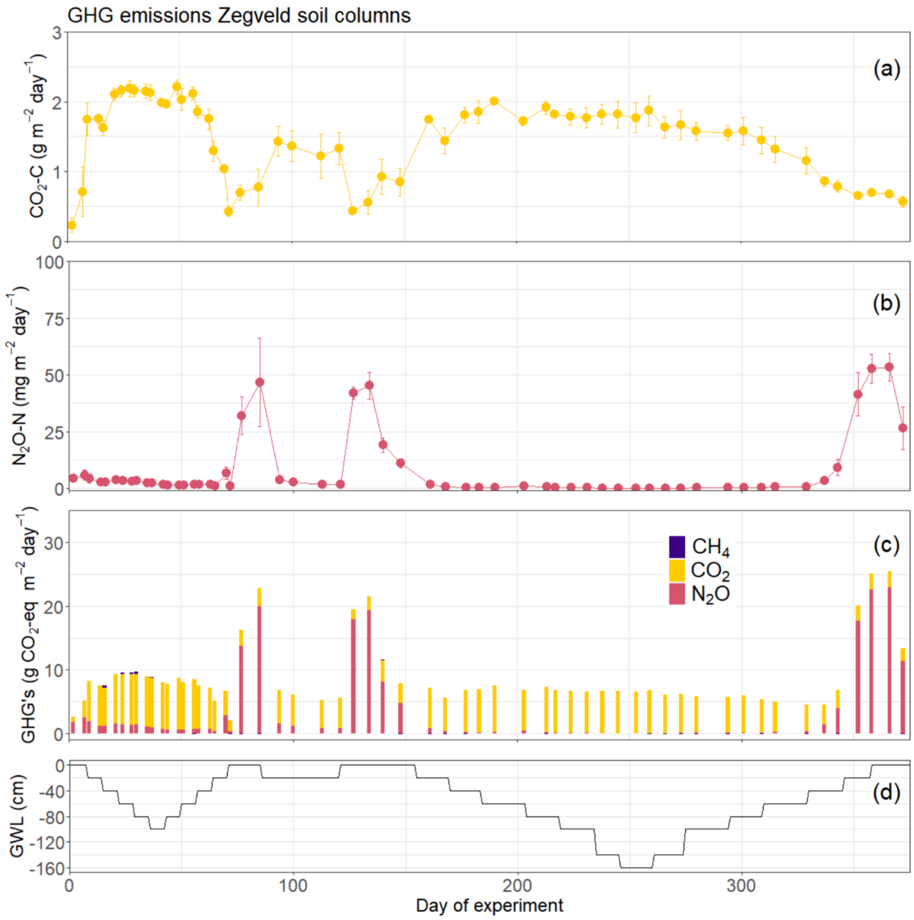

Fig. 4. Average CO_2 (a), N_2O (b) and total CO_2-equivalent emissions (c) from Zegveld bare soil columns (n = 3) during the corresponding GWL treatments (d). Error bars in (a) and (b) indicate the standard error of the mean.

4. Discussion

4.1. Effect of GWL on CO_2 emissions

We hypothesized to find a negative relationship between GWL and CO_2 emissions, meaning higher emissions for deeper groundwater tables. Soil moisture content controls C mineralization by affecting oxygen content as well as substrate mobility and availability (Linn & Doran, 1984; Moyano et al., 2013). Therefore, we had hypothesized to find an optimum GWL for CO_2 emissions, with lower emissions both for deeper and shallower GWL conditions. Our measurement series consisted of two drying tracks and two wetting tracks. While drainage always led to an increase in CO_2 emissions, in accordance with our hypothesis, different response patterns were observed in the four tracks (Fig. 6a, 6c, 6e, Table 3). Other studies that measured CO_2 emissions in the lab under a range of suction treatments have found maximum emissions at a water potential of −50 cm (Kechavarzi et al., 2010), −40 cm (Berglund & Berglund, 2011) or between −20 and −60 cm (Saurich et al., 2019), with emissions decreasing at dryer and wetter conditions. Our expectation was to find a similar parabolic relationship with GWL. We only found this characteristic response in the first drying track, however. There, CO_2 emissions increased with drainage up to a peak at −60 cm on average, after which emission rates decreased with lower GWLs. In this experiment, only a GWL treatment of 0 cm was associated with lower average CO_2 emissions than the other GWL steps (Fig. 5a).

Rewetting of the soil did not result in a CO_2 response resembling the path during GWL lowering. Instead of rising back to their drying-track optima, emission rates decreased further during rewetting (Fig. 6a, 6c, 6e). It was on average associated with a steady, linear decrease in emissions, although individual differences between soil types were found (Table 3). From the drying trajectory we conclude that, on average, GWL lowering increased decomposition rates by enlarging the volume of aerated soil, up to a GWL of 60 cm below the surface. Further drainage, while increasing the volume of aerated soil, increased the volume potentially under drought stress. This may have intensified moisture limitation in the shallow soil to such an extent that the total CO_2 flux was negatively affected by deeper GWLs than −60 cm (Mäkiranta et al., 2009). Raising the GWL, and thereby potentially mediating the moisture limitation on respiration within soil zones subject to capillary rise, did not result in higher CO_2 emission rates. This may be an indication that the top soil layer was the main contributor to the total flux, which is in accordance with findings in other studies that reported stronger respiration rates in top soil than in sub soil samples (Berglund & Berglund, 2011; Saurich et al., 2019). In support of this, the VWC observations suggested that the moisture content of the shallow soil layers was not directly affected by an initial GWL rise, particularly after a prolonged dry period, as seen in cycle 2 (Fig. S4). Above a soil depth of 20 cm, the VWC would often continue to decrease, even during rewetting, until the soil layer was overtaken by the water table again. It is generally known that in peat soils the VWC is subject to hysteresis, which can be substantial (e.g., Dasilva et al., 1993; Schwärzel et al., 2002). Schwärzel et al. (2002) showed that, at the same soil water potentials, the VWC can differ up to 30 volume percentages, with lower VWC during the wetting track. Alternatively, the prolonged drought throughout the drained profile has imposed such a stress on the microbial community, that the recovery of activity and therefore respiration is delayed.

High DOC concentrations were mainly found at the start of the

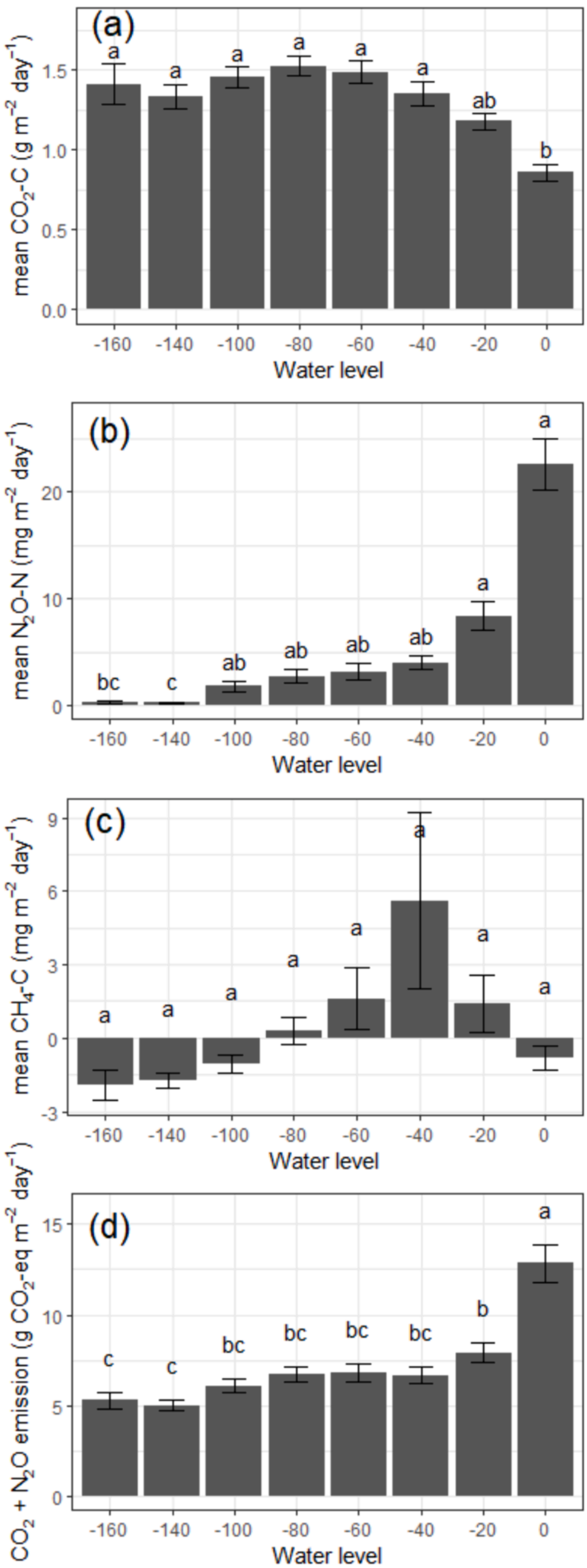

Fig. 5. Mean emissions grouped per GWL treatment, and averaged over locations, drying-wetting cycles and tracks. Letters indicate significant differences using a significance level of alpha = 0.05, following from linear mixed effects models using the experimental objects as random factor and GWL and/or location as fixed factors. Note that the y-axis of (a) uses units of g C·m^{-2} day^{-1}, while fluxes in (d) are expressed in g CO_2-eq. m^{-2} day^{-1}.

experiment, before drying had commenced, and during subsequent drying (Fig. S8a). This pattern indicates increased DOC consumption in response to drying, rather than increased production. The latter would have been reflected in higher DOC concentrations during the rewetting track. However, we can't rule out the possibility that a large portion of the dissolved C left the soil with the outflowing water during drainage. In this study, we did not quantify outgoing fluxes of DOC.

CO_2 emissions were on average lower in the second drying-rewetting cycle, which we attribute to the fast decomposition of labile C compounds in cycle 1. Although the grass sward including the upper 5 cm of the soil had been removed before the start of the experiment, some root material may still have been present.

4.2. *Effect of GWL on N_2O emissions*

Our observations agreed with the hypothesis that N_2O emissions would be strongest under GWLs close to the soil surface, and in particular during soil rewetting (Fig. 2b, 3b, 4c). Lowest cumulative N_2O emissions were measured in the Vlist soil (Table 2), which is surprising, as it had the highest N content in all soil horizons, with the exception of horizon 2, a clay deposit layer (Table 1). During the course of the experiment, strong pulses of N_2O emissions solely occurred after a GWL rise from −20 to 0 cm, or from −40 to −20 cm. These peaks coincided with sudden increases in the top horizons' VWC values (Fig. S4), which suggests that they can mainly be attributed to denitrification in the top soil layers. Denitrification sharply increases when the soil moisture content rises above a certain threshold degree of saturation (Heinen, 2006). Due to the relatively high organic matter content, originating from inputs of plant material and root exudates, the top soil generally provides advantageous conditions for denitrification (Berglund & Berglund, 2011; Van Beek et al., 2004; Velthof & Oenema, 1995). In peat soils, however, organic matter is abundant throughout the vertical profile, making all layers potentially suitable for denitrification (particularly those below the water table) (Van Beek et al., 2004). Still, similarly to mineral soils, in cultivated peat soils a relative abundance of easily degradable C compounds is found in the top soil compared to the sub soil, serving as a limiting factor to denitrification (Bader et al., 2018; Koops et al., 1996). Additionally, N_2O produced at greater depth will face a longer travel time and distance to diffuse up towards the surface, during which it may be reduced to N_2 (Koops et al., 1997). Therefore, the large peaks of N_2O in our experiment may be best explained by incomplete denitrification in the top soil.

In finding higher emissions under wet conditions, our observations differ from some studies, where a mitigation in N_2O emissions was found in rewetted peat meadows or undrained peatlands, compared to drained fields (Martikainen et al., 1993; Regina et al., 1996; Regina et al., 2015; Van Beek et al., 2010). This is not surprising, since our experiment looked at GWL dynamics, while some of the above studies involved permanent pristine peatland (Martikainen et al., 1993; Regina et al., 1996). Regina et al. (2015) included only drainage levels in their study that would still allow for grass cultivation and identified their most shallow drainage treatment of 30 cm depth as the best in terms of GHG mitigation. In other cases, both in field and lab experiments, N_2O emissions were quite low in general, or even negative (Dinsmore et al., 2009; Hendriks et al., 2007; Nielsen et al., 2023; Van de Riet et al., 2013; Wilson et al., 2016). In bare peat cores in an outdoor setup, Nielsen et al. (2023) found 72 % lower emissions at a −5 cm GWL than at −40 cm, but emissions were low in general. Peak emissions measured in their bare soil cores were a factor 1000 lower than our peaks, despite their soils having comparable N contents and being subject to precipitation. In a drought experiment on a minerotrophic fen, Goldberg et al. (2010) did observe strong N_2O emission pulses during the weeks after rewetting. Dinsmore et al. (2009) observed N_2O peaks in response to both drainage and rewetting of their soil cores. Some other cases have been reported of increased N_2O emissions under higher GWL conditions (Berglund & Berglund, 2011; Norberg et al., 2021).

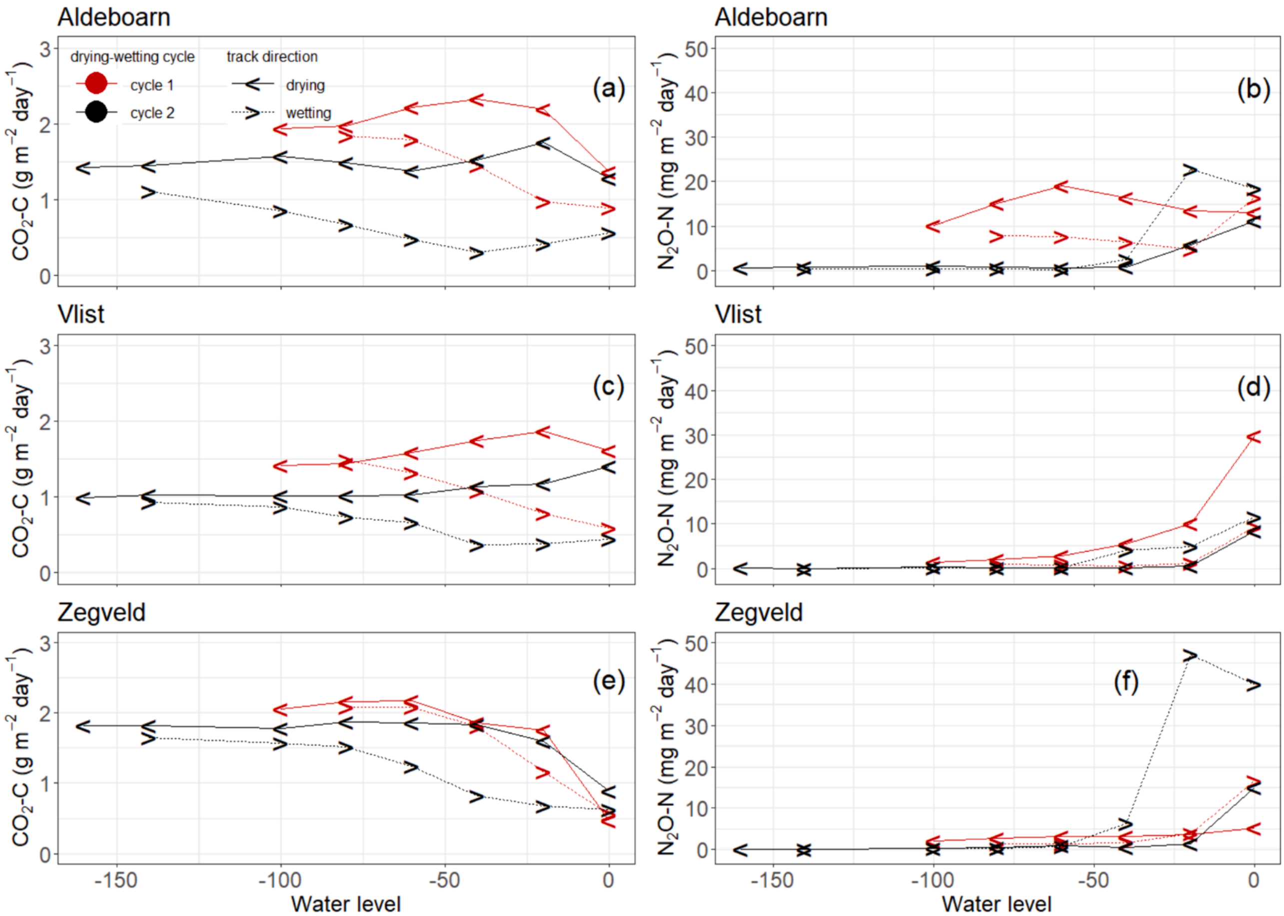

Fig. 6. Mean CO_2 (left) and N_2O (right) emissions released from bare soil columns from Aldeboarn (a, b), Vlist (c, d) and Zegveld (e, f) against the applied GWL. Each point is the mean of three replicates that were measured twice, each, during a GWL step (n = 6). The intermediate period between the two drying wetting cycles is not included here. The two drying-wetting cycles are indicated in different colours and the direction of the GWL change (drying or wetting) by shape and line-type.

A positive relationship between drainage and N_2O emissions is generally attributed to mineralization of organic nitrogen in the peat SOM. Since our experiment involved unvegetated, unfertilized soil cores, mineralization will have been the main N source to N_2O production as well. Indeed, NH_4^+ concentrations in pore water samples had decreased and NO_3^- concentrations had increased by the time of rewetting, compared to the drying track concentrations (Fig.s S8b, S8c), indicating subsequent mineralization and nitrification under the preceding aerobic conditions. The main contrast with other studies is the magnitude of the N_2O pulses which we measured after a GWL rise to −20 or 0 cm, which could be as high as 67 mg N m^{-2} day^{-1}. In the vegetated cores, of which we had sampled one replicate per location, N_2O peaks were generally lower, which is due to plant N uptake (Schimel & Bennett, 2004). Here, N_2O peaks occurred up to 29.3 mg N m^{-2} day^{-1}, and followed similar patterns as the bare core fluxes (Fig. S5).

Our flux rates showed more resemblance with N_2O responses to precipitation events in field experiments on peat meadows (Berendt et al., 2023; Van Beek et al., 2010; Velthof et al., 1996a; Velthof & Oenema, 1995). Velthof et al. (1996a) have illustrated the effect that a GWL rise due to precipitation can have on N_2O production in the soil layers above. This was particularly the case in the top layer, during a shallow GWL. In our experiment, we have consistently seen strong N_2O emissions after a GWL increase from −20 to 0 cm. A GWL rise to 0 cm did not completely saturate the top layer yet, as was evident from the VWC (Fig. S4). Air pockets may remain in the peat soil due to hysteresis (Schwärzel et al., 2002), resulting in a lower degree of saturation and intermittent aerobic and anaerobic conditions, which are optimal for N_2O production during incomplete denitrification (Wagner-Riddle et al., 2020). In the field, maintaining a GWL at −20 cm may pose a risk of such hot spot events, where the previously aerated top layer is rapidly overtaken by the water table due to a heavy rainfall event. Although these events may be temporary and infrequent, the large N_2O peaks brought forth by a sudden shift in the GWL can contribute significantly to the total GHG emissions (Anthony & Silver, 2021). Such a risk will be much smaller at a permanent GWL above 0 cm, where mineralization and nitrification are limited (Taft et al., 2018). In practice a peat meadow is rarely a homogeneous level surface however, complicating the achievement of a stable GWL and prevention of N_2O peaks.

This experiment did not take the effect of N input through fertilizers into account, while fertilizer application is a common practice on peat meadows cultivated for dairy farming. Fertilizer events are potential key moments for N_2O emissions and the fertilizer type and soil moisture content are determining factors in the emission size (Velthof et al., 1996b). Therefore, the interaction with fertilizer amendment should be taken into account in the consideration of a GWL rise in peat meadow soils.

4.3. GHG balance

In accordance with our hypothesis, CO_2 was, with 56.8 to 67.6 %, the main contributor to the net cumulative GWP over the entire measurement series (Table 2). This result can be attributed to GWL treatments deeper than 0 cm (Fig. 5a). At a GWL of 0 cm, however, N_2O was the main contributor to total GHG emissions (Fig. 2c, 3c, 4c, 5). The cumulative N_2O emissions contributed 33.5 to 41.9 % to the net cumulative GWP. Cumulative emissions of CH_4 contributed only 1.3 % to total

Table 3
Least squares coefficient estimates describing the relationships between the CO_2 emission rates (g CO_2-C $m^{-2}d^{-1}$) and the GWL (m) at the time of measuring, for GWL levels within the ranges of −1.0 to 0 m for cycle 1 and −1.6 to 0 m for cycle 2.

Location	Cycle	Direction	Model	Adjusted R^2
Aldeboarn	1	Drying	$CO_2 = 1.3 - 6.0GWL - 11GWL^2 - 5.6GWL^3$	0.44
	1	Wetting	$CO_2 = 0.84$—$1.4GWL$	0.57
	2	Drying	n.s.	–
	2	Wetting	$CO_2 = 0.57 + 1.6GWL+3.1GWL^2 + 1.2GWL^3$	0.48
Vlist	1	Drying	$CO_2 = 1.6 - 2.0GWL - 5.4GWL^2 - 3.1GWL^3$	0.57
	1	Wetting	$CO_2 = 0.59 - 1.2GWL$	0.77
	2	Drying	$CO_2 = 1.4 + 0.66GWL+0.28GWL^2$	0.31
	2	Wetting	$CO_2 = 0.35 - 0.45GWL$	0.54
Zegveld	1	Drying	$CO_2 = 0.63 - 4.7GWL - 3.3GWL^2$	0.83
	1	Wetting	$CO_2 = 0.52 - 4.2GWL - 2.8GWL^2$	0.93
	2	Drying	$CO_2 = 0.94 - 3.6GWL - 4.2GWL^2 - 1.4GWL^3$	0.61
	2	Wetting	$CO_2 = 0.60 - 7*10^{-4}GWL+2.3GWL^2 + 1.3GWL^3$	0.78
All locations	1	Drying	$CO_2 = 1.17$–$5.04GWL - 8.77GWL^2 - 4.38GWL^3$	0.33
	1	Wetting	$CO_2 = 0.72$—$1.50GWL$	0.61
	2	Drying	n.s.	–
	2	Wetting	$CO_2 = 0.43 - 0.60GWL$	0.36

CO_2-equivalent emissions from the Aldeboarn soils, while cumulative CH_4 uptake led to a net 1.1 % or 0.4 % abatement of the summed CO_2 and N_2O emissions in the Vlist and Zegveld soils, respectively. The relatively low contribution of CH_4 agreed with our hypothesis. However, this study did not include the effect of long term inundation on CH_4 emissions.

Although the relation between oxidation and soil moisture content or groundwater conditions has been studied often using static treatments in a factorial design or in stepwise drying schemes, studies on the response to rewetting are lacking. In this experiment, we observe an effect of the direction of GWL change on oxidation dynamics. Our results suggest that the moisture conditions in the top 40 cm, rather than those in the subsoil, are a major factor for the total CO_2 and N_2O emission. In a field situation, top soil conditions are largely affected by fertilization, grazing and rainfall events, factors that were not included in this experiment. To understand the dynamics contributing to total peat oxidation and subsequent GHG emissions in a peat meadow soil, we would recommend to study grass growth, fertilization, grazing and/or weather effects in interaction with shallow (−20 to 0 cm) GWL treatments.

4.4. *Volume loss due to oxidation*

A vertical displacement of 0.87 to 1.32 cm was observed after two drying-rewetting cycles (Fig. 7, Table 4). While we used the surface level elevation measured 427 days after the start of the experiment (we

Table 4
Mean surface subsidence in the bare soil cores and the volume loss of the top horizon in vertical and horizontal direction. The potential carbon loss is calculated as the amount of carbon in the lost volume and compared to the carbon loss derived through interpolation of flux observations (cumulative CO_2-C emission). The proportion of volume loss that can be attributed to oxidation is the ratio of the actual and the potential C loss.

	Aldeboarn	Vlist	Zegveld
Top horizon depth in field (cm)	0 to 12	0 to 15	0 to 30
Subsidence (cm)	0.917	0.873	1.32
Horizontal shrinkage (cm, change in surface radius)	1.87	1.25	2.30
Subsidence induced volume loss (m^3 m^{-2} yr^{-1})*	0.00917	0.00873	0.0132
Subsidence induced potential C loss (kg m^{-2} yr^{-1})	0.560	0.555	0.795
Measured C loss (kg m^{-2} yr^{-1})	0.440	0.364	0.522
Oxidation component subsidence (%)	78.6	65.6	65.6

*Volume loss of top horizon is expressed as m^3 per m^2 of initial soil surface (before horizontal shrinkage).

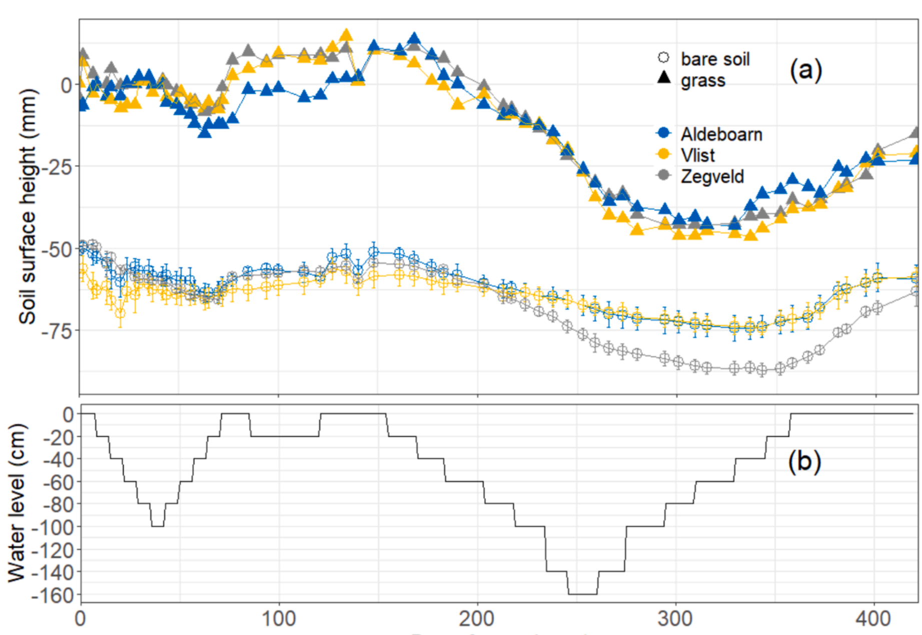

Fig. 7. Soil surface movement of bare (n = 3) and vegetated (grass) soil cores (n = 1) over the course of two drying wetting cycles. The reference heights, 0 mm for grass cores and −50 mm for bare cores, are chosen relative to the soil surface in the field: the top 5 cm of the soil was removed before sampling the bare cores.

continued volume measurements for two more months after the end of cycle 2, to allow the soil to swell at a constant GWL of 0 cm), we consider the subsidence as a yearly rate, as the time span encompassed one period resembling summer drought conditions. We tried to estimate the relative contribution of oxidation to the total volume loss, using a function relating volume loss to C loss, and comparing the result to our measured C losses (cumulative CO_2-C emissions per year). A conventional way to estimate the C loss (kg C $ha^{-1}yr^{-1}$) in the absence of (sufficient) emission observations is to use yearly subsidence observations (Hiraishi et al., 2014). Carbon emissions can be calculated based on subsidence and peat properties, according to (e.g., Van den Akker et al., 2008):

$$C_{loss} = S \bullet F_{ox} \bullet \rho_{peat} \bullet SOC \bullet 10^4 \quad (1)$$

Where, C_{loss} is the carbon loss (kg C ha^{-1} yr^{-1}), S is subsidence rate (m yr^{-1}), F_{ox} is the fraction of subsidence caused by oxidation of organic matter (−), ρ_{peat} is bulk density (kg m^{-3}), SOC is soil organic carbon content of the peat (kg C kg^{-1} soil), and 10^4 is a unit conversion factor (m^2 ha^{-1}). Van den Akker et al. (2008) proposed that a constant F_{ox} value of 1 could be taken, when using the ρ_{peat} and SOC values of the reduced, fibric peat below the groundwater table. The reasoning behind this approach is that, in peat grasslands with controlled GWLs, a steady state situation is created where the periodically lowering GWL exposes new layers of fibric peat to permanent aerobic conditions. This approach has been applied in the Dutch national Greenhouse gas inventory, to estimate annual C losses from drained peat soils (Arets et al., 2022). Following this relationship, by using the ρ_{peat} and SOC values of the deepest horizons in our soil cores, the subsidence values should correspond with a C_{loss} of 0.56 to 0.80 kg C $m^{-2}yr^{-1}$ (Table 4). These potential C-losses are higher than the actual measured CO_2-C emission, and the actual fraction of volume loss that could be attributed to oxidation (F_{ox}) was, therefore, estimated to range between 66 and 79 % (Table 4). Other studies have reported similar relative contributions of oxidation to volume loss (Ishikura et al., 2018; Leifeld et al., 2011; Schothorst, 1977; Wosten et al., 1997). Our mean subsidence observations approximate typical subsidence rates, which are reported to range between 1 and 15 mm per year in temperate peat meadows (Hoogland et al., 2012; Koster et al., 2018; Leifeld et al., 2011; Schothorst, 1977; Van den Akker et al., 2008; Van der Meulen et al., 2007). The surface of the Zegveld cores subsided stronger than those in the Vlist and Aldeboarn cores, which both contained a 16 to 22 cm clay layer just below the top soil (Table 1). This observation agrees with the suggestion by Van den Akker et al. (2008) of an inhibiting effect by a clay cover on subsidence.

The estimates did not include the horizontal volume losses, which occurred in particular in the highest soil layers. At the last recording, three weeks after the last GWL rise, the radiuses of the surface areas were still 1.25 to 2.30 cm smaller than at the start of the experiment (Table 4). Considering this, we can assume that the reported volume losses are actually underestimations, and the oxidation components to volume loss are overestimations.

5. Conclusion

In this study, we manipulated GWL conditions in incubated, undisturbed bare peat cores from three peat areas in the Netherlands. Over a period of 370 days, we applied two drying-rewetting cycles on the cores and measured surface emissions of CO_2, N_2O and CH_4, as well as solute pore water concentrations, VWC and matric potential throughout the vertical profile and volume change at the surface. In accordance with our hypothesis, cumulative total greenhouse gas emissions over the course of the experiment were dominated by CO_2. Raising the GWL to −20 or 0 cm, however, resulted in strong N_2O emissions. Due to these peaks, the 0 cm GWL step was associated with highest N_2O emissions and the highest CO_2-eq sum of CO_2 and N_2O. A small number of peak N_2O emission events provided a substantial contribution to total CO_2-equivalent emissions. CH_4 emissions or uptake was small and showed no relation with the GWL, contrary to our hypothesis. We confirmed our hypothesis that CO_2 emissions increase with drainage, and observed on average the highest emissions at a GWL of −80 cm. Relationships between the GWL and CO_2 emissions were different for drying and rewetting pathways, however, indicating hysteresis in the WFPS and the oxidation process, and suggesting that the upper 40 cm of the soil was the largest contributor to CO_2 emissions. Similarly, the high N_2O peaks after a GWL rise to −20 or 0 cm, may indicate that the conditions in the top 40 cm are most important to N_2O production. In accordance with our hypothesis, the results suggest that a GWL of 20 cm constitutes the risk for large N_2O pulse emissions during rewetting of the soil. This is a potential risk in the field, were soil moisture in the top soil can be highly variable due to precipitation events. A GWL of 0 cm would be preferential in terms of mitigation of both CO_2 and N_2O emissions, but the continuation of current dairy farming practices would demand less conditions. Further research is therefore needed to study the interactions between the potential GWL scenario's and nitrogen input through fertilizers and grazing.

CRediT authorship contribution statement

Erne Blondeau: Writing – original draft, Visualization, Methodology, Investigation, Formal analysis, Conceptualization. **Gerard L. Velthof:** Writing – review & editing, Supervision, Conceptualization. **Marius Heinen:** Writing – review & editing, Supervision, Formal analysis. **Rob F.A. Hendriks:** Writing – review & editing, Methodology, Investigation, Conceptualization. **Anneke Stam:** Methodology, Investigation. **Jan J.H. van den Akker:** Writing - Review & Editing, Conceptualization. **Monne Weghorst:** Methodology, Investigation. **Jan Willem van Groenigen:** Writing – review & editing, Supervision, Conceptualization.

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.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.geoderma.2024.117043.

References

Anthony, T.L., Silver, W.L., 2021. Hot moments drive extreme nitrous oxide and methane emissions from agricultural peatlands. Glob. Chang. Biol. 27 (20), 5141–5153. https://doi.org/10.1111/gcb.15802.

Arets, E. J. M. M., van Baren, S. A., Kramer, H., Lesschen, J. P., & Schelhaas, M. J. (2022). *Greenhouse gas reporting of the LULUCF sector in the Netherlands; Methodological background, update 2022.* Wettelijke Onderzoekstaken Natuur & Milieu, WOt-technical report 217.

Bader, C., Muller, M., Schulin, R., Leifeld, J., 2018. Peat decomposability in managed organic soils in relation to land use, organic matter composition and temperature. Biogeosciences 15 (3), 703–719. https://doi.org/10.5194/bg-15-703-2018.

Baggs, E.M., Philippot, L., 2010. Microbial terrestrial pathways to nitrous oxide. In: Smith, K.A. (Ed.), Nitrous Oxide and Climate Change. London, UK, Earthscan.

Berendt, J., Jurasinski, G., Wrage-Mönnig, N., 2023. Influence of rewetting on N2O emissions in three different fen types. Nutr. Cycl. Agroecosyst. 125 (2), 277–293. https://doi.org/10.1007/s10705-022-10244-y.

Berglund, O., Berglund, K., 2011. Influence of water table level and soil properties on emissions of greenhouse gases from cultivated peat soil. Soil Biol. Biochem. 43 (5), 923–931. https://doi.org/10.1016/j.soilbio.2011.01.002.

Boonman, J., Hefting, M.M., van Huissteden, C.J.A., van den Berg, M., van Huissteden, J., Erkens, G., van der Velde, Y., 2022. Cutting peatland CO2 emissions with water management practices. Biogeosciences 19 (24), 5707–5727. https://doi.org/10.5194/bg-19-5707-2022.

Butterbach-Bahl, K., Baggs, E.M., Dannenmann, M., Kiese, R., Zechmeister-Boltenstern, S., 2013. Nitrous oxide emissions from soils: how well do we understand the processes and their controls? Philos. Trans. R. Soc. B-Bio. Sci. 368 (1621). https://doi.org/10.1098/rstb.2013.0122.

Canadell, J. G., Monteiro, P. M. S., Costa, M. H., Cotrim da Cunha, L., Cox, P. M., Eliseev, A. V., . . . Zickfeld, K. (2021). *Global Carbon and other Biogeochemical Cycles and Feedbacks.* Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA, pp. 673–816, doi:10.1017/9781009157896.007.: In Climate Change 2021: The Physical Science Basis. Contribution of Working Group I to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change [Masson-Delmotte, V., P. Zhai, A. Pirani, S.L. Connors, C. Péan, S. Berger, N. Caud, Y. Chen, L. Goldfarb, M.I. Gomis, M. Huang, K. Leitzell, E. Lonnoy, J.B.R. Matthews, T.K. Maycock, T. Waterfield, O. Yelekçi, R. Yu, and B. Zhou (eds.)].

Dasilva, F.F., Wallach, R., Chen, Y., 1993. Hydraulic properties of sphagnum peat moss and tuff (scoria) and their potential effects on water availability. Plant and Soil 154 (1), 119–126. https://doi.org/10.1007/bf00011080.

Dinsmore, K.J., Skiba, U.M., Billett, M.F., Rees, R.M., 2009. Effect of water table on greenhouse gas emissions from peatland mesocosms. Plant and Soil 318 (1–2), 229–242. https://doi.org/10.1007/s11104-008-9832-9.

Erkens, G., van der Meulen, M.J., Middelkoop, H., 2016. Double trouble: subsidence and CO2 respiration due to 1,000 years of Dutch coastal peatlands cultivation. Hydrgeol. J. 24 (3), 551–568. https://doi.org/10.1007/s10040-016-1380-4.

Evans, C., Peacock, M., Baird, A., Artz, R., Burden, A., Callaghan, N., Morrison, R., 2021. Overriding water table control on managed peatland greenhouse gas emissions. Nature 593, 548–552. https://doi.org/10.1038/s41586-021-03523-1.

Firestone, M.K., Davidson, E.A., 1989. Microbiological basis of NO and N2O production and consumption in soil. In: Andreae, M.O., Schimel, D.S. (Eds.), Exchange of Trace Gases between Terrestrial Ecosystems and the Atmosphere. John Wiley & Sons Ltd, New York, pp. 7–21.

Forster, P., Storelvmo, T., Armour, K., Collins, W., Dufresne, J.-L., Frame, D., . . . Zhang, H. (2021). *The Earth's Energy Budget, Climate Feedbacks, and Climate Sensitivity.* Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA, pp. 923–1054, doi:10.1017/9781009157896.009: In Climate Change 2021: The Physical Science Basis. Contribution of Working Group I to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change [Masson-Delmotte, V., P. Zhai, A. Pirani, S.L. Connors, C. Péan, S. Berger, N. Caud, Y. Chen, L. Goldfarb, M.I. Gomis, M. Huang, K. Leitzell, E. Lonnoy, J.B.R. Matthews, T.K. Maycock, T. Waterfield, O. Yelekçi, R. Yu, and B. Zhou (eds.)].

Fuss, R. (2023). gasfluxes: Greenhouse Gas Flux Calculation from Chamber Measurements. https://CRAN.R-project.org/package=gasfluxes: R package version 0.5.

Goldberg, S.D., Knorr, K.H., Blodau, C., Lischeid, G., Gebauer, G., 2010. Impact of altering the water table height of an acidic fen on N2O and NO fluxes and soil concentrations. Glob. Chang. Biol. 16 (1), 220–233. https://doi.org/10.1111/j.1365-2486.2009.02015.x.

Heinen, M., 2006. Simplified denitrification models: Overview and properties. Geoderma 133 (3–4), 444–463. https://doi.org/10.1016/j.geoderma.2005.06.010.

Hendriks, D.M.D., van Huissteden, J., Dolman, A.J., van der Molen, M.K., 2007. The full greenhouse gas balance of an abandoned peat meadow. Biogeosciences 4 (3), 411–424. https://doi.org/10.5194/bg-4-411-2007.

Hiraishi, T., Krug, T., Tanabe, K., Srivastava, N., Baasansuren, J., Fukuda, M., Troxler, T. G., 2014. 2013 Supplement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories: Wetlands. IPCC, Switzerland.

Hoogland, T., van den Akker, J.J.H., Brus, D.J., 2012. Modeling the subsidence of peat soils in the Dutch coastal area. Geoderma 171, 92–97. https://doi.org/10.1016/j.geoderma.2011.02.013.

Hoving, I.E., van Riel, J.W., Massop, H.T.L., van Houwelingen, K., Bijman, M., 2022. Waterinfiltratie met drukdrains en greppels voor veenbehoud en emissiereductie; Deelonderzoek 'natte veeteelt' in het Innovatie Programma Veen (2018–2021). Wageningen Livestock Research, Openbaar Rapport, p. 1345.

Ishikura, K., Hirano, T., Okimoto, Y., Hirata, R., Kiew, F., Melling, L., Ishii, Y., 2018. Soil carbon dioxide emissions due to oxidative peat decomposition in an oil palm plantation on tropical peat. Agr Ecosyst Environ 254, 202–212. https://doi.org/10.1016/j.agee.2017.11.025.

Jobbagy, E.G., Jackson, R.B., 2000. The vertical distribution of soil organic carbon and its relation to climate and vegetation. Ecol. Appl. 10 (2), 423–436. https://doi.org/10.2307/2641104.

Karki, S., Elsgaard, L., Kandel, T.P., Laerke, P.E., 2016. Carbon balance of rewetted and drained peat soils used for biomass production: a mesocosm study. Glob. Change Biol. Bioenergy. 8 (5), 969–980. https://doi.org/10.1111/gcbb.12334.

Kechavarzi, C., Dawson, Q., Bartlett, M., Leeds-Harrison, P.B., 2010. The role of soil moisture, temperature and nutrient amendment on CO2 efflux from agricultural peat soil microcosms. Geoderma 154 (3–4), 203–210. https://doi.org/10.1016/j.geoderma.2009.02.018.

Koops, J.G., Oenema, O., Van Beusichem, M.L., 1996. Denitrification in the top and sub soil of grassland on peat soils. Plant and Soil 184 (1), 1–10. https://doi.org/10.1007/bf00029269.

Koops, J.G., Van Beusichem, M.L., Oenema, O., 1997. Nitrogen loss from grassland on peat soils through nitrous oxide production. Plant and Soil 188 (1), 119–130. https://doi.org/10.1023/a:1004252012290.

Koster, K., Stafleu, J., Stouthamer, E., 2018. Differential subsidence in the urbanised coastal-deltaic plain of the Netherlands. Netherlands Journal of Geosciences-Geologie En Mijnbouw 97 (4), 215–227. https://doi.org/10.1017/njg.2018.11.

Laiho, R., 2006. Decomposition in peatlands: Reconciling seemingly contrasting results on the impacts of lowered water levels. Soil Biol. Biochem. 38 (8), 2011–2024. https://doi.org/10.1016/j.soilbio.2006.02.017.

Leifeld, J., Müller, M., Fuhrer, J., 2011. Peatland subsidence and carbon loss from drained temperate fens. Soil Use Manag. 27 (2), 170–176. https://doi.org/10.1111/j.1475-2743.2011.00327.x.

Linn, D.M., Doran, J.W., 1984. Effect of water-filled pore space on carbon dioxide and nitrous oxide production in tilled and nontilled soils. Soil Sci. Soc. Am. J. 48 (6), 1267–1272. https://doi.org/10.2136/sssaj1984.03615995004800060013x.

Mäkiranta, P., Laiho, R., Fritze, H., Hytonen, J., Laine, J., Minkkinen, K., 2009. Indirect regulation of heterotrophic peat soil respiration by water level via microbial community structure and temperature sensitivity. Soil Biol. Biochem. 41 (4), 695–703. https://doi.org/10.1016/j.soilbio.2009.01.004.

Martikainen, P.J., Nykanen, H., Crill, P., Silvola, J., 1993. Effect of a lowered water table on nitrous oxide fluxes from northern peatlands. Nature 366 (6450), 51–53. https://doi.org/10.1038/366051a0.

Moore, T.R., Dalva, M., 1993. The influence of temperature and water table position on carbon dioxide and methane emissions from laboratory columns of peatland soils. J. Soil Sci. 44 (4), 651–664. https://doi.org/10.1111/j.1365-2389.1993.tb02330.x.

Moyano, F.E., Manzoni, S., Chenu, C., 2013. Responses of soil heterotrophic respiration to moisture availability: An exploration of processes and models. Soil Biol. Biochem. 59, 72–85. https://doi.org/10.1016/j.soilbio.2013.01.002.

Nielsen, C.K., Elsgaard, L., Jorgensen, U., Lærke, P.E., 2023. Soil greenhouse gas emissions from drained and rewetted agricultural bare peat mesocosms are linked to geochemistry. Sci. Total Environ. 896. https://doi.org/10.1016/j.scitotenv.2023.165083.

NOBV. (2021). *Nationaal onderzoeksprogramma Broeikasgassen Veenweiden (NOBV) Operationele jaarrapportage 2020-2021.*

Norberg, L., Hellman, M., Berglund, K., Hallin, S., Berglund, Ö., 2021. Methane and nitrous oxide production from agricultural peat soils in relation to drainage level and abiotic and biotic factors. Front. Environ. Sci. 9. https://doi.org/10.3389/fenvs.2021.631112.

Pinheiro, J., Bates, D., & Team, R. C. (2023). nlme: Linear and Nonlinear Mixed Effects Models. https://CRAN.R-project.org/package=nlme: R package version 3.1-164.

R Core Team. (2023). R: A Language and Environment for Statistical Computing. Vienna, Austria. https://www.R-project.org/: R Foundation for Statistical Computing.

Regina, K., Nykanen, H., Silvola, J., Martikainen, P.J., 1996. Fluxes of nitrous oxide from boreal peatlands as affected by peatland type, water table level and nitrification capacity. Biogeochemistry 35 (3), 401–418. https://doi.org/10.1007/bf02183033.

Regina, K., Sheehy, J., Myllys, M., 2015. Mitigating greenhouse gas fluxes from cultivated organic soils with raised water table. Mitig. Adapt. Strat. Glob. Chang. 20 (8), 1529–1544. https://doi.org/10.1007/s11027-014-9559-2.

RIVM. (2022). *Greenhouse gas emissions in the Netherlands 1990–2020, National Inventory Report 2022.* Bilthoven, the Netherlands.

Saurich, A., Tiemeyer, B., Dettmann, U., Don, A., 2019. How do sand addition, soil moisture and nutrient status influence greenhouse gas fluxes from drained organic soils? Soil Biol. Biochem. 135, 71–84. https://doi.org/10.1016/j.soilbio.2019.04.013.

Schimel, J.P., Bennett, J., 2004. Nitrogen mineralization: Challenges of a changing paradigm. Ecology 85 (3), 591–602. https://doi.org/10.1890/03-8002.

Schothorst, C.J., 1977. Subsidence of low moor peat soils in western Netherlands. Geoderma 17 (4), 265–291. https://doi.org/10.1016/0016-7061(77)90089-1.

Schwärzel, K., Renger, M., Sauerbrey, R., Wessolek, G., 2002. Soil physical characteristics of peat soils. J. Plant Nutr. Soil Sci. 165 (4), 479–486. https://doi.org/10.1002/1522-2624(200208)165:4<479::aid-jpln479>3.0.co;2-8.

Taft, H.E., Cross, P.A., Jones, D.L., 2018. Efficacy of mitigation measures for reducing greenhouse gas emissions from intensively cultivated peatlands. Soil Biol. Biochem. 127, 10–21. https://doi.org/10.1016/j.soilbio.2018.08.020.

Tiemeyer, B., Borraz, E.A., Augustin, J., Bechtold, M., Beetz, S., Beyer, C., Zeitz, J., 2016. High emissions of greenhouse gases from grasslands on peat and other organic soils. Glob. Chang. Biol. 22 (12), 4134–4149. https://doi.org/10.1111/gcb.13303.

Van Asselen, S. v., Erkens, G., Jansen, S., Aben, R., Hessel, R., Akker, J. J. H. v. d., . . . Gerritsen, P. (2023). *Effects of subsurface water infiltration systems on phreatic groundwater levels in peat meadows.* NOBV.

Van Beek, C.L., Hummelink, E.W.J., Velthof, G.L., Oenema, O., 2004. Denitrification rates in relation to groundwater level in a peat soil under grassland. Biol. Fertil. Soils 39 (5), 329–336. https://doi.org/10.1007/s00374-003-0685-3.

Van Beek, C.L., Pleijter, M., Jacobs, C.M.J., Velthof, G.L., van Groenigen, J.W., Kuikman, P.J., 2010. Emissions of N2O from fertilized and grazed grassland on organic soil in relation to groundwater level. Nutr. Cycl. Agroecosyst. 86 (3), 331–340. https://doi.org/10.1007/s10705-009-9295-2.

Van de Riet, B. P., Hefting, M. M., & Verhoeven, J. T. A. (2013). Rewetting drained peat meadows: risks and benefits in terms of nutrient release and greenhouse gas exchange. *Water Air and Soil Pollution, 224*(4). doi:10.1007/s11270-013-1440-5.

Van den Akker, J. J. H., Kuikman, P. J., de Vries, F., Hoving, I., Pleijter, M., Hendriks, R. F. A., . . . Kwakernaak, C. (2008). *Emission of CO2 from agricultural peat soils in the Netherlands and ways to limit this emission.* Paper presented at the the 13th International Peat Congress After Wise Use – The Future of Peatlands, Vol. 1 Oral Presentations, Tullamore, Ireland.

Van den Born, G.J., Kragt, F., Henkens, D., Rijken, B., Van Bemmel, B., Van der Sluis, S., 2016. Dalende bodem, stijgende kosten. PBL, The Hague.

Van den Pol-van Dasselaar, A., van Beusichem, M.L., Oenema, O., 1997. Effects of grassland management on the emission of methane from intensively managed grasslands on peat soil. Plant and Soil 189 (1), 1–9. https://doi.org/10.1023/a:1004219522404.

Van der Meulen, M.J., van der Spek, A.J.F., de Lange, G., Gruijters, S., van Gessel, S.F., Nguyen, B.L., Krogt, R., 2007. Regional sediment deficits in the Dutch lowlands: Implications for long-term land-use options. J. Soil. Sediment. 7 (1), 9–16. https://doi.org/10.1065/jss2006.12.199.

Ministerie van Landbouw, N. e. V. (2019). *Klimaatakkoord- C4 Landbouw en Landgebruik*. Den Haag.

Velthof, G. L., & Oenema, O. (1995). Nitrous oxide fluxes from grassland in the Netherlands: 2. Effects of soil type, nitrogen fertilizer application and grazing. *European Journal of Soil Science, 46*(4), 541-549. doi:10.1111/j.1365-2389.1995.tb01350.x.

Velthof, G.L., Brader, A.B., Oenema, O., 1996a. Seasonal variations in nitrous oxide losses from managed grasslands in The Netherlands. Plant and Soil 181 (2), 263–274. https://doi.org/10.1007/bf00012061.

Velthof, G.L., Oenema, O., Postma, R., Van Beusichem, M.L., 1996b. Effects of type and amount of applied nitrogen fertilizer on nitrous oxide fluxes from intensively managed grassland. Nutr. Cycl. Agroecosyst. 46 (3), 257–267. https://doi.org/10.1007/bf00420561.

Wagner-Riddle, C., Baggs, E.M., Clough, T.J., Fuchs, K., Petersen, S.O., 2020. Mitigation of nitrous oxide emissions in the context of nitrogen loss reduction from agroecosystems: managing hot spots and hot moments. Curr. Opin. Environ. Sustain. 47, 46–53. https://doi.org/10.1016/j.cosust.2020.08.002.

Wilson, D., Farrell, C.A., Fallon, D., Moser, G., Muller, C., Renou-Wilson, F., 2016. Multiyear greenhouse gas balances at a rewetted temperate peatland. Glob. Chang. Biol. 22 (12), 4080–4095. https://doi.org/10.1111/gcb.13325.

Wosten, J.H.M., Ismail, A.B., vanWijk, A.L.M., 1997. Peat subsidence and its practical implications: A case study in Malaysia. Geoderma 78 (1–2), 25–36. https://doi.org/10.1016/s0016-7061(97)00013-x.

Yu, Z.C., Loisel, J., Brosseau, D.P., Beilman, D.W., Hunt, S.J., 2010. Global peatland dynamics since the Last Glacial Maximum. Geophys. Res. Lett. 37. https://doi.org/10.1029/2010gl043584.