

1 **A full year of continuous net soil and ditch CO<sub>2</sub>, CH<sub>4</sub>, N<sub>2</sub>O  
2 fluxes, soil hydrology and meteorology for a drained fen in  
3 Denmark**

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11 **Abstract.** We present a detailed dataset (<https://doi.org/10.60612/DATADK/BZQ8JE>) of automated greenhouse  
12 gas (GHG) net soil and ditch fluxes of carbon dioxide (CO<sub>2</sub>), methane (CH<sub>4</sub>), and nitrous oxide (N<sub>2</sub>O) from a  
13 drained fen in Denmark covering a full year. The dataset resolves small scale spatial and hourly-daily-seasonal  
14 dynamics of GHG soil fluxes. The GHG flux dataset is accompanied by simultaneous time series of soil  
15 temperature and moisture, as well as groundwater table depth and covers spatiotemporal gradients in soil  
16 hydrological and climatic variability. The GHG fluxes of CO<sub>2</sub>, CH<sub>4</sub> and N<sub>2</sub>O were measured simultaneously by  
17 a high-precision cavity ring down laser spectrometer connected with a novel automated GHG system platform  
18 called SkyLine2D (Earthbound Scientific Ltd., UK) that allowed up to 27 individual chamber measurement  
19 points along a 24 meter transect. In total 47.483 chamber measurements were completed and after quality  
20 control 44.631 CO<sub>2</sub> fluxes, 44.099 N<sub>2</sub>O and 42.515 CH<sub>4</sub> fluxes remained.

21 The average ( $\pm$ SE) net soil CO<sub>2</sub> efflux observed at the site ( $2.6 \pm 0.02 \text{ } \mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$  or  $35 \pm 0.3 \text{ tCO}_2 \text{ ha}^{-1} \text{ y}^{-1}$ )  
22 aligns with findings from similar drained fens in northern Europe covering substantial spatial variability. The  
23 organic soil at the site was a larger net source of N<sub>2</sub>O ( $8.9 \pm 0.1 \text{ nmol N}_2\text{O m}^{-2} \text{ s}^{-1}$  or  $123 \pm 1.4 \text{ kg N}_2\text{O m}^{-2} \text{ ha}^{-1} \text{ y}^{-1}$ )  
24 to the atmosphere compared to other temperate drained organic grassland soils in northern Europe with similar  
25 spatial variability as soil CO<sub>2</sub> effluxes. However, the temporal variability of N<sub>2</sub>O fluxes were closely linked to  
26 fluctuations of the groundwater table depth with emission bursts of soil N<sub>2</sub>O emissions during low water table  
27 depth. N<sub>2</sub>O fluxes decreased to near-zero fluxes when the water table depth increased. Net soil CH<sub>4</sub> fluxes were  
28 near-zero and the site overall acted as a smaller net source ( $0.18 \pm 0.06 \text{ nmol CH}_4 \text{ m}^{-2} \text{ s}^{-1}$  or  $0.91 \pm 0.3 \text{ kg CH}_4 \text{ ha}^{-1} \text{ y}^{-1}$ )  
29 compared to other drained organic grassland soils, although net uptake of atmospheric CH<sub>4</sub> was observed as  
30 well especially in drier conditions. Compared to the peat soil, the ditch was a smaller net source of CO<sub>2</sub>  
31 ( $0.94 \pm 0.05 \text{ } \mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$  or  $1.3 \pm 0.7 \text{ tCO}_2 \text{ ha}^{-1} \text{ y}^{-1}$ ) and N<sub>2</sub>O ( $0.35 \pm 0.03 \text{ nmol N}_2\text{O m}^{-2} \text{ s}^{-1}$  or  $4.9 \pm 0.4 \text{ kg N}_2\text{O ha}^{-1} \text{ y}^{-1}$ ). The ditch emission of CH<sub>4</sub> ( $161 \pm 13 \text{ nmol CH}_4 \text{ m}^{-2} \text{ s}^{-1}$  or  $812 \pm 66 \text{ kg CH}_4 \text{ ha}^{-1} \text{ y}^{-1}$ ) average of diffusive  
32 and ebullition fluxes) to the atmosphere was more than two orders of magnitude larger than net the soil CH<sub>4</sub>  
33 emissions.

35 The very large number of fluxes of CO<sub>2</sub>, N<sub>2</sub>O and CH<sub>4</sub> for peat soils and a ditch linked to both groundwater  
36 table data, soil moisture/temperature as well as groundwater and soil physicochemical parameters are unique to  
37 northern temperate peatlands and holds a potential for exploring and testing basic hypothesis on the  
38 simultaneous regulation of these gas fluxes by both soil hydrology and temperature, including soil and

39 groundwater chemistry. The high temporal detail also allows for time series analyses as well as investigations  
40 into diurnal and seasonal patterns of fluxes in response to physical drivers. Similarly, the high frequency of  
41 measured variables and the large number of spatial replicates are furthermore well suited for testing  
42 biogeochemical models as it is possible to have both calibration and validation dataset covering the same period.  
43 Furthermore, the surprisingly large spatial variability of flux data is ideal to include in model sensitivity tests  
44 which can aid in constraining model outputs and develop model routines.

45

46 **1 Introduction**

47 Understanding the climate feedback of temperate drained and rewetted wetlands requires robust observational  
48 datasets of net fluxes, e.g. whether the rewetted peatlands act as net sources to the atmosphere or sinks of  
49 greenhouse gases (GHG). This necessitates being able to capture spatial and temporal variability from these  
50 systems. Flux data covering all three major GHGs are rare for temperate peatlands, and despite growing efforts  
51 to quantify GHG fluxes from drained peatlands, existing datasets often suffer from limited temporal resolution,  
52 short monitoring periods, or a lack of concurrent hydrological and meteorological data. Many studies rely on  
53 manual chamber-based campaigns that may be able to capture overall seasonal dynamics, but fail to capture  
54 short term transient emission phenomenon in response to fluctuations in physical drivers, for example  
55 fluctuating groundwater. Also, manual based measurements are labour intensive limiting the number of spatial  
56 replicates. Moreover, current high temporal resolution datasets for wetlands using eddy covariance typically  
57 offer high good quality for a specific wetland site, but it is challenging to derive the specific spatial variability  
58 across the different sub-environments within the wetland, for example between hummocks and hollows with  
59 different GHG emission profiles. This discrepancy between spatial and temporal coverage of current flux  
60 methodologies in wetlands in turn hampers the ability to develop precise models that integrate spatiotemporal  
61 patterns and can forecast GHG fluxes at the ecosystem scale more precisely. This can impact the ability to  
62 predict climatic feedback of wetlands now and under future alteration of these systems driven by land use and  
63 climatic changes.

64 However, automated GHG closed chamber flux measurements from ecosystems are becoming increasingly  
65 common, also in peatland research (Anthony and Silver, 2023; Boonman et al., 2024) as equipment costs  
66 decrease and awareness grows about the importance of resolving temporal variability of GHG fluxes to better  
67 understand soil biogeochemical processes and soil-climate feedback. But high-frequency data of GHG fluxes  
68 are still scarce for peatlands and spatial variability of fluxes is rarely represented as well due to limited number  
69 of spatial replicates. Thus, most automated chamber systems are setup around a multiplexer control unit linking  
70 multiple chambers with one or more GHG analysers. State-of-the-art automatic chamber systems, like the LI-  
71 8250 Automated Gas Flux System (LiCOR, USA) or the eosAC-LT/LO (Eosense Inc. Canada), i.e. allow for a  
72 standard number of 8 or 16 chambers, respectively, that can be upgraded to 36 chambers with additional  
73 manifolds. Such large replicate chambers allow for improved characterization of spatial variation or treatment  
74 effects coupled with temporal variations but are costly to establish.

75 We here present a dataset that addresses the abovementioned limitations by combining high-frequency,  
76 continuous measurements of net soil fluxes of carbon dioxide (CO<sub>2</sub>), methane (CH<sub>4</sub>) and nitrous oxide (N<sub>2</sub>O)  
77 with detailed hydrological and meteorological variables. The GHG fluxes were measured with an automated  
78 GHG, called SkyLine2D, chamber system over 12 months resolving spatiotemporal patterns of GHG fluxes  
79 including 27 individual collars (26 on organic soil and 1 in a ditch) over a 24 m transect on a temperate drained  
80 fen peatland. Integrated quality control, flagging of erroneous or uncertain flux measurements enabled objective  
81 filtering of poor-quality data on the entire dataset. This comprehensive spatiotemporal coverage enables robust  
82 calibration and validation of biogeochemical and hydrological models, particularly those aiming to simulate the  
83 complex interactions between water table dynamics, soil processes, and GHG emissions in managed peatland  
84 systems.

85 Considering the critical need for obtaining high-quality data on soil GHG fluxes from natural and restored  
86 peatlands in Europe and globally, our dataset marks an important contribution to this endeavour as it addresses  
87 current data shortcomings for Danish and European peatlands by providing detailed data on temporal and spatial  
88 patterns of GHG fluxes from organic soils and drainage ditches together with environmental drivers of soil  
89 hydrology and temperature, organic soil properties and groundwater geochemistry. We publish this data with the  
90 aim of it being used by the scientific community for both experimentalists to test hypothesis of how GHG  
91 dynamics are related to hydrology, soil, geochemistry and climate, as well as for the modelers to test and  
92 develop biogeochemical models for peat lands.

## 93 **2 Materials and Methods**

### 94 **2.1 Site description**

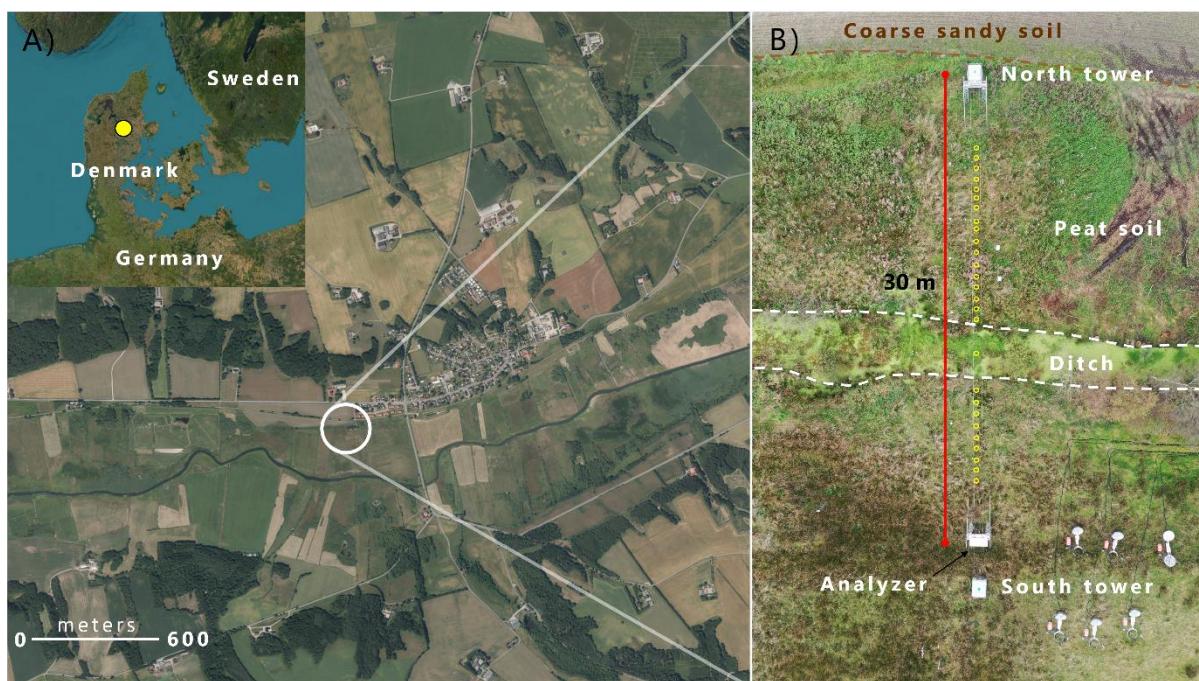
95 The field site, Vejrumbro (N 56.43819 E 9.54527 (WGS 84)), is located in Central Jutland, in Denmark near the  
96 city of Viborg (Fig. 1A) with a mean annual temperature of 8.3°C and annual precipitation of 675 mm for the  
97 period 1991–2020 (measured 6 km away at Aarhus University Viborg Meteorological Station in Foulum  
98 (Jørgensen et al., 2023)). It is situated in the Nørre Å valley and is characterized as a riparian fen peat soil (Reza  
99 Mashhadi et al., 2024). The riparian fen developed in a former glacial river valley with flat topography gently  
100 sloping (<2.5 meters over 300 meters) towards the Nørre Å that forms the central river in this area (Fig. S1).  
101 The site was drained in 1950 with ditches and tile drains for cultivation and was used to cut hay for fodder as the  
102 conditions were unfavourable for cereal production (Nielsen et al., 2024). Since 2018, Vejrumbro has been a  
103 living lab for agroecological research managed by the Department of Agroecology at Aarhus University. From  
104 2018, the site had a passive rewetting strategy by terminating maintenance of the open ditches. During 2022, the  
105 main ditches were gradually blocked.

#### 106 2.1.1 Site preparation

107 We chose to perform the flux measurements without aboveground plants as the small chamber dimensions  
108 (height of 20 cm) prohibited inclusion of these in the chamber as the plants typically reach over 100 cm in  
109 height at this site. The strategy was therefore to focus on measuring net soil GHG fluxes, where we assume the  
110 contribution of gases are derived from heterotrophic respiration of older peat C/N, root exudated C/N from  
111 adjacent plants, dissolved N in groundwater and belowground autotrophic respiration (CO<sub>2</sub>) from roots  
112 inhabiting the peat below the collars. We are aware that omitting plants prohibit a full evaluation of the net  
113 ecosystem exchange of GHG and hence its net climate impact, as the aboveground plants represent a net sink of  
114 atmospheric CO<sub>2</sub> and can increase the emission of CH<sub>4</sub> and N<sub>2</sub>O (Jørgensen et al., 2012; Vroom et al., 2022).  
115 However, by removing plants we isolate the soil processes leading to net soil emission/uptake of the GHG.  
116 Collectively, this can provide a mechanistic insight into the regulation of fluxes by hydrology and temperature.  
117 We acknowledge that studies of GHG fluxes in peatlands should seek to include the aboveground plant  
118 component to the net GHG flux from the ecosystem if possible.

119 Two months prior to collar installation in summer 2021, we cleared vegetation within and around each collar  
120 (~40 × 40 cm) by harvesting and applying a single recommended dose of glyphosate (~100 mg m<sup>-2</sup>) to  
121 aboveground plants only, avoiding soil contact. Glyphosate's average half-life in mineral soils is ~21 days,

122 ranging from 6–87 days and increasing with clay content (Padilla and Selim, 2020). Given the low dose and  
 123 absence of clay, residual glyphosate was likely minimal during flux measurements. Although repeated  
 124 applications can suppress microbial activity (Nguyen et al., 2016), the single treatment months prior suggests  
 125 limited direct impact on microbial respiration. Still, transient effects cannot be ruled out, and the lack of an  
 126 untreated control prevents quantification. Regrowth inside collars was manually removed at least weekly,  
 127 minimizing photosynthetic CO<sub>2</sub> uptake. While regrowth abundance was not measured, stable net CO<sub>2</sub> efflux  
 128 between removals suggests minimal impact. Aboveground plant removal is standard for isolating soil GHG  
 129 fluxes, though belowground autotrophic respiration from adjacent roots remained, as trenching was avoided to  
 130 reduce site disturbance. Without a control plot, the direct effect of disturbance on GHG fluxes remains  
 131 uncertain.



132  
 133 **Figure 1: A)** The Vejrumbro location in Jutland (N 56.43819 E 9.54527 (WGS 84)) in the Nørre Å valley near the  
 134 village of Vejrumbro. The grey circle marks the placement of the SkyLine2D system. Satellite images: © Google  
 135 Earth. **B)** Drone image of the measurement transect (September 27<sup>th</sup>, 2023) after flux measurements had stopped.  
 136 Dashed brown line marks the approximate boundary between the agricultural field, coarse sandy soil (north) and the  
 137 peat/organic soil (south). The red line marks the end points of the SkyLine2D system (30 meters). The open yellow  
 138 circles (n=27) mark the approximate position of individual collars across the transect of the field (24 meters in length)  
 139 where greenhouse gas fluxes were measured. The ditch is located between the dashed white lines. The analyser was  
 140 placed at the south tower. Elevation above sea level along the 24-meter collar transect varied from 3.77 m in the south  
 141 to 4.06 m in the north.

142 **2.2 Overview of time series of GHG fluxes, soil temperature/moisture, air temperature, wind direction  
 143 and groundwater level**

144 The dataset is comprised of a 12-month time series of net soil fluxes of CO<sub>2</sub>, CH<sub>4</sub> and N<sub>2</sub>O, accompanied by a  
 145 longer timeseries of soil temperature and moisture at 5 cm depth, meteorological variables (air temperature,  
 146 wind speed and direction measured at 2 meter height) and a shorter time series groundwater table level, depth  
 147 and temperature (Table 1). Due to equipment failure of the SkyLine2D the GHG flux measurements started on

148 February 2<sup>nd</sup>, 2022 and ended January 28<sup>th</sup>, 2023 in total 360 days (Table 1). Groundwater level measurements  
149 started between March 9<sup>th</sup> to 31<sup>st</sup>, 2022 (Table 1). All other variables were measured continuously from July 1<sup>st</sup>,  
150 2021, until January 31<sup>st</sup>, 2023 (Table 1). In the period between December 7<sup>th</sup> and 19<sup>th</sup>, 2022 intermittent periods  
151 of snow cover (depth was not measured) on the ground occurred. This snow cover did not impede flux  
152 measurements.

Table 1: Available time series data from the Vejrumbro SkyLine2D system. Coloured Data availability for each variable in 2021 to 2023 is indicated with horizontal bars.

Variable	Unit	Model/sensor type	Frequency (minutes)			2021			2022			2023							
			Aug	Sep	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
CO <sub>2</sub> flux*	μmol CO <sub>2</sub> m <sup>-2</sup> s <sup>-1</sup>	G2508 (Picarro Inc., USA)	~10**	----	----	----	----	----	----	----	----	----	----	----	----	----	----	----	----
CH <sub>4</sub> flux*	nmol CH <sub>4</sub> m <sup>-2</sup> s <sup>-1</sup>	G2508 (Picarro Inc., USA)	~10**	----	----	----	----	----	----	----	----	----	----	----	----	----	----	----	----
N <sub>2</sub> O flux*	nmol N <sub>2</sub> O m <sup>-2</sup> s <sup>-1</sup>	G2508 (Picarro Inc., USA)	~10**	----	----	----	----	----	----	----	----	----	----	----	----	----	----	----	----
Soil temperature at 5 cm depth***	°C	RXXW-TMB-868 (Onset, USA)	5	----	----	----	----	----	----	----	----	----	----	----	----	----	----	----	----
Soil water content at 5 cm depth***	(cm <sup>3</sup> cm <sup>-3</sup> )	RXXW-SMD-868 (HS) (Onset, USA)	5	----	----	----	----	----	----	----	----	----	----	----	----	----	----	----	----
Air temperature at 2 m height	°C	S-THC-M002 (Onset, USA)	5	----	----	----	----	----	----	----	----	----	----	----	----	----	----	----	----
Wind speed	m s <sup>-1</sup>	S-WSB-M003 (Onset, USA)	5	----	----	----	----	----	----	----	----	----	----	----	----	----	----	----	----
Wind direction	°	S-WDA-M003 (Onset, USA)	5	----	----	----	----	----	----	----	----	----	----	----	----	----	----	----	----
Groundwater level****	m.a.s.l.	DCL532 (BD sensors, Germany)	15	----	----	----	----	----	----	----	----	----	----	----	----	----	----	----	----
Groundwater table depth****	cm	DCL532 (BD sensors, Germany)	15	----	----	----	----	----	----	----	----	----	----	----	----	----	----	----	----
Groundwater temperature****	°C	Dallas DS 18B20	15	----	----	----	----	----	----	----	----	----	----	----	----	----	----	----	----

\*Net soil/ditch fluxes for all collars 1 - 27.

\*\*Time in between two consecutive flux measurements. The 10 minutes comprise actual flux measurement of 5 minutes and 5 minutes headspace flushing between flux measurements.

\*\*\*Measured for a subset of collars: 4, 7, 9, 23, 27.

\*\*\*\*Measured for a subset of collars: 1, 5, 10 (ditch), 13, 18, 22, 27.

158 **2.3 The SkyLine2D system at Vejrumbro**

159 The SkyLine2D system is an automated chamber based system for measuring GHG fluxes. The system is  
160 designed and built by Earthbound Scientific Ltd. (United Kingdom). We used the SkyLine2D system to measure  
161 the net soil fluxes of CO<sub>2</sub>, CH<sub>4</sub> and N<sub>2</sub>O measured with an automated GHG chamber system over 12 months  
162 resolving spatiotemporal patterns of GHG fluxes including 27 individual collars (26 on organic soil and 1 in a  
163 ditch) over a 24 m transect on a temperate drained fen peatland (Fig. 1B and 3).

164 The SkyLine2D system transect was oriented in a north-south direction (Fig. 1B). Two 2.5-meter-tall scaffold  
165 towers marked the end of the 30 m SkyLine2D system (Fig. 1B and Fig. S2D). The towers were fixed by ropes  
166 attached to 1000L pallet tanks filled with water (Fig. S2D) that maintained a stable position of the towers and  
167 ropes and hence placement of the chamber over the collars. The GHG analyser (model G2508, Picarro Inc.,  
168 USA) was installed in a waterproof and temperature-controlled shelter at the south end of the transect (Fig. 1B  
169 and Fig. S2C). The transect was situated on the edge of the riparian fen near the mineral upland soils, where  
170 active agriculture was practiced (Fig. 1B). Along the transect volumetric soil water content (SWC) and soil  
171 temperature (ST) as well as water table depth (WTD) were measured at seven locations (Fig. 2). The agricultural  
172 field north of the SkyLine2D was sown with annual crops in rotation according to common practice.

173 **2.3.1 Greenhouse gas flux measurements with the SkyLine2D system**

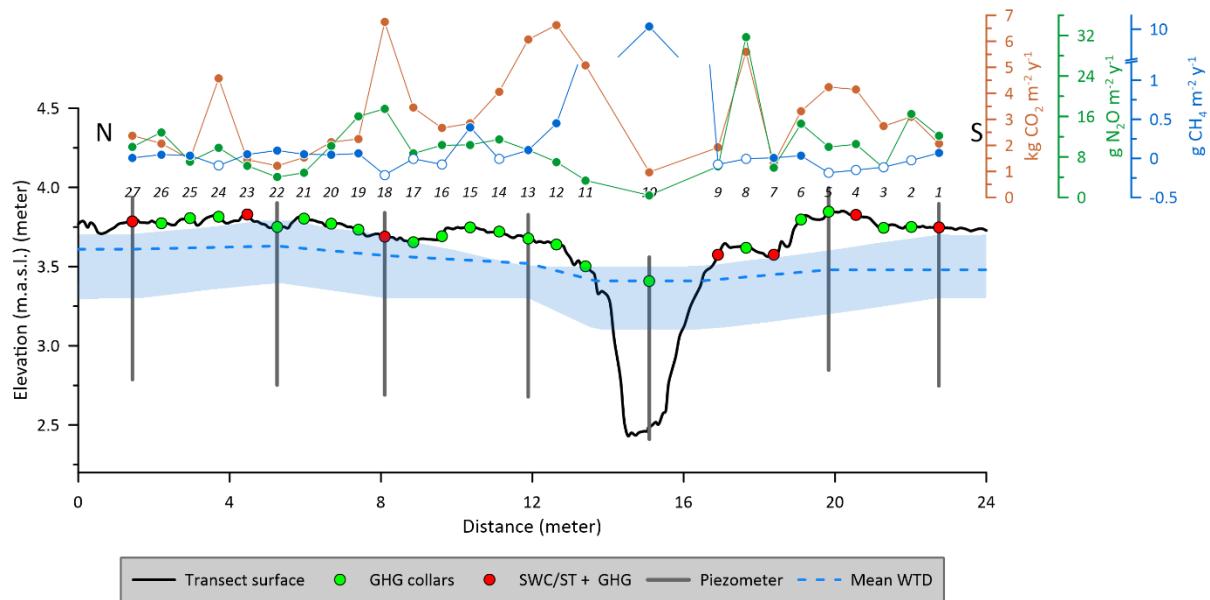
174 Along the SkyLine2D transect the 26 individual collars (Ø19 cm) along the 24 meters transect on organic soil  
175 (Fig. 2) were inserted 5 cm into the peat leaving 5 cm above the surface. The collars were distanced app. 70 cm  
176 apart. One collar was installed in the ditch by inserting a tube (Ø19 cm, length 100 cm) to the bottom of the  
177 ditch with holes deeper than the minimum water level in the ditch to allow water flow. Thus, it was avoided that  
178 air entered in the collar in the ditch due to low water levels in the ditch. On top of this longer tube a collar (Ø19  
179 cm, length 10 cm) was glued allowing for flux measurements. The chamber was programmed to stop when the  
180 bottom of the chamber sat the water surface if the water level in the ditch extended above the top of the collar.  
181 For most of the time the collar was not submerged, and the chamber therefore hit the collar.

182 There was one round transparent chamber (height: 39.5 cm and inner Ø: 19 cm, volume: 11.2 L) on the  
183 SkyLine2D, hanging below a moving trolley, which was suspended on two ropes stretched between the north  
184 and south towers (Fig. S2A and B). At defined positions along the rope, neodymium magnets had been inserted,  
185 and a magnet sensor (Fig. S2B) on the trolley informed the internal computer to stop and lower the chamber  
186 over positions with a collar on the surface. The chamber was lowered and guided down to the collar by  
187 supporting rods shaping a funnel (Fig. S2A). The chamber stopped when it hit the collar, achieved through a  
188 pressure sensor on top of the chamber connected to a hollow rubber gasket (Ø 3 cm) at the bottom, which also  
189 sealed the chamber with the collar. There was no fan installed in the chamber as the mixing was ensured by the  
190 main pump (Fig. S2C). A vent was installed in the top of the chamber to allow for pressure equilibration under  
191 windy conditions and chamber deployment.

192 One entire flux + flushing sequence lasted 10 minutes (Table 1). The chamber closure period was set to 5  
193 minutes with a purging time of 5 minutes in between measurements when chamber was open and hanging  
194 underneath the trolley at approximately 1 meter above the ground (Fig. S2D). This provided on average 10 min

195 between flux measurements on consecutive collars (Table 1). Due to small variations in mechanical operations,  
 196 flux measurements were occasionally farther apart than 10 minutes, but overall, the timing of the SkyLine2D  
 197 system was consistent. After each cycle of 27 flux measurements there was a 30-minute delay until the start of  
 198 the next cycle. On average this resulted in 4-5 flux measurements per collar per day throughout the period.

199 To determine the concentrations of CO<sub>2</sub>, CH<sub>4</sub> and N<sub>2</sub>O in the chamber air, a laser spectroscopy GHG analyser  
 200 (model G2508, Picarro Inc., USA) was used. The sample output frequency was set to 1 Hz with a manufactured  
 201 specified raw precision on 1 Hz data for CO<sub>2</sub>: 240 ppb, CH<sub>4</sub>: 0.3 ppb and N<sub>2</sub>O: 5 ppb at ambient conditions  
 202 (Picarro Inc., USA). A main pump (model: N86 KN.18, KNF, Germany) circulated the air to and from the  
 203 chamber at 6 L min<sup>-1</sup>. The GHG analyser was installed in parallel to the inflow from the chamber due to the  
 204 much lower flow of 250 mL min<sup>-1</sup> of the vacuum pump. There was a 30-meter tube between the chamber and  
 205 main pump to allow for the GHG analyser to remain stationary in the hut while the trolley moved.



206 **Figure 2: Schematic representation of the measurement transect at Vejrumbro and associated measurement**  
 207 **variables. The annual cumulative fluxes of CO<sub>2</sub> (red) (kg CO<sub>2</sub> m<sup>-2</sup> y<sup>-1</sup>), N<sub>2</sub>O (green) (g N<sub>2</sub>O m<sup>-2</sup> y<sup>-1</sup>) and CH<sub>4</sub> (blue) (g**  
 208 **CH<sub>4</sub> m<sup>-2</sup> y<sup>-1</sup>) are shown for each collar across the measurement transect at Vejrumbro. Closed and open symbols for**  
 209 **CH<sub>4</sub> represent net cumulative emission and uptake, respectively. Mean groundwater level is the mean water table**  
 210 **elevation measured in piezometers (blue dashed line) with shaded blue area represent maximum and minimum**  
 211 **observed groundwater elevation. GHG collars (green symbols) mark the positions of greenhouse gas flux**  
 212 **measurements of CO<sub>2</sub>, CH<sub>4</sub> and N<sub>2</sub>O. SWC/ST + GHG mark the positions where volumetric soil water content**  
 213 **(SWC) and soil temperature (ST) at 5 cm depth were measured alongside greenhouse gas fluxes. Numbers on top of**  
 214 **plot show the collar numbers (from 1 – 27). N and S mark the north and south ends of the transect (see Fig. 1B). The**  
 215 **peat depth was at least one meter in all points. Elevation is given meters above sea level (m.a.s.l.).**

217 **2.4 Peat sampling and analysis**

218 In November 2023 the peat across the SkyLine2D transect was sampled to 1 meter depth using a Russian auger  
219 and cores split into five layers of 20 cm thickness. Collars 1, 2, 5, 6, 8, 13 – 27 were sampled (Fig. 2). For the  
220 remaining collars it was not possible to retrieve a sample due to excessive wetness of the peat. The  
221 decomposition of the peat samples were assessed by a 10-point Von Post scale of humification (1 = completely  
222 undecomposed and 10 = completely decomposed) together with  $\text{pH}_{\text{H}_2\text{O}}$  determined by suspending peat in  
223 demineralized water (1:5 peat:water mix), dry bulk density ( $\text{g cm}^{-3}$ ) and total C and N by dry combustion ( $\text{g C/N}$   
224  $100 \text{ g peat}^{-1}$  or %).

225 **2.5 Groundwater table level, depth and sampling**

226 To measure the groundwater level piezometers (inner diameter 5 cm) were installed at collars 1, 5, 10 (ditch),  
227 13, 18, 22, 27 (Fig. 2) to 1 meter depth below the surface, which is deeper than the lowest groundwater level in  
228 summer (~60 cm below the surface) with openings from 0.1 – 1.2 meter below terrain. In the ditch the  
229 piezometer bottom was deeper than one meter to secure anchoring in the peat. The piezometers were installed  
230 approximately 50-60 cm beside the collars to avoid interference with the SkyLine2D system. After installation,  
231 piezometers were cleaned and sealed at the surface with bentonite pellets to avoid surface infiltration along the  
232 piezometers which can distort water level measurements.

233 Pressure transducers (Table 1) connected to Arduino-loggers were installed in each piezometer (at collars 1, 5,  
234 10, 13, 18, 22 and 27 – Fig. 2) approximately 1 m below terrain measuring water levels every 15 minutes. The  
235 pressure transducers were vented and thus do not need correction for atmospheric pressure.

236 The groundwater levels were described using two metrics: hydraulic head and groundwater table depth (WTD).  
237 Hydraulic head represents the water level relative to mean sea level, based on the Danish Vertical Reference  
238 (DVR90), while WTD indicates the depth of the groundwater below the surface terrain and represented in  
239 positive values, where WTD of zero is equivalent to groundwater level at the terrain surface. The elevation of  
240 top of the piezometers were measured using a GPS (model GS07 High Precision GNSS Antenna with a CS20  
241 Controller, Leica, Germany) and used as a local reference for hydraulic head. Manual measurements of  
242 groundwater levels were conducted every 2 months and used to calibrate the logger water levels to hydraulic  
243 head and WTD.

244 **2.5.1 Groundwater water sampling and chemical analysis**

245 Groundwater was sampled monthly in the piezometers placed at collars 1, 5, 13, 18, 22 and 27 (Fig. 2) by  
246 retrieving a 200 mL sample 20-30 cm below the groundwater level at the sampling time. The water sample was  
247 retrieved using a syringe and transferred to a plastic bottle that was capped immediately to avoid exchange with  
248 the atmosphere and contamination. Water samples were frozen immediately after sampling and subsequently  
249 after thawing analyzed for pH, EC and alkalinity on an 855 Robotic Titrosampler (Metrohm, Germany). Total N  
250 and DOC were measured on a TOC-V CPH Analyzer with Total Nitrogen Unit TNM-1 & ASI-V Autosampler  
251 (Shimadzu, Japan). Ion chromatograph (IC) analyses of  $\text{Cl}^-$ ,  $\text{NO}_3^-$ , and  $\text{SO}_4^{2-}$  were performed on a 930Compact  
252 IC Flex (Metrohm, Germany) and  $\text{NH}_4^+$  concentrations were measured with continuous flow analysis using a  
253 Seal AA500 Autoanalyzer (SEAL Analytic, USA). Total dissolved Fe and P were analyzed with coupled

254 plasma–mass spectrometry (ICP-MS) on an iCAP-Q ICP-MS (Thermo Fisher Scientific, USA) in KED mode  
255 using He as the collision gas. Prior to analysis the 10 mL subsamples were acidified with 200  $\mu$ L concentrated  
256 nitric acid. Elemental ICP-MS analyses also included dissolved base cations of  $\text{Ca}^{2+}$ ,  $\text{Mg}^{2+}$ ,  $\text{K}^+$ ,  $\text{Na}^+$  as well as  
257 total dissolved Al and Mn cations (not shown but included in the data set).

258 **2.6 Soil moisture and temperature measurements**

259 Soil moisture and temperature probes were initially inserted for collars 1, 4, 7, 9, 18, 23, 27 (Fig. 2) in order to  
260 obtain a representation of the entire transect. Soil moisture probes (6 cm length) were inserted at an approximate  
261 30° angle 5 cm outside the collar, while the soil temperature probes were inserted vertically adjacent to the soil  
262 moisture probe. Due to sensor failures soil moisture was measured for collars 1, 7, 9, 18, 23 and 27 and soil  
263 temperature at 4, 7, 9, 23 and 27.

264 **2.7 Wireless data transfer**

265 Wireless sensors for air temperature, wind speed, wind direction, soil temperature and volumetric soil water  
266 content were set up with Wi-Fi data transfer to HOBO RX3000 Weather Station (Onset, USA) equipped with  
267 HOBOnet Manager (RXMOD-RXW-868) module for wireless communication with sensors and logged data  
268 every 5 minutes. Data access was through the HOBOlink cloud software.

269 Groundwater loggers were interfaced with the I<sup>2</sup>C (Inter-integrated Circuit) protocol and data was collected on  
270 Arduino custom-built logger (<https://vandstande.dk/logger.php>) with wireless connection via LoRaWANor  
271 SigFox.

272 **2.8 Calculation of diffusive fluxes**

273 Fluxes were calculated and quality checked using the goFlux R package (Rheault et al., 2024) and presented as  
274  $\mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$ ,  $\text{nmol N}_2\text{O m}^{-2} \text{ s}^{-1}$  and  $\text{nmol CH}_4 \text{ m}^{-2} \text{ s}^{-1}$ . Prior to flux calculations, the gas concentration data  
275 from the G2508 analyzer was matched to the chamber closure time and chamber id to determine the start time of  
276 the chamber measurement, so it was possible to separate individual flux measurements from each collar over the  
277 measurement time (see examples of flux detection and calculation in Fig. S3A-D). An automatic deadband  
278 detection method was applied based on maximal  $R^2$  of a linear regression over the first 180 s (in 10 s steps) after  
279 chamber closure. The deadband was allowed to attain values between 0 to 150 seconds thereby also allowing for  
280 compensation for the ~60 s delay between chamber headspace gas concentration change and GHG analyser  
281 detection due to transport time through the 30 m tube connecting the chamber and GHG analyser.

282 Flux calculations were done with both linear (LM) and non-linear (Hutchinson-Mosier – HM) regression models  
283 (Pihlatie et al., 2013) to determine the slope at time zero. The best flux estimates with either the LM or HM  
284 regression model was determined using the *best.flux* function in the goFlux package (Rheault et al., 2024).  
285 Shortly, if the RMSE of the HM model was lower than minimum detectable flux (MDF), HM was chosen.  
286 However, if the ratio (g-factor) between HM and LM was larger than 2, LM was chosen, as this indicates over-  
287 fitting of the HM, which may result in unrealistic large HM flux estimates. If the relative SE of the slope  
288 (SE/slope) at time zero for the HM model was larger than 100% it indicated overfitting of the HM model and  
289 the LM was chosen. This approach is conservative as it will discard non-linear flux behaviour and instead  
290 provide a conservative linear flux estimate. Out of 47.438 detected flux measurements for  $\text{CO}_2$ ,  $\text{CH}_4$  and  $\text{N}_2\text{O}$ ,

291 respectively, a total of 2807 CO<sub>2</sub> fluxes (5.9%), 3339 N<sub>2</sub>O fluxes (7%) and 4923 CH<sub>4</sub> fluxes (10.3%) were  
292 discarded due to the following two situations: 1) chamber mechanical malfunction either resulting in imperfect  
293 sealing on collar due to erroneous lowering of chamber on collar indicated by background atmospheric or  
294 fluctuating gas concentrations in the headspace and 2) at *in situ* flux levels close to the minimum detectable flux  
295 of the Picarro G2508 analyser (Christiansen et al., 2015) non-significant regression (between concentration and  
296 time and GHG concentration) ( $p>0.05$ ) were also discarded as it was not possible to statistically distinguish  
297 whether there was a real flux or the lack of significant regression was because of chamber malfunction. It is  
298 acknowledged that discarding low fluxes can bias annual means and cumulative values, but the data quality did  
299 not allow us to determine whether the flux measurement was performed correctly and hence a conservative  
300 approach was chosen as including false low fluxes would also bias the data set.

301 For flux measurements the air temperature in 2 meters was used as an estimate of the chamber headspace  
302 temperature along with a 1 atm air pressure.

303 The annual cumulated fluxes from the soil or the ditch (diffusive only) were estimated simply by multiplying the  
304 daily average CO<sub>2</sub>, CH<sub>4</sub> or N<sub>2</sub>O flux for the measurement period with 365 days. We believe for the purpose of  
305 data presentation that this simplistic methodology is adequate here, also given the very few data gaps in the  
306 timeseries. However, there are other more sophisticated methods using interpolation and response variable  
307 functions that may refine the annual budget. However, it is not the goal of this manuscript to present these  
308 methodologies but to provide the data so other users can test different temporal upscaling methodologies.

### 309 **2.9 Calculation of ebullition fluxes in the ditch**

310 Methane ebullition fluxes were occasionally observed only in the ditch. The resultant CH<sub>4</sub> time series for the  
311 chamber would have a characteristic appearance (Fig. S4) where the measurement would essentially start out as  
312 diffusive flux measurement, then CH<sub>4</sub> bubbles entered the chamber headspace, and the concentration would  
313 quickly increase to a maximum value and reach a threshold concentration corresponding to the mixed headspace  
314 concentration. In these cases, the LM/HM flux calculation assumptions are violated and instead the ebullition  
315 flux would be calculated as the total increase in CH<sub>4</sub> mass m<sup>-2</sup> per 5 min enclosure. The mass flux of CH<sub>4</sub> per  
316 enclosure (nmol m<sup>-2</sup> per 5 min enclosure) was calculated according to Eq. (1):

$$317 F_{CH_4-ebu} = dCH_4 * \frac{V_{system}*P}{A*R*T} \quad (1)$$

318 Where dCH<sub>4</sub> is the concentration difference in nmol between start of chamber enclosure (CH<sub>4,start</sub>) and end CH<sub>4</sub>  
319 concentration (CH<sub>4,end</sub>) after it reached a plateau (Fig. S4), V<sub>system</sub> is the total volume (11.7 L) of the system  
320 (collar, chamber, tubes and GHG analyser) in L, P is the pressure (1 atm), A is the area of the collar (0.028 m<sup>2</sup>),  
321 R is the gas constant (0.082057 L atm K<sup>-1</sup> mol<sup>-1</sup>) and T is the chamber headspace temperature (K). To calculate  
322 the ebullition flux per second the ebullition flux estimate was divided by 12\*60 seconds (300), equivalent to the  
323 number of seconds over the 5 minute measurement period.

324 Out of a total of 1728 flux measurements from the ditch (collar 10), 334 were classified as ebullitions according  
325 to our definition above. indicating that ebullition was erratic which is in line with studies of ebullition of fluxes  
326 from ponds (Sø et al., 2023; Wik et al., 2016). Hence, it can be assumed that ebullition occurred around 19.3%  
327 of the time during the measurement period (360 days). An annual estimate of the ebullition flux was calculated

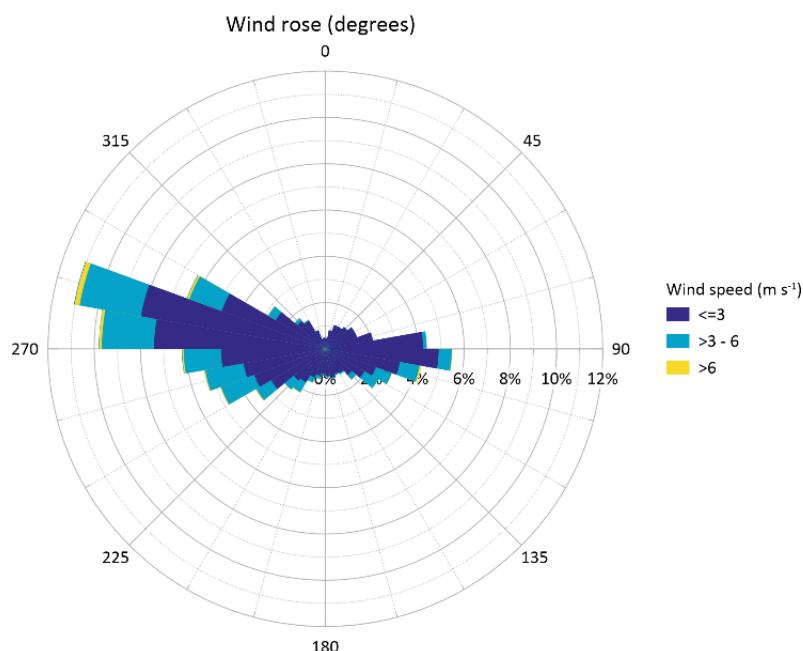
328 as the average ebullition flux in  $\text{nmol CH}_4 \text{ m}^{-2} \text{ s}^{-1}$  by multiplying with number of seconds over 365 days and the  
329 19.3% during period where ebullition occurred.

330 Ebullitions could also be caused by mechanical disturbance of the chamber landing on the collar. Ebullition  
331 fluxes were discarded if the sudden increase in  $\text{CH}_4$  headspace concentration (Fig. S4) occurred 60 seconds after  
332 recorded chamber closure as this indicated bubbles released by chamber deployment on top of the collar.

333 **3 Data presentation**

334 **3.1 Wind speed and direction**

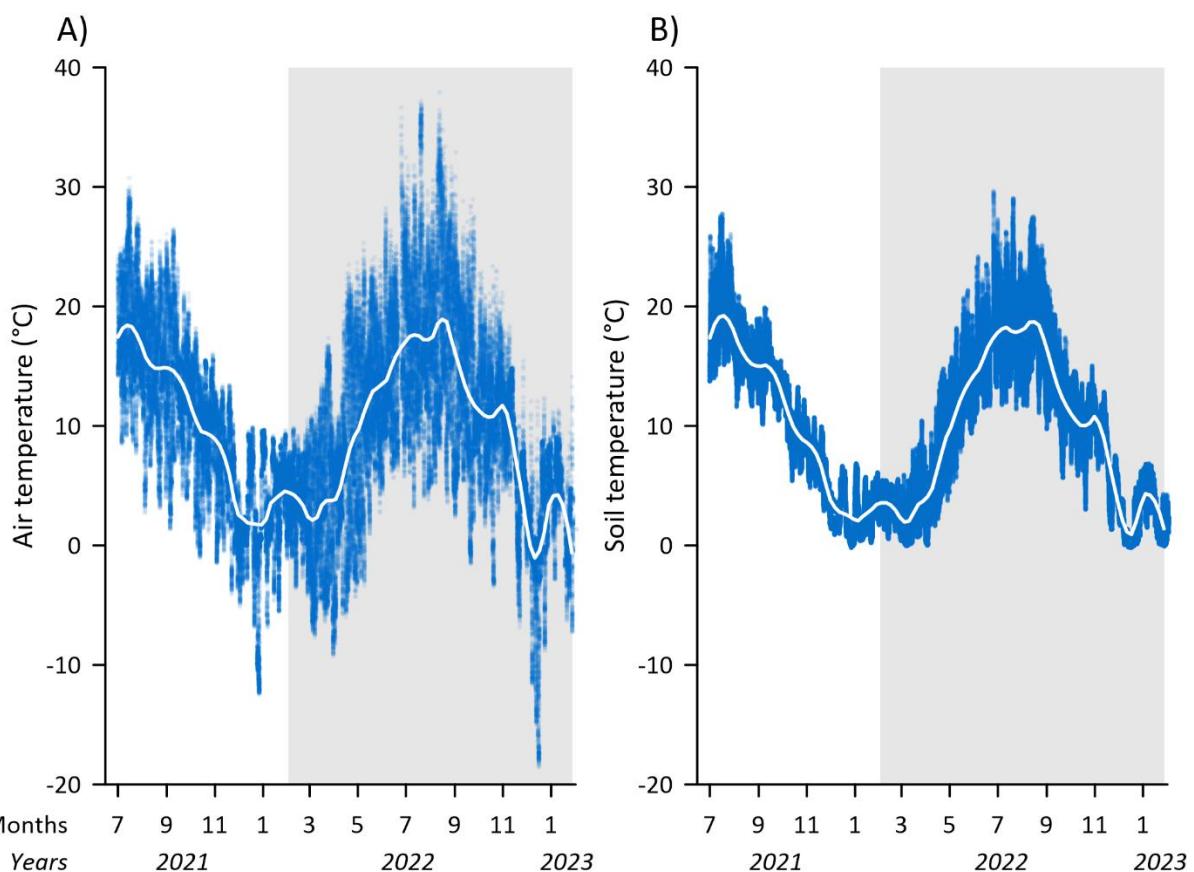
335 Generally, the wind regime during the measurement period (February 2<sup>nd</sup>, 2022 to January 28<sup>th</sup>, 2023) was rather  
336 mild with monthly average wind speeds ranging between 1.2 to 2.9  $\text{m s}^{-1}$  and maximum gust up to 20  $\text{m s}^{-1}$ . The  
337 wind direction was uniformly from the west for 52% of the time, with easterly winds constituting 27% and  
338 northern and southern winds 8 and 13% of the time (Fig. 3). Winds from western directions were highest for the  
339 longest period, while easterly winds were of similar magnitude, but less frequent (Fig. 3). Northern and  
340 southerly winds were generally below 3  $\text{m s}^{-1}$  and represented periods with still conditions. The very uniform  
341 western-eastern wind field at Vejrumbro may also partly be explained by the W-E direction of the valley in  
342 which the site is situated, that effectively blocks or dampens winds from S and N.



343

344 **Figure 3: Wind regime at Vejrumbro for the period July 1<sup>st</sup>, 2021 to January 31<sup>st</sup>, 2023 presented as a wind rose**  
345 **diagram with wind speed and direction for the period.**

346 3.2 Air and soil temperature



347  
348 **Figure 4:** Time series of A) air temperature in °C measured at 2 meter height above the surface and B) soil  
349 temperature (°C) at 5 cm depth for collars 4, 7, 9, 23 and 27 along the measurement transect. The blue dots are the  
350 raw 5 min measurements of air temperature and the white lines represent LOESS fit to show overall seasonal trend.  
351 The periods of GHG measurements with the SkyLine2D system are shown with the shaded area.

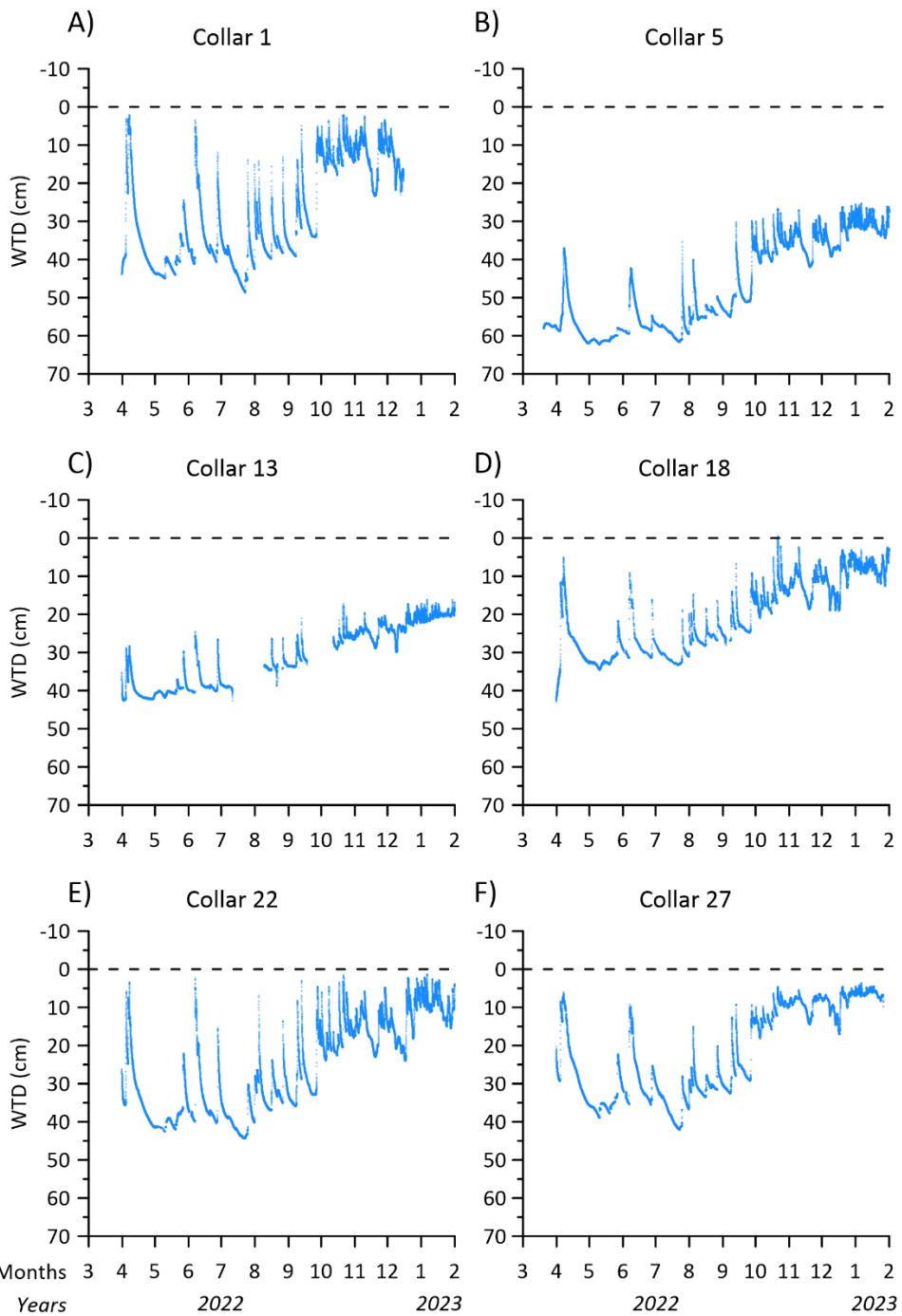
352 Over the study period the average air temperature was 9.6°C ranging between maximum 37.9°C and minimum  
353 of -18.6°C (Fig. 4A). Monthly ranges of air temperatures (Table 2) show >20°C variation between minimum  
354 and maximum, except for February, pointing towards large diurnal variations. Soil temperature magnitude and  
355 temporal variation were similar across the transect, varying between 0 to 28°C (Fig. 4B) and followed that of air  
356 temperature (Fig. 4A) with less variability (Fig. 4B and Table 2). The annual site average soil temperature was  
357 similar to the air temperature (Table 2).

358 **Table 2: Monthly mean, maximum and minimum air temperature and soil temperature (°C), groundwater table**  
 359 **depth (cm) and volumetric soil water content (cm<sup>3</sup> cm<sup>-3</sup>) at Vejrumbro in the measurement period from February 1<sup>st</sup>,**  
 360 **2022 to January 31<sup>st</sup>, 2023.**

Variable	Month	Year												
		2022						2023						
Air temperature (°C)	Mean	3.8	3.0	6.6	12.0	15.4	17.7	16.6	13.4	10.7	6.9	1.2	3.7	9.6
	Max	10.6	17.4	23.7	25.3	36.7	37.2	37.9	32.9	23.3	18.4	12.4	14.1	-
	Min	-4.3	-9.3	-8.3	-3.4	4.3	3.2	2.7	-1.5	-3.5	-6.9	-18.6	-7.3	-
Soil temperature (°C)	Mean	3.0	3.2	2.9	6.4	12.3	16.1	18.4	17.0	13.8	10.3	7.2	2.1	9.6
	Max	6.5	5.3	9.1	12.5	18.8	25.1	27.0	24.7	19.3	14.3	12.6	6.3	-
	Min	0.3	1.1	0.4	0.8	6.6	10.7	12.4	11.8	7.0	4.0	2.1	0.0	-
Groundwater table depth (WTD) (cm)	Mean	-	39	35	41	36	41	35	31	20	18	17	13	29
	Max	-	58	39	58	43	52	46	36	30	31	28	28	-
	Min	-	23	5	24	9	28	22	9	5	6	3	2	-
Volumetric soil water content (cm <sup>3</sup> cm <sup>-3</sup> )	Mean	0.53	0.45	0.40	0.37	0.38	0.43	0.43	0.45	0.50	0.53	0.52	0.51	0.46
	Max	0.56	0.51	0.50	0.41	0.47	0.55	0.56	0.56	0.57	0.58	0.56	0.57	-
	Min	0.43	0.39	0.37	0.33	0.32	0.26	0.32	0.35	0.40	0.47	0.42	0.34	-

### 361 3.3 Groundwater table depth

362 Average groundwater table depth (WTD) below terrain during the period was between 47 to 21 cm across the  
 363 transect (Fig. 2, Table 2). During summer, the peat drained between 18 – 31 cm below the annual average and in  
 364 winter the WTD increased to 0 – 22 cm above the annual average across the transect (Fig. 2, Table 2).  
 365 Generally, the WTD elevation was lower in the ditch across the entire study period (Fig. 2). It was only on the  
 366 northern end of the transect that the surface occasionally was flooded during winter periods (Fig. 2).

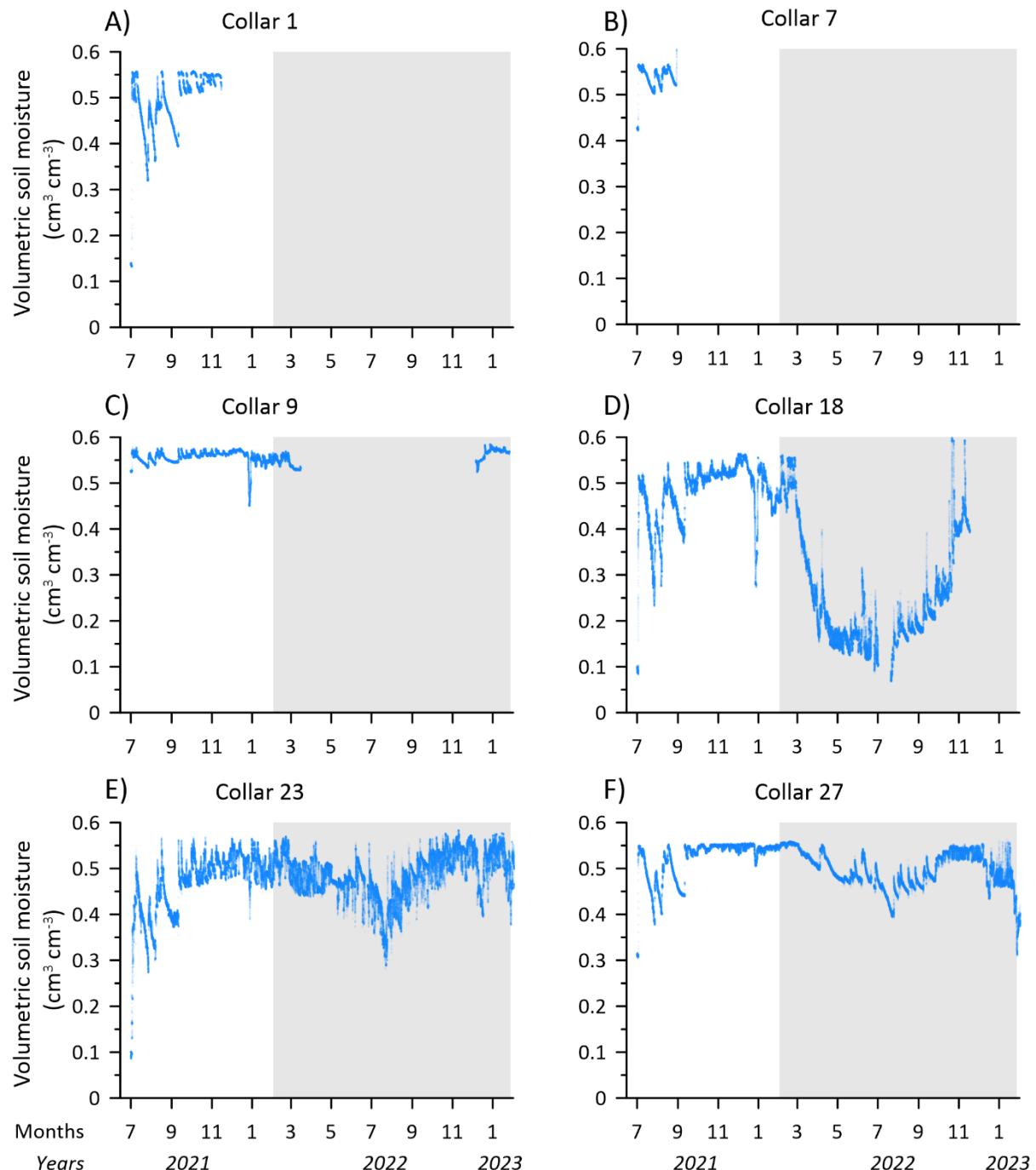


367 **Figure 5: Time series of groundwater table depth (WTD) below terrain for the six piezometer locations along the**  
 368 **SkyLine2D transect in the period March 31<sup>st</sup>, 2022 and January 31<sup>st</sup>, 2023 when the flux measurements stopped.**  
 369 **Dashed line show surface.**

370 The temporal variability of WTD was similar across the transect despite different absolute water table depths  
 371 (Fig. 5A-F). In the summer periods, the WTD was most variable decreasing to below 40 cm for collars 1, 13, 18,  
 372 22 and 27, whereas the WTD for collar 5 showed the deepest WTD measured over the transect. WTD responded  
 373 quickly (within hours) to precipitation events that could increase the WTD by almost 40 cm at some plots,  
 374 indicating that the entire aerated soil volume above the groundwater table was flooded. There was a slight  
 375 tendency to lower response to precipitation events for piezometers at collar 5 and collar 13 that were placed  
 376

377 closer to the ditch (Fig. 2 and Fig. 5B and C). As the ditch water level was lower than in the peat this could be  
 378 explained by more efficient lateral drainage into the ditch from the areas closer to the ditch. In the winter  
 379 periods, the WTD was less responsive to precipitation and was closer to the surface (Fig. 5A-F) across the  
 380 transect.

381 **3.5 Soil water content**



382  
 383 **Figure 6: Time series of volumetric soil water content ( $\text{cm}^3 \text{cm}^{-3}$ ) in 0-5 cm for the six collars 1, 7, 9, 18, 23 and 27**  
 384 along the SkyLine2D transect in the period July 1<sup>st</sup>, 2021 – January 31<sup>st</sup>, 2023 when the measurements terminated.  
 385 The periods of GHG measurements with the SkyLine2D system are shown with the shaded area.

386 Due to instrument failure the temporal coverage of soil moisture in the topsoil (5 cm) was not similar across the  
 387 transect (Fig. 6A-F). For collars 18, 23 and 27 the entire period of greenhouse gas measurements was covered  
 388 by soil moisture measurements (Fig. 6D-F). While SWC for collars 1, 9, 18, 23 and 27 was similar in the winter  
 389 periods (around  $0.55 \text{ cm}^3 \text{ cm}^{-3}$ ) the SWC for collar 18 decreased to lower minima between  $0.1 - 0.2 \text{ cm}^3 \text{ cm}^{-3}$ ,  
 390 than the minima observed between  $0.3 - 0.4 \text{ cm}^3 \text{ cm}^{-3}$  for collars 23 and 27 in the summer periods (Fig. 6, Table  
 391 2). Similar for all collars it was observed that SWC was more variable in summer, responding similarly as WTD  
 392 to precipitation events (Fig. 6, Table 2). Since plants were removed regularly from the collars the decrease of  
 393 SWC for collar 18 cannot be explained by plant transpiration, and the dynamic behaviour could indicate the  
 394 impact of soil evaporation, but the different levels of SWC also show that there is spatial variation across the  
 395 transect in the water retention properties of the peat soil that will impact the rate of drying. However, it cannot  
 396 be ruled out that the SWC sensor at collar 18 experienced malfunction or that soil contact was lost in the dry  
 397 periods of 2022 (Fig. 6D) which could lead to erroneous and too low SWC. Therefore, these data should be  
 398 considered with care.

### 399 **3.6 Peat soil characteristics**

400 **Table 3 Mean ( $\pm$ standard error of the mean (SE)) peat/organic soil characteristics of humification degree (Von Post),**  
 401 **pH (H<sub>2</sub>O), dry bulk density ( $\rho_{\text{dry}}$ ), total C (TC) concentration, total N concentration (TN) and the C/N ratio for collars**  
 402 **1, 2, 5, 6, 8 and 13 - 27 at the Vejrumbro transect.**

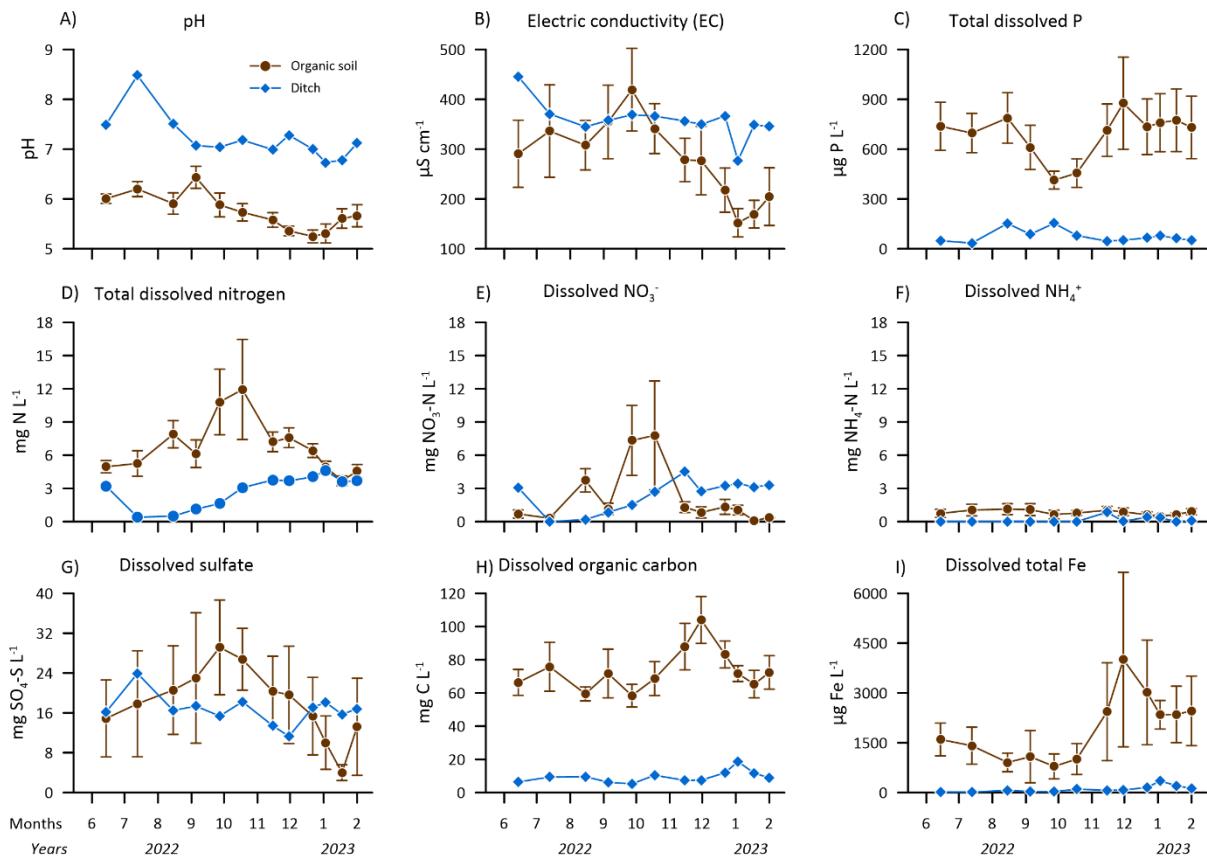
Depth (cm)	N	Von post		pH (H <sub>2</sub> O)		$\rho_{\text{dry}}$ (g cm <sup>-3</sup> )		TC (%)		TN (%)		C/N	
		Min	Max	Mean	$\pm$ SE	Mean	$\pm$ SE	Mean	$\pm$ SE	Mean	$\pm$ SE	Mean	$\pm$ SE
0-20	20	7	10	4.2	0.08	0.31	0.02	26	1.1	1.6	0.06	16	0.4
20-40	20	5	10	4.6	0.06	0.20	0.01	43	1.3	1.8	0.04	24	0.7
40-60	11	3	8	4.9	0.10	0.15	0.01	48	1.8	1.9	0.05	25	1.1
60-80	11	3	6	5.3	0.09	0.11	0.01	47	1.8	1.9	0.05	24	0.6
80-100	10	1	8	5.4	0.09	0.10	0.02	44	2.1	1.9	0.05	24	0.6

403 Generally, there was peat/organic soil to one meter depth except for one collar (25) where gyttja was found in a  
 404 depth of 80 cm (Table 3). The organic soil was more decomposed in the top 40 cm indicated by higher Von Post  
 405 values between 5 and 10. Below 40 cm peat still displayed high levels of decomposition along the transect, but  
 406 was more often found to be less decomposed, values ranging from 1-8 (Table 3). This corresponds well to the  
 407 previous land use with drainage of the topsoil leading to higher degree of humification. Also, the organic soil  
 408 was most dense in the top 20 cm (on average  $0.31 \pm 0.02 \text{ g cm}^{-3}$ ) and bulk density decreased to  $0.10 - 0.12 \text{ g cm}^{-3}$   
 409 from 40 – 100 cm depth. Total C and N was lowest in the 0-20 cm layer, but still classified as organic soil.  
 410 Below 20 cm total C and N concentrations, respectively were similar. C/N ratio was lowest in the top 20 cm  
 411 ( $16 \pm 0.4$ ) and increased to 22-25 in 20 – 100 cm depth (Table 3).

### 412 **3.7 Groundwater and ditch water chemical composition**

413 Site mean pH of the groundwater in the organic soil was  $5.8 \pm 0.1$  and was lower than the pH of the ditch  
 414 ( $7.3 \pm 0.6$ ). There was a tendency towards lower pH in groundwater and ditch towards the end of the  
 415 measurement period (Fig. 7A). Electric conductivity was generally higher in the ditch water ( $359 \pm 36 \mu\text{S cm}^{-1}$ )  
 416 compared to the groundwater in the organic soil ( $276 \pm 18 \mu\text{S cm}^{-1}$ ) but varied less over the season. The  
 417 groundwater shows a clear peak in EC around September 2022 (Fig. 7B). Total dissolved P was markedly

418 higher in the groundwater ( $687\pm45 \mu\text{g P L}^{-1}$ ) compared to the ditch water ( $76\pm10 \mu\text{g P L}^{-1}$ ). Whereas there was  
419 little seasonal trend in ditch P concentrations, dissolved P in groundwater dipped to below average  
420 concentrations between August to October, likely indicating plant uptake during the growing season (Fig. 7C).  
421 Similarly, total dissolved N was higher in groundwater ( $6.7\pm0.5 \text{ mg N L}^{-1}$ ) than in ditch ( $2.6\pm1.6 \text{ mg N L}^{-1}$ ) with  
422 increasing concentrations during the growing season (Fig. 7D). This temporal trend was also observed for  $\text{NO}_3^-$   
423 (Fig. 7E), but average groundwater ( $2\pm0.5 \text{ mg NO}_3\text{-N L}^{-1}$ ) and ditch ( $2.2\pm1.5 \text{ mg NO}_3\text{-N L}^{-1}$ ) concentrations  
424 were similar. As expected, dissolved  $\text{NH}_4\text{-N}$  was lowest among investigated N-species and there was more  
425 dissolved  $\text{NH}_4\text{-N}$  present in groundwater ( $0.8\pm0.1 \text{ mg NH}_4\text{-N L}^{-1}$ ) than in the ditch ( $0.14\pm0.25 \text{ mg NH}_4\text{-N L}^{-1}$ ).  
426 However, there was no discernable temporal trend for  $\text{NH}_4^+$  (Fig. 7F). Collectively, the temporal trend of TN  
427 and  $\text{NO}_3^-$  could point to temperature driven mineralization of the peat. Also, the organic N (TN – inorganic N-  
428 species) was on average 10 times higher in the groundwater than in the ditch. Average  $\text{SO}_4^{2-}$  concentrations  
429 were similar between the groundwater ( $17.5\pm2.4 \text{ mg SO}_4\text{-S L}^{-1}$ ) and ditch ( $17\pm1.5 \text{ mg SO}_4\text{-S L}^{-1}$ ), but  $\text{SO}_4^{2-}$   
430 concentration peaked during September and October in the groundwater whereas it remained more constant in  
431 the ditch over the season (Fig. 7G). Like the dissolved organic N, DOC concentrations were consistently higher  
432 in the groundwater ( $73\pm3.1 \text{ mg DOC L}^{-1}$ ) than in the ditch ( $9.4\pm3.5 \text{ mg DOC L}^{-1}$ ), but peaked later in the season,  
433 around December 2022, whereas there was little temporal variability of DOC in the ditch (Fig. 7H). Dissolved  
434 total Fe displayed the same temporal trend as DOC (Fig. 7I) but was higher groundwater ( $1916\pm163 \mu\text{g Fe L}^{-1}$ )  
435 compared to the ditch ( $98\pm95 \mu\text{g Fe L}^{-1}$ ). The geochemical parameters of groundwater and ditch water point to  
436 different mechanisms regulating peat decomposition and possibly plant uptake, where the chemical composition  
437 of groundwater varied more over time than ditch water. Generally, there were no systematic spatial pattern of  
438 groundwater chemistry across the transect.

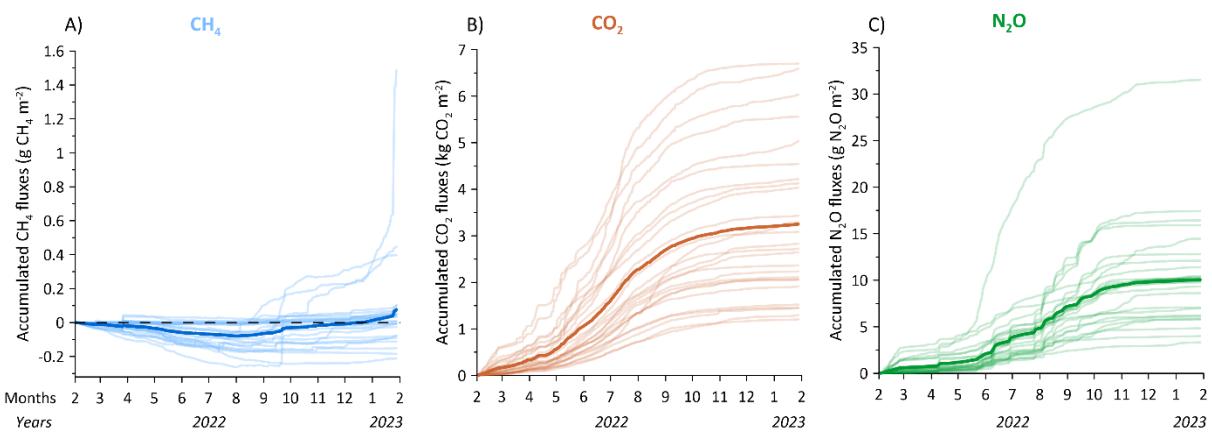


439

440 **Figure 7** Groundwater (brown closed circles) and ditch water (closed blue diamonds) chemistry at Vejrumbro for the  
 441 period June 2022 to February 2023 for A) pH, B) Electric conductivity and dissolved C) total phosphor (P), D) total  
 442 nitrogen (N), E) nitrate ( $\text{NO}_3^-$ ), F) ammonium ( $\text{NH}_4^+$ ), G) sulfate ( $\text{SO}_4^{2-}$ ), H) organic carbon and I) total iron (Fe).  
 443 Values for organic soils are means for the transect with error bars showing the standard error of the mean (N=6 per  
 444 sampling date).

#### 445 3.8 Net soil and ditch $\text{CO}_2$ , $\text{CH}_4$ and $\text{N}_2\text{O}$ fluxes

##### 446 3.8.1 Spatial variation of net soil $\text{CO}_2$ , $\text{CH}_4$ and $\text{N}_2\text{O}$ fluxes



447

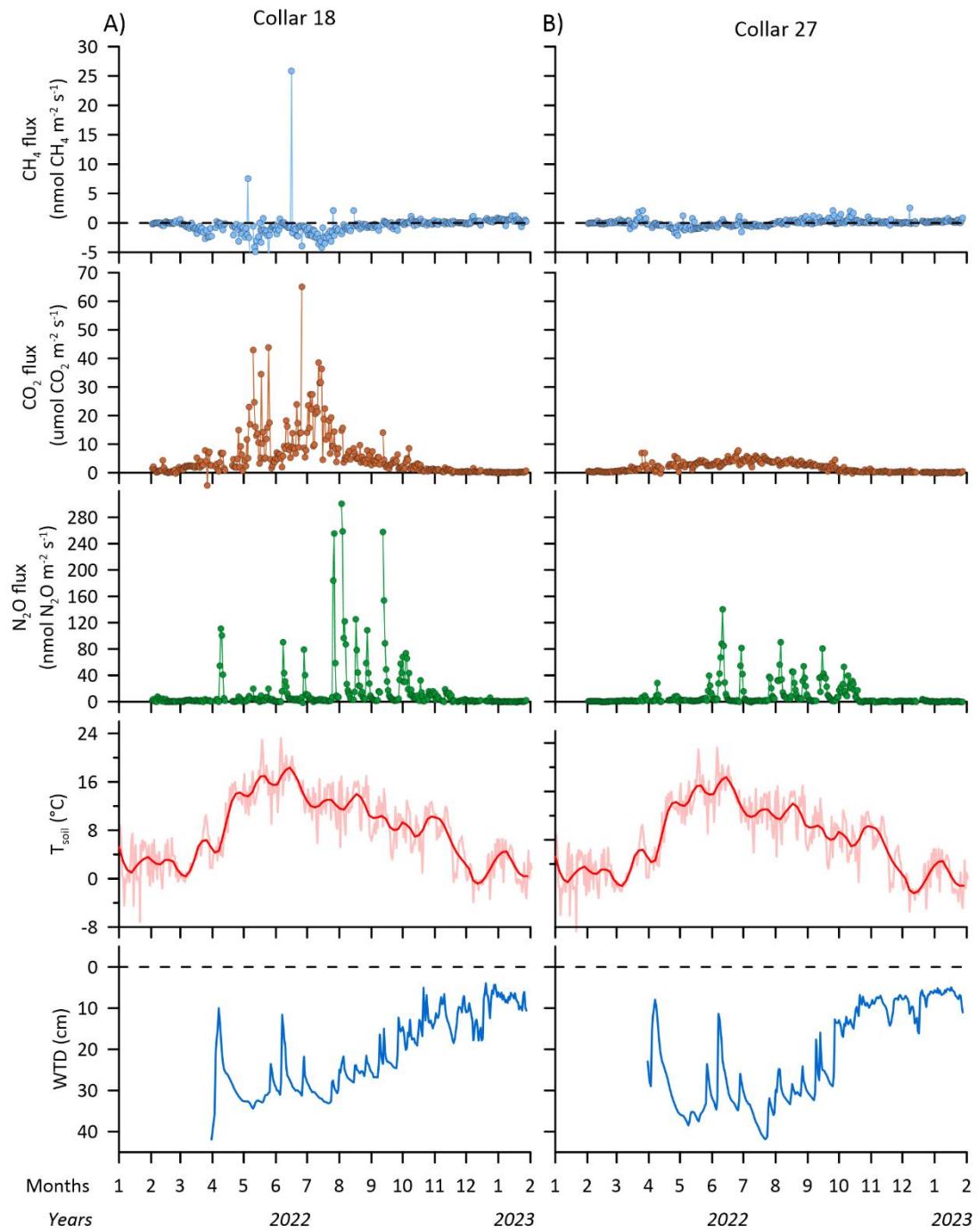
448 **Figure 8:** Cumulative fluxes of A)  $\text{CH}_4$ , B)  $\text{CO}_2$ , and C)  $\text{N}_2\text{O}$  for 26 individual collars along the SkyLine2D transect in  
 449 the measurement period February 2022 to January 2023. Units for  $\text{CH}_4$  and  $\text{N}_2\text{O}$  are in  $\text{g CH}_4/\text{N}_2\text{O m}^{-2}$  and for  $\text{CO}_2$   
 450 in  $\text{kg CO}_2 \text{m}^{-2}$ . The cumulative fluxes represent the raw dataset. The ditch data was excluded. Transect average is  
 451 shown as thick lines.

452 Within the transect, cumulative CH<sub>4</sub> fluxes over the study period (360 days) varied between -0.21 to 1.48 g CH<sub>4</sub>  
453 m<sup>-2</sup> over the study period, with a transect average ( $\pm$ SE) cumulative flux of 0.07 $\pm$ 0.06 g CH<sub>4</sub> m<sup>-2</sup> (Fig. 2 and Fig.  
454 8A). Out of the 26 collars, excluding the ditch collar, 11 displayed a net uptake over the measurement period  
455 and the remaining were small net emitters (Fig. 2 and Fig. 8A). There was generally little spatial variation in the  
456 absolute CH<sub>4</sub> fluxes among the soil collars, but three collars (11, 12 and 15) showed increasing net positive  
457 cumulative fluxes towards the ditch (Fig. 2). The low spatial and similar temporal variation between collars  
458 indicate both hydrological indicators of SWC and WTD are poor predictors of CH<sub>4</sub> fluxes across the transect.  
459 However, as we excluded plants from the collars we might have decreased the net emission of CH<sub>4</sub> directly by  
460 restricting gas transport in aerenchyma from deep peat layers potentially sustaining net CH<sub>4</sub> emission even  
461 though the observed growing season WTD was 20-40 cm (Askaer et al., 2011; Vroom et al., 2022) and  
462 indirectly by potentially reducing plant carbon supply to methanogens. The lack of consistent hot moments of  
463 CH<sub>4</sub> emissions, low cumulative emissions during periods of shallow WTD in the growing season (Fig. 5A-F) is  
464 in line with the measured free NO<sub>3</sub><sup>-</sup>, SO<sub>4</sub><sup>2-</sup>, Fe ions (Fig. 7E, G, I) in the groundwater. It is well known that the  
465 presence of other electron acceptors, such as sulphur, iron and nitrate, inhibit CH<sub>4</sub> production (Bridgman et al.,  
466 2013), in turn limiting net CH<sub>4</sub> emission. Also, the often deeper WTD in the summer between to 40 cm below  
467 terrain also suggest that CH<sub>4</sub> oxidation could aid to reduce net CH<sub>4</sub> emission from the peat (Christiansen et al.,  
468 2016).

469 The CO<sub>2</sub> effluxes displayed tremendous spatial variation across the 24-meter transect (Fig. 2 and Fig. 8B) and  
470 measurements indicated that the organic soil was a net source of CO<sub>2</sub>, with cumulative fluxes over the study  
471 period ranging between 1214 – 6740 g CO<sub>2</sub> m<sup>-2</sup>, and a transect average ( $\pm$ SE) of 3269 $\pm$ 328 g CO<sub>2</sub> m<sup>-2</sup>, over the  
472 study period of 360 days (Fig. 2 and Fig. 8B). There was no apparent relation between the magnitude of  
473 cumulative CO<sub>2</sub> efflux to the position along the transect and average WTD (Fig. 2). The cumulative net soil CO<sub>2</sub>  
474 emission is equal to 8.9 tCO<sub>2</sub>-C ha<sup>-1</sup> y<sup>-1</sup> (range of 3.3 to 18 tCO<sub>2</sub>-C ha<sup>-1</sup> y<sup>-1</sup> across the transect) and compares  
475 well to estimates of annual soil C loss (8.8 tCO<sub>2</sub>-C ha<sup>-1</sup> y<sup>-1</sup>) from a drained unfertilized grassland on organic soil  
476 in Denmark (Kandel et al., 2018) as well as annual carbon budgets of similar Danish, British and German  
477 wetlands (Evans et al., 2021; Koch et al., 2023; Tiemeyer et al., 2020).

478 Similarly, the particular site at Vejrumbro where the SkyLine2D was located was overall a net source of N<sub>2</sub>O,  
479 with cumulative fluxes ranging between 3.3 – 32 g N<sub>2</sub>O m<sup>-2</sup>, with a transect average ( $\pm$ SE) of 10.1 $\pm$ 1.1 g N<sub>2</sub>O m<sup>-2</sup>  
480 (Fig. 2 and Fig. 8C) over the study period (360 days). Thus, there is a 10-fold difference between minimum  
481 and maximum cumulative N<sub>2</sub>O fluxes within the transect, without any apparent relation to the position along the  
482 transect and WTD. The highest cumulative N<sub>2</sub>O fluxes occurred at collar 8 situated close to the ditch (Fig.  
483 2). The transect average cumulative N<sub>2</sub>O emission is equivalent to a net N loss from N<sub>2</sub>O emission alone of 64  
484 kg N ha<sup>-1</sup> y<sup>-1</sup>, was very high and exceeding previously reported fluxes from the Vejrumbro site (1.5 – 2.1 g N<sub>2</sub>O  
485 m<sup>-2</sup> y<sup>-1</sup>) (Nielsen et al., 2024) and German organic soils (0.04 – 6.3 g N<sub>2</sub>O m<sup>-2</sup> y<sup>-1</sup> for grassland and cropland  
486 land uses) (Tiemeyer et al., 2020). The high N<sub>2</sub>O emission from the transect during the measurement period  
487 indicate that N<sub>2</sub>O may in fact dominate the GHG budget in relation to the global warming potential at this  
488 specific location at the Vejrumbro site had gross primary production (reducing net ecosystem CO<sub>2</sub> emission)  
489 been included in the measurements. It is important to reiterate here that the flux measurements of this study  
490 were done on bare soil whereas the studies referenced above included vegetation.

491 The high N<sub>2</sub>O fluxes may be a result of high rates of denitrification in the subsoil from either *in situ* produced  
492 NO<sub>3</sub><sup>-</sup> from peat decomposition or as NO<sub>3</sub>-enriched agricultural runoff from the surrounding intensively  
493 cultivated areas, which was not affecting groundwater NO<sub>3</sub><sup>-</sup> concentration in the center of the wetland with  
494 lower N<sub>2</sub>O (Nielsen et al., 2024). The groundwater enters the northern peripheral zone of the wetland at  
495 Vejrumbro coinciding with the position of the measurement transect. The highest NO<sub>3</sub><sup>-</sup> concentrations in  
496 groundwater at the SkyLine2D transect corresponded roughly with highest N<sub>2</sub>O emissions during summer and  
497 early autumn (Fig. 7D-F and Fig. 8C), but the frequency of water sampling was too low to fully link  
498 groundwater NO<sub>3</sub><sup>-</sup> temporal dynamics to N<sub>2</sub>O emissions.

499 **3.8.2 Temporal variability of net soil CO<sub>2</sub>, CH<sub>4</sub> and N<sub>2</sub>O fluxes**500 **3.8.2.1 Time series of raw data of net soil CO<sub>2</sub>, CH<sub>4</sub> and N<sub>2</sub>O fluxes**

501 **Figure 9: Examples of daily average time series of CH<sub>4</sub>, CO<sub>2</sub> and N<sub>2</sub>O fluxes for collars 18 and 27 at the SkyLine2D**  
 502 **transect in Vejrumbro, soil temperature (T<sub>soil</sub>) in celsius (°C) and groundwater table depth (WTD) in cm below**  
 503 **terrain is shown in two lower panels for the measurement period from February 2022 to January 2023.**

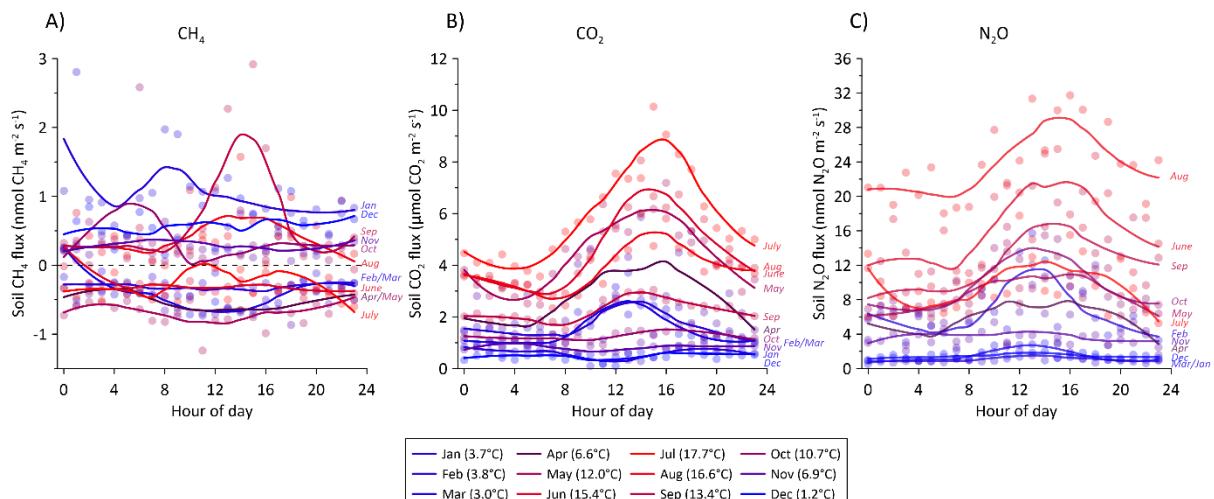
504 With the high frequency of GHG flux measurements (on average 5 measurements per day per collar) it was  
 505 possible to observe short term flux phenomena that in most studies deploying manual chambers are missed or if  
 506 captured can lead to biased conclusions on flux magnitudes. For example, in most of the measurement points,  
 507

508 CH<sub>4</sub> fluxes were generally near zero, but occasionally displayed elevated net emission for short periods even in  
 509 periods with deeper WTD (Fig. 9A) for most chambers (see supplementary Fig. S5). This flux dynamic might  
 510 be related to episodic release of accumulated CH<sub>4</sub> from deeper soil layers that are not fully oxidized in the  
 511 aerated root zone and that were not released through plants (Askaer et al., 2011). As plants were not included in  
 512 the collars these bursts cannot be attributed to plant emission pathways.

513 Generally, it was observed that soil CO<sub>2</sub> fluxes increased over the season with increasing temperature. However,  
 514 for some collars displayed rapid bursts of CO<sub>2</sub> emissions (example in Fig. 9A), while other collars at the same  
 515 period did not display this behaviour (Fig. 9B). This dynamic points to different emission pathways from the  
 516 soil not related to plant mediated transport. Thus, while we purposely omitted aboveground autotrophic  
 517 respiration by clipping the vegetation, it cannot be ruled out that living roots inhabited the soil below the  
 518 chambers and hence contributed to the observed CO<sub>2</sub> emission rates.

519 For N<sub>2</sub>O, the spatiotemporal pattern was even more pronounced than for CO<sub>2</sub>, with N<sub>2</sub>O primarily emitted in  
 520 bursts related to rapidly increasing or decreasing WTD that coincided with precipitation events. In drier periods  
 521 with deeper WTD and little fluctuations, N<sub>2</sub>O fluxes quickly dropped to near zero (Fig. 9A and B). Despite N<sub>2</sub>O  
 522 being emitted in similar temporal patterns across the transect, the magnitude of the N<sub>2</sub>O peaks were not similar  
 523 across the transect (Fig. 2, 8 and supplementary Fig. S5). Hence, the majority of N<sub>2</sub>O is emitted in hot moments  
 524 is likely driven by fluctuations in WTD (Fig. 9) as it has also been shown in other drained temperate peatland  
 525 soils (Anthony and Silver, 2023).

### 526 3.8.2.2 Diurnal variation of net soil CO<sub>2</sub>, CH<sub>4</sub> and N<sub>2</sub>O fluxes



527  
 528 **Figure 10: Average hourly flux for all soil collars of A) CH<sub>4</sub>, B) CO<sub>2</sub>, and C) N<sub>2</sub>O during a 24 hour period. The**  
 529 **diurnal variation is split between each month during the 2022-2023 measurement period. The fluxes were assigned**  
 530 **the hour of measurement during the day and averaged per month. Color shade between blue and red corresponds to**  
 531 **average air temperature for the specific month shown in parenthesis in the figure legend. Solid lines are loess fits for**  
 532 **visualization of the diurnal variation in each month.**

533 With the SkyLine2D system we observed a clear diurnal cycle for CO<sub>2</sub> and N<sub>2</sub>O fluxes, but not for CH<sub>4</sub> (Fig.  
 534 10A-C). The lack of diurnal variability of CH<sub>4</sub> fluxes could also be due the removal of plants from the collars  
 535 that would have facilitated light-driven fluxes (Askaer et al. 2011). The amplitude of diurnal variability

increased with higher air temperature for CO<sub>2</sub> (Fig. 10B) and partly for N<sub>2</sub>O (Fig. 10C). The month of July was an exception as it resembled the pattern observed in May although the July soil temperature was about 5°C higher (Table 2). The lower N<sub>2</sub>O fluxes observed in July can be attributed to lower and more constant WTD in July compared to May, June and September across the transect (Fig. 5). Diurnal variability of soil CO<sub>2</sub> fluxes are well known and can be related to both increased heterotrophic respiration during the warmer day and autotrophic respiration in response to photosynthesis. Previously, similar diurnal patterns of N<sub>2</sub>O emissions were observed in a Danish fen (Jørgensen et al., 2012).

### 3.8.2.3 Monthly variability of net soil GHG fluxes

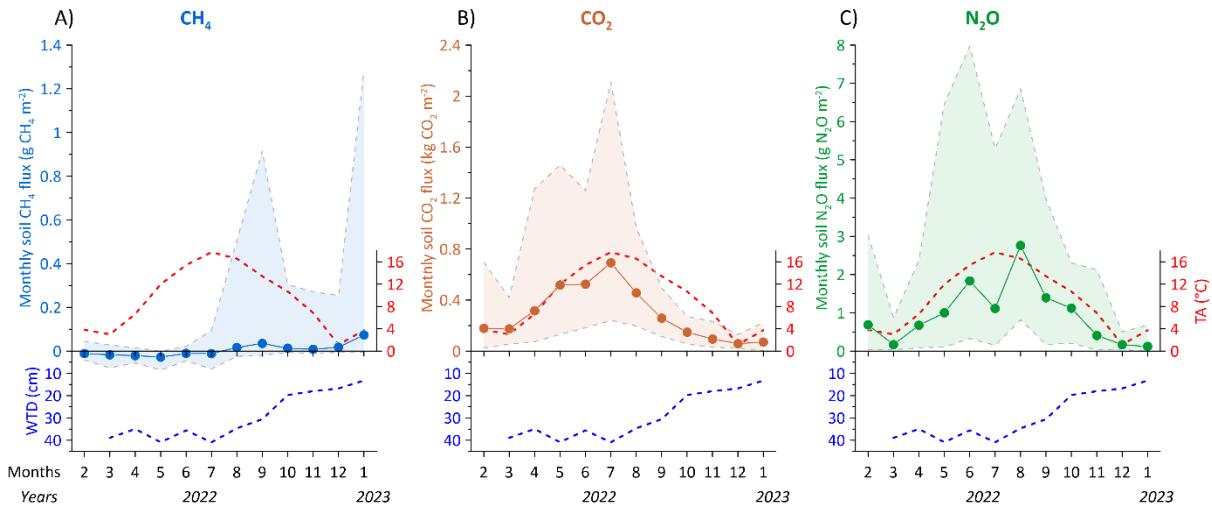


Figure 11: Monthly summed soil fluxes of A) CH<sub>4</sub> in g CH<sub>4</sub> m<sup>-2</sup>, B) CO<sub>2</sub> in kg CO<sub>2</sub> m<sup>-2</sup>, and C) N<sub>2</sub>O in g N<sub>2</sub>O m<sup>-2</sup> for all organic soil collars for the measurement period from February 2022 to January 2023. Shaded areas for CH<sub>4</sub>, CO<sub>2</sub> and N<sub>2</sub>O graphs represent the maximum and minimum monthly average fluxes. Blue dashed line below CH<sub>4</sub>, CO<sub>2</sub> and N<sub>2</sub>O represent the measured monthly average transect groundwater table depth (WTD) in cm below terrain. Red dashed line shows the monthly average air temperature (TA).

The average soil GHG fluxes for all collars were summed to monthly transect sums to illustrate long term drivers on the flux magnitude. Overall, monthly sums of CO<sub>2</sub> and N<sub>2</sub>O emissions increase with temperature and fluxes are highest under deeper WTD, but CH<sub>4</sub> net fluxes were less responsive to long term changes in both temperature and hydrology (Fig. 11A-C). Net uptake of CH<sub>4</sub> increased slightly with increasing temperature and lower WTD during the spring and summer. With increasing water table and high temperatures in August the soils across the transect turned into a small net CH<sub>4</sub> source continuing in fall and winter (Fig. 11A).

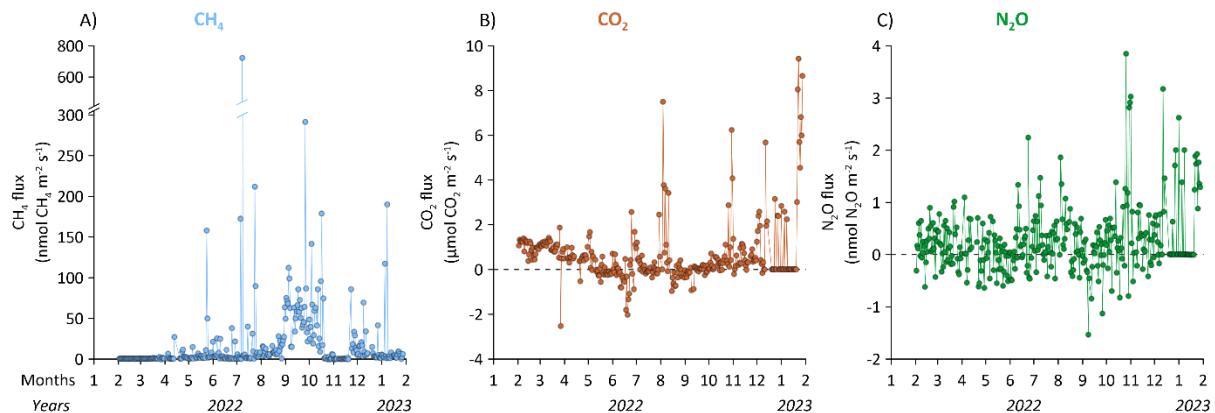
For CO<sub>2</sub> the seasonal variation was pronounced and closely followed soil temperature until peak values in July for both transect average, minimum and maximum fluxes, respectively (Fig. 11B). From July to August, it was observed that WTD across the transect began to increase again and CO<sub>2</sub> fluxes departed from the close relation to soil temperature, indicating an inhibitory role of the WTD in this period, but reaching minimum fluxes in December, corresponding to the wettest and coldest month (Fig. 11B).

Similarly, N<sub>2</sub>O fluxes increased with soil temperature reaching peak monthly values in August, corresponding to the period of the year with highest soil temperature and increasing WTD (Fig. 11C). This supports the

563 promoting role of soil water saturation on the production of  $\text{N}_2\text{O}$  when temperature is favourable for  
564 denitrification.  $\text{N}_2\text{O}$  fluxes reached minimum values in December when WTD and ST were lowest (Fig. 11C).

### 565 3.8.3 Ditch $\text{CO}_2$ , $\text{CH}_4$ and $\text{N}_2\text{O}$ fluxes

#### 566 3.8.3.1 Time series of raw data of ditch $\text{CO}_2$ , $\text{CH}_4$ and $\text{N}_2\text{O}$ fluxes



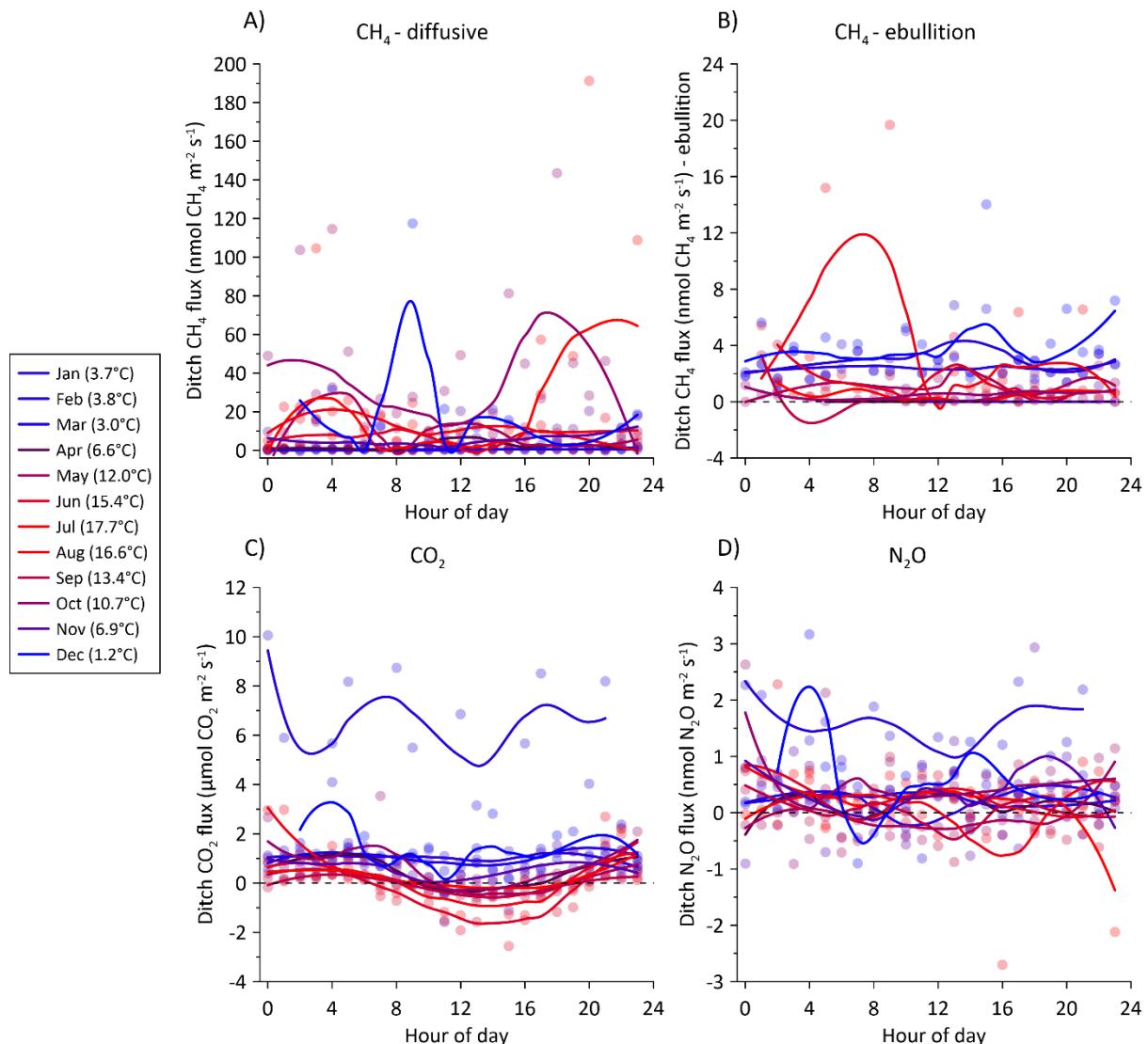
567 **Figure 12: Daily average time series of net ditch total A)  $\text{CH}_4$  (diffusion and ebullition), B)  $\text{CO}_2$ , and C)  $\text{N}_2\text{O}$  fluxes at**  
568 **the Vejrumbro site for the measurement period from February 2022 to January 2023.**

569 Common for all three gases is that ditch emissions are dynamic and net fluxes change from zero to large net  
570 positive or negative fluxes within hours or days (Fig. 12A-C). Compared to net soil  $\text{CH}_4$  fluxes the ditch can be  
571 considered an emission hotspot at the Vejrumbro site (sum of diffusion and ebullition:  $8.3 \text{ g CH}_4 \text{ m}^{-2} \text{ y}^{-1}$ ), but  
572 fluxes are lower than earlier reports for ditches in other drained wetlands (between  $0.1 - 44.3 \text{ g CH}_4 \text{ m}^{-2} \text{ y}^{-1}$ )  
573 (Peacock et al., 2021). Methane varies most throughout the measurement period with maximum diffusive flux  
574 close to  $700 \text{ nmol CH}_4 \text{ m}^{-2} \text{ s}^{-1}$  and there was a tendency toward consistently higher net  $\text{CH}_4$  emission from  
575 August to September, becoming close to zero in colder seasons (Fig. 12A). Ebullition of  $\text{CH}_4$  did occur  
576 occasionally in the ditch, e.g. about 19.3% of flux measurements for the ditch was comprised of ebullitions but  
577 constituted on average only 2.9% of the total  $\text{CH}_4$  emission ( $0.24 \text{ g CH}_4 \text{ m}^{-2} \text{ y}^{-1}$ ) from the ditch which is lower,  
578 but in the same range as a recent estimate from a ditch in a similar drained German peatland (Köhn et al., 2021).  
579 According to the flux calculation methodology, flux separation and extrapolation to daily sums, diffusive fluxes  
580 dominated ( $6.56 \text{ g CH}_4 \text{ m}^{-2} \text{ y}^{-1}$ ). However, it cannot be ruled out that the classification as diffusive flux may in  
581 fact be ebullition by nature. It has been suggested that microbubbles resulting from mass transport can resemble  
582 diffusive fluxes in a chamber making it difficult, if not impossible, to fully separate the two emission  
583 mechanisms in a continuous time series if headspace  $\text{CH}_4$  concentrations do not abruptly increase (Prairie and  
584 del Giorgio, 2013), such as in the example shown in Fig. S4.

585 For  $\text{CO}_2$ , there was a general tendency towards lower fluxes during the summer months and fluxes increased in  
586 magnitude and variability towards the end of the study period (Fig. 12B). For  $\text{N}_2\text{O}$ , the fluxes fluctuated around  
587 zero for most of the study period, except towards the end (December and January) where net fluxes became  
588 positive (Fig. 12C). Compared to the net soil  $\text{N}_2\text{O}$  and  $\text{CO}_2$  fluxes the ditch fluxes of these gases are low  
589 showing that the ditch is not contributing significantly to the  $\text{CO}_2$  and  $\text{N}_2\text{O}$  budget at the Vejrumbro site.

591 Per square meter, the ditch emitted less  $\text{N}_2\text{O}$  ( $0.41 \text{ g N}_2\text{O m}^{-2}$  or  $2.6 \text{ kg N}_2\text{O-N ha}^{-1} \text{ y}^{-1}$ ) and  $\text{CO}_2$  ( $961 \text{ g CO}_2 \text{ m}^{-2}$   
 592  $\text{y}^{-1}$  or  $2.6 \text{ tCO}_2\text{-C ha}^{-1} \text{ y}^{-1}$ ) than the organic soil, but was a hotspot of  $\text{CH}_4$  emission ( $8.4 \text{ g CH}_4 \text{ m}^{-2} \text{ y}^{-1}$  or  $63 \text{ kg}$   
 593  $\text{CH}_4\text{-C ha}^{-1} \text{ y}^{-1}$ ) during the measurement period. Although these emissions estimates are lower than previously  
 594 reported for ditches in organic soil (up to  $44 \text{ g CH}_4 \text{ m}^{-2} \text{ y}^{-1}$ ) (Peacock et al., 2021). For the ditch  $\text{CH}_4$  budget,  
 595 ebullition only constitutes 2.9% of net  $\text{CH}_4$  emissions during the study period. This proportion may be  
 596 underestimated as the count of ebullition events may have been underestimated (Prairie and del Giorgio, 2013).

597 **3.8.3.2 Diurnal variability in ditch fluxes**

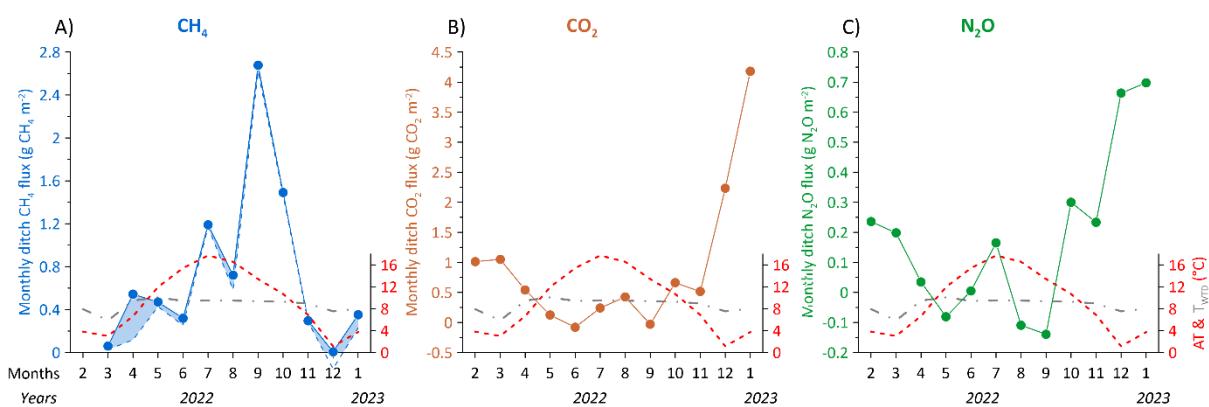


598  
 599 **Figure 13: Average hourly fluxes for the ditch collar of A) diffusive  $\text{CH}_4$  fluxes, B)  $\text{CH}_4$  ebullition fluxes, C)  $\text{CO}_2$ , and**  
 600 **C)  $\text{N}_2\text{O}$  during a 24 hour period. The fluxes were assigned the hour of measurement during the day and averaged per**  
 601 **month. The diurnal variation is split between each month during the 2022-2023 measurement period. Color shade**  
 602 **between blue and red corresponds to average air temperature for the specific month shown in parenthesis in the**  
 603 **figure legend. Solid lines are loess fits for visualization of the diurnal variation in each month. Note different axes.**

604 For  $\text{CH}_4$  fluxes, both diffusion and ebullition, there was no clear diurnal variability in any month (Fig. 13A and  
 605 B). This is expected for ebullition emissions which is known to be erratic without any clear diurnality (Sø et al.,  
 606 2023; Wik et al., 2016). For net  $\text{CO}_2$  fluxes from the ditch there was no diurnal variability in colder seasons

607 (Jan, Feb, Mar, Nov and Dec), but consistent positive net CO<sub>2</sub> efflux (Fig. 13C). Diurnal patterns became clearer  
 608 with higher temperatures from May to October (Fig. 13C) and in this period CO<sub>2</sub> fluxes decreased during the  
 609 day to sometimes reach net negative fluxes (net uptake of CO<sub>2</sub>) during and after midday (Fig. 13C), although the  
 610 net emissions were also observed in the daytime period (Fig. 13C). The net negative fluxes can likely be  
 611 explained by photosynthetic activity of aquatic plants on the surface of the ditch or by algae in the water column  
 612 which was measured due to the transparency of the chamber. Using an opaque chamber instead would likely  
 613 have resulted in different net CO<sub>2</sub> efflux in daytime. For N<sub>2</sub>O, the same pattern as for CH<sub>4</sub> was observed, where  
 614 flux magnitude across the day fluctuated around zero, except for January where N<sub>2</sub>O fluxes were consistently  
 615 above zero (Fig. 13D).

### 616 3.8.3.3 Monthly variability in ditch fluxes



617 **Figure 14: Monthly summed ditch fluxes of A) CH<sub>4</sub> in g CH<sub>4</sub> m<sup>-2</sup>, B) CO<sub>2</sub> in g CO<sub>2</sub> m<sup>-2</sup> and C) N<sub>2</sub>O in g N<sub>2</sub>O m<sup>-2</sup> for**  
 618 **the measurement period from February 2022 to January 2023. In A) the blue dashed line is the contribution of**  
 619 **diffusive fluxes and the shaded blue area between the full and dashed blue lines represent the monthly contribution of**  
 620 **ebullition to the total flux. Red and grey dashed lines show the monthly average air (AT) and groundwater**  
 621 **temperature (TwTD) in °C, respectively.**

623 The monthly sums of CH<sub>4</sub> tend to increase with air temperature, although peak CH<sub>4</sub> emissions (September)  
 624 occurred after air temperature peak (July) (Fig. 14A). Diffusive fluxes comprised the major emission pathway of  
 625 CH<sub>4</sub> in the ditch (between 21% - 99%), with the contribution from ebullition being highest in March (55%) and  
 626 April (78%) (Fig. 14A). Water temperature in the ditch was relatively stable throughout the year, varying  
 627 between 5.8 – 10.1°C being highest from April to November and lowest from December to March. However,  
 628 there is little indication of a direct relation between ditch water temperature and net GHG fluxes (Fig. 14A-C).  
 629 For CO<sub>2</sub> and N<sub>2</sub>O, the seasonal pattern is reversed with lowest fluxes during the warmest periods, approaching  
 630 net zero or even net negative fluxes (Fig. 14B and C).

## 631 4 Data availability

632 Data for this publication is available for download via <https://doi.org/10.60612/DATADK/BZQ8JE>.

## 633 5 Conclusion

634 The dataset presented here is unique for temperate fens and demonstrates the advantage of using automated  
 635 GHG measurement systems to resolve temporal and spatial patterns of GHG dynamics in high detail. It

636 represents a full year of data from 2022–2023 and must be considered specific to this period and the location at  
637 Vejrumbro. Consequently, it is expected that the annual budget of all GHGs in other years will likely differ due  
638 to varying climatic and hydrological conditions.

639 Specifically, the dataset demonstrates how temporal variation in soil hydrology and temperature is linked to the  
640 temporal variation of fluxes. Interestingly, the temporal variability of GHG fluxes across the transect appears to  
641 be lower than the spatial variation highlighting that spatial variability in hydrology and temperature may not  
642 necessarily be the best predictor of flux magnitudes across the transect. The cause of spatial variability in GHG  
643 fluxes remains unresolved and does not clearly link directly to either water table depth (WTD), soil temperature,  
644 or soil/groundwater chemical parameters.

645 The initial harvest and herbicide application represent ecosystem disturbances that could potentially alter soil  
646 biogeochemistry. However, these were conducted months prior to the start of flux measurements, minimizing  
647 the direct effect of herbicide. Continued plant removal from inside the collars was necessary for flux  
648 measurements, meaning the fluxes may only be regarded as net soil GHG fluxes and not representative of net  
649 ecosystem exchange. Excluding vegetation likely influenced measured fluxes of soil respiration (e.g., excluding  
650 root exudates) and reduced plant-mediated CH<sub>4</sub> and N<sub>2</sub>O emissions, potentially also reducing interannual  
651 variability.

652 *Carbon dioxide fluxes:* The magnitude of annual cumulative CO<sub>2</sub> fluxes is in the same range as other studies of  
653 temperate fens. Temporal variability is largely governed by the seasonality of WTD and soil temperature (T<sub>soil</sub>).  
654 Soil CO<sub>2</sub> fluxes showed diurnal variability with higher fluxes during midday, where the amplitude between  
655 night and day was augmented with T<sub>soil</sub>.

656 *Nitrous oxide fluxes:* Cumulative soil N<sub>2</sub>O fluxes exceed previously reported values for temperate fens at the  
657 Vejrumbro site and others. Unlike CO<sub>2</sub>, N<sub>2</sub>O is emitted largely in pulses related to rapid fluctuations of WTD,  
658 which increase in size with T<sub>soil</sub>, indicating a seasonal regulation of N<sub>2</sub>O production by temperature. These  
659 measurements suggest an important but difficult-to-capture dynamic of N<sub>2</sub>O in peatlands, where hot moments  
660 during warm periods determine most of the annual emissions. Soil N<sub>2</sub>O fluxes also showed diurnal variability  
661 similar to CO<sub>2</sub>.

662 *Methane fluxes:* The peat soils across the transect were insignificant sources of CH<sub>4</sub> during the measurement  
663 period. This could be linked to deeper WTD (20–40 cm) during summer, a cold wet winter, and the presence of  
664 alternative electron acceptors (NO<sub>3</sub><sup>-</sup>, SO<sub>4</sub><sup>2-</sup>, and Fe<sup>3+</sup>), which provide suboptimal conditions for CH<sub>4</sub> production.  
665 Vegetation removal may have further impeded CH<sub>4</sub> emissions by restricting plant-mediated pathways. Soil CH<sub>4</sub>  
666 fluxes did not show diurnal variability.

667 The ditch at the transect was a net source of both N<sub>2</sub>O and CO<sub>2</sub>, but at magnitudes 27 and 4 times lower than the  
668 soil GHG fluxes, respectively. It acted as a CH<sub>4</sub> source, comparable to other ditches in temperate fens. CH<sub>4</sub> was  
669 emitted mostly through diffusive emissions from the water surface, with occasional observations of ebullition.

670 This dataset provides a unique opportunity to test hypotheses regarding spatial and temporal patterns of GHG  
671 emissions and their drivers in peatlands. It supports the development of models that predict soil GHG fluxes in  
672 response to soil temperature and hydrology (WTD), aiding in the prediction of reliable budgets for locations

673 beyond Vejrumbro. We intend to publish this dataset to the research community so that experimentalists and  
674 modelers can use it to explore basic hydrological and thermal regulation of GHG fluxes and develop predictive  
675 models for spatiotemporal variability.

676 **Competing interests**

677 The authors declare that they have no conflict of interest.

678 **Author contributions**

679 JRC, PEL and KSL designed the experiment and carried them out. ASN performed flux calculation and quality  
680 checking. RJP and PEL installed the equipment for groundwater measurements. All authors contributed to  
681 writing of this manuscript.

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684 5229-0002b) projects hosted by University of Copenhagen and Aarhus University. ReWet is part of the Danish  
685 roadmap for research infrastructure funded by The Danish Agency for Science and Higher Education. INSURE  
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