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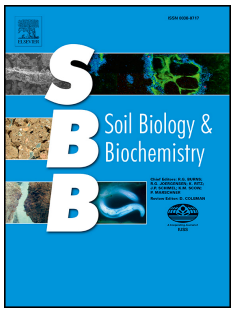

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Efficacy of mitigation measures for reducing greenhouse gas emissions from intensively cultivated peatlands

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ABSTRACT

Drained and cultivated fen peats represent some of the world's most productive soils, however, they are susceptible to degradation and typically exhibit high rates of greenhouse gas (GHG) emission. We hypothesised that GHG losses from these soils could be reduced by manipulating water table depth, tillage regime, crop residue application or horticultural fleece cover. Using intact soil columns from a horticultural peatland, emissions of CO_2, N_2O and CH_4 were monitored over a six-month period, using a closed-chamber method. Concurrent measurements of soil properties allowed identification of the key controls on GHG emissions. Raising the water table to the soil surface provided the strongest reduction in global warming potential (GWP_{100}; 26 ± 6 kg CO_2-e ha^{-1} d^{-1}), compared to a free-draining control (81 ± 1 kg CO_2-e ha^{-1} d^{-1}), but this effect was partially negated by an emission pulse when the water table was subsequently lowered. The highest emissions occurred when the water table was maintained 15 cm below the surface (172 ± 12 kg CO_2-e ha^{-1} d^{-1}), as this stimulated N_2O loss. Placement of horticultural fleece over the soil surface during spring had no significant effect on GWP_{100}, but prolonged fleece application exacerbated GHG emissions. Leaving lettuce crop residues on the surface increased soil GWP_{100} (106 ± 4 kg CO_2-e ha^{-1} d^{-1}) in comparison to when residues were incorporated into the soil (85 ± 4 kg CO_2-e ha^{-1} d^{-1}), however, there was no evidence that this promoted positive priming of native soil organic matter (SOM). For maximum abatement potential, mitigation measures should be applied during the growing season, when GHG emissions are greatest. Our results also suggest that introduction of zero- or minimum-till practices may not reduce GHG emissions. Maintaining a high water table was the only option that reliably reduced GHG emissions, however, this option is impractical to implement within current horticultural systems. We conclude that alternative strategies or a major change in land use (e.g., conversion from horticulture/arable to wetland) should be explored as a means of preserving these soils for future generations.

Keywords: Carbon cycling; Food security; Greenhouse gases; Histosol; Sustainable cropping

1. Introduction

Approximately 14-20% of peatlands globally are used for agriculture and when drained and cultivated they represent some of the world's most productive agricultural soils (IPS, 2008). Their management is highly problematic, however, due to the potential for soil loss, either from wind or water erosion or from microbial mineralisation of the peat substrate (Dawson and Smith, 2007). Whilst microbial activity results in the release of nutrients previously locked up in soil organic matter (SOM), thereby enhancing crop productivity, it also progressively diminishes the resource base (Cannell et al., 1999). There is therefore a clear ecosystem services trade-off between (1) preserving (and enhancing) peat carbon (C) storage for climate change mitigation, maintaining high biodiversity habitats, and improving water quality, and (2) using this resource to promote food security.

In many temperate and tropical countries, agricultural peatland emissions dominate national emissions of greenhouse gas (GHGs) from peat sources (IPS, 2008). For example, it has been estimated that 39% of English deep fen peats are currently under intensive cultivation and classed as being at risk from severe soil loss (Natural England, 2010). Within these sites, the depth of soil has been declining by 0.27-3.09 cm y^{-1} since the onset of drainage and cultivation in 1850 (Richardson and Smith, 1977; Hutchinson, 1980; Dawson et al., 2010). It has been estimated that 35-100% of drained Histosol loss may be attributable to microbially mediated CO_2 production (Leifeld et al., 2011). The small net consumption of CH_4 in these soils does little to offset CO_2 loss, whilst N_2O emissions can be substantial, forming approximately one third to one half of the total GHG budget (Taft et al., 2017). Mitigating GHG emissions from these soils is therefore a priority, especially as this could

substantially reduce the agricultural C footprint in some countries (UK Parliament, 2008; Kløve et al., 2017).

Agricultural soil GHG emissions are influenced by a large number of interacting factors, including those associated with soil (e.g., porosity, labile C), climate (rainfall, temperature), and vegetation (growth rate, rooting depth), which in turn are driven by agricultural management strategy (Li, 2007). Modifying a single factor may simultaneously increase emissions of one GHG and result in the reduction of another (Smith et al., 2008). Therefore, mitigation studies should consider the overall effect of a measure on the total emissions of CO_2, CH_4 and N_2O, rather than on a single GHG, as in some previous studies (Dalal et al., 2008; Henault et al., 2012; Musarika et al., 2017). This is particularly important where measures to reduce CO_2 emission increase the release of the more radiatively powerful CH_4 and N_2O, causing a disproportionately large increase in the overall global warming potential (GWP) of the system. Given the relationship between GHG efflux and soil organic C (SOC) loss (Dawson and Smith, 2007), and the importance of SOC to long-term soil sustainability, it is also useful for mitigation studies to include an estimate of the effects of treatments on SOC retention.

While many reviews on GHG mitigation in arable systems exist, few contain interventions specific to cultivated peatlands (e.g., Jauhiainen et al., 2016). Further, much of the evidence remains inconclusive. Our aim was to evaluate whether common management practices (i.e. tillage, manipulating water table depth, crop protection with fleece, and crop residue management) promoted or repressed GHG emissions and whether these could be used to promote SOC retention in cultivated peatlands. We hypothesised that tillage would promote soil aeration and net GHG loss, while conversely, raising the water table would reduce aeration and reduce net GHG loss. In addition, we hypothesized that fleece cover would increase soil temperature and moisture retention thereby promoting GHG emissions,

while addition of crop residues might reduce GHG emissions through negative priming of SOM.

2. Methods and materials

2.1. Study sites

Soils (Sapric Histosols; FAO, 2006) utilised in this study originate from a horticultural lowland peatland in East Anglia, UK (52º32' N, 0º29' E). The site has a mean annual rainfall of < 700 mm, a mean annual temperature of 10.2 °C (ranging from mean 4.2 °C in winter to 17.2 °C in summer), and mean annual sunshine hours of 1550 (UK MetOffice, 2014). The study area comprises drained lowland fen typified by flat topography, which is under intensive commercial-scale horticultural and arable production, growing primarily vegetables (including lettuces [*Lactuca sativa* L.], potatoes [*Solanum tuberosum* L.], leeks [*Allium porrum* L.], onions [*Allium cepa* L.], red beet [*Beta vulgaris* L.], and celery [*Apium graveolens* L.]), sometimes in rotation with cereals (primarily wheat [*Triticum aestivum* L.]). Soil was collected from a representative field (~70% SOM content; Taft et al., 2017), which had been under a typical rotation for the previous growing season. Table 1 shows the physical and chemical characteristics of the soils used in the experiments.

2.2. Field sampling

Intact soil cores were taken from a visually representative area (10 m^2) of a field to minimise any microsite variability caused by soil heterogeneity. A PVC pipe ($d_{internal}$ = 103 mm; h = 400 mm) with a chamfered base was slowly driven into the soil to give a final core depth of 300 mm with c. 100 mm remaining at the top of the core to act as chamber headspace when GHG sampling. After excavation, the cores were transported (10 °C) to the experimental site at Bangor University (53°13' N, 4°9' W), where they were laid out in a

randomised design with four blocks to allow for monitoring of background emissions of CO_2, CH_4 and N_2O prior to experimentation (no significant differences among cores were apparent; data not presented).

2.3. Preliminary soil and residue analysis

Five additional cores were taken from the field and a number of chemical and physical analyses performed before commencement of the experiment; the same analyses were conducted at the end of the experiment on all cores (Table 1). The cores were split into three layers (0-10, 10-20 and 20-30 cm depth) and analyses were performed on each layer. A Rhizon® suction sampler was inserted to 10 cm depth and a soil water sample obtained then stored at c. -20 °C to await analysis. Next, a soil sample was taken using a bulk density ring (h_{total} = 10 cm, V_{total} = 200 cm^3) for calculation of soil gravimetric moisture content and bulk density after oven drying (105 °C, 24 h). The remaining soil was homogenised and stored at 4°C prior to chemical analysis within 48 h. Soil samples extracts were performed in triplicate for each soil layer for the determination of available NO_3^- and NH_4^+ (5 g soil in 25 ml 0.5 M KCl), available P (5 g soil in 25 ml 0.5 M acetic acid), and available K (5 g soil in 25 ml 1 M NH_4Cl). Extracts were obtained by shaking (200 rev min^{-1}, 30 min), centrifugation (3,250 × *g*, 10 min), filtering through a Whatman 42 filter paper and storage at -20 °C to await analysis. Available soil NO_3^-, NH_4^+ and P were determined colorimetrically on a PowerWave XS microplate spectrophotometer (BioTek UK, Bedfordshire, UK) using the methods of Mulvaney (1996), Miranda et al. (2001), and Murphy and Riley (1962) respectively. Available K in the acetic acid extracts was determined with a Model 410 flame photometer (Sherwood Scientific Ltd., Cambridge, UK). The moisture content of residue samples was determined by oven drying (80 °C, 72 h), while total C and N was determined with a CHN2000 analyser (Leco Corp., St Joseph, MI, USA).

135

136 *2.4. Experimental treatments*

137 The cores were randomly assigned to six treatments as follows: (1) Control, (2) Water
138 table maintained at 15 cm below the surface (WT_{15}), (3) Water table maintained at the soil
139 surface (WT_0), (4) Soil surface covered with horticultural fleece (C_{fleece}), (5) Simulated tillage
140 (S_{till}), (6) Crop residues applied to the soil surface (CR_{surf}), and (7) Soil tilled and crop
141 residues incorporated into the soil (CR_{incorp}) (Table 2). Each core had mesh covering the base
142 and was placed in larger plastic container to allow accurate water table control
143 (Supplementary information Appendix A, Fig. A.1). Sand surrounded the outside of the core
144 to minimise thermal gradients and holes drilled in the side of the containers to allow drainage,
145 or maintenance of the water table in the WT_0 and WT_{15} treatments. The mesocosms were laid
146 out in a randomised block design with five replicates of each treatment, with blocks aligned
147 to the prevailing wind direction (SW-NE) to account for differences in sheltering and
148 evapotranspiration. Water tables were established by filling the containers with artificial
149 rainwater solution (containing 96 µmol L^{-1} NaCl, 10 µmol L^{-1} K_2SO_4, 5 µmol L^{-1}
150 $CaCl_2.2H_2O$, 6 µmol L^{-1} $MgCl_2.6H_2O$, 15 µmol L^{-1} NH_4NO_3, and 0.1 µmol L^{-1} KH_2PO_4,
151 reflecting average Welsh rainwater composition; Stevens et al., 1997) until the excess ran out
152 of the lateral drainage holes. Subsequently, water table height was maintained with natural or
153 artificial rain water. For the C_{fleece} treatment, white horticultural, unwoven polypropylene
154 fleece was secured over the top of the core headspace using plastic-coated wire. Horticultural
155 fleece can be used for a variety of purposes including crop protection from frosts or pests and
156 diseases, and soil warming and protection from wind or water erosion (e.g., Olle and Bender,
157 2010). At our study site, it is used primarily for soil warming and crop protection against
158 frosts, to facilitate the production of early crops. Cultivation treatments were based on the
159 typical ploughing depth at the field site (c. 30-35 cm), and were implemented by removing

the whole volume of soil from the core, mixing in crop residues where appropriate, and packing loosely back into the core. Soil residue treatments involved the addition of Iceberg lettuce (*Lactuca sativa* L.) residues (c. 5 × 5 cm pieces) to the soil based on rates measured in the field post-harvest (52% of the total crop; 0.9 t C ha^{-1}). The residues were pressed into the soil surface to simulate post-harvest tractor traffic.

Mesocosm measurements were made for seven consecutive days following treatment application (May and Aug. 2013), then twice per week for two weeks, then weekly until the end of each experimental period (Aug. and Nov 2013). The experiment had two phases for the water table treatments (WT_0 and WT_{15}): Phase I involved maintaining the water table at the target depth for 3 months (i.e. 0 or -15 cm), while in Phase II the water table was lowered (by drilling holes in the base of the container) to match the control treatment (i.e. -30 cm). After 6 months, observable differences in GHG emissions among the water table treatments were largely negligible. Consequently, the cores were dismantled, split into 10 cm depth fractions and analysed as outlined in Section 2.3.

2.5. Greenhouse gas monitoring

Closed, non-vented static chambers were used to measure emissions of CH_4 and N_2O. These consisted of white opaque polypropylene cylindrical chambers (headspace 0.66 dm^3) with a rubber septum sampling port in the lid (Supplementary information Appendix A, Fig. A.1). Each chamber was attached immediately before taking the first gas sample ($t = t_0$), giving a final average enclosed headspace of 1.72 dm^3. Subsequent samples were taken at approximately 10 min intervals ($t = t_{10}$, t_{20} and t_{30}). Gas sampling and storage procedures and materials followed those described in Taft et al. (2017). Sample analysis was undertaken with a gas chromatograph (Varian 450-GC, Bruker UK Ltd., Coventry, UK), equipped with a flame ionisation detector (FID, operated at 120-125 °C) and electron capture detector (ECD,

operated at 300 °C), and attached to a QUMA QHSS1-40 Headspace Autosampler (QUMA Electronik & Analytik GmbH, Wuppertal, Germany), which injected 2 ml of sample into the GC. We measured CO_2 emissions from the cores with an EGM-4 infra-red gas analyser (PP Systems, Hitchin, UK) equipped with an SRC-1 soil respiration chamber.

2.6. Soil water, climate and redox measurements

Soil temperature was measured with a Checktemp1® probe (±0.3 °C; Hanna Instruments Ltd, Leighton Buzzard, UK) over a 0-10 cm depth. Soil solutions were recovered non-destructively throughout the experiment using Rhizon® soil water samplers (Rhizosphere Research Products, Wageningen, The Netherlands) inserted into the topsoil (0-10 cm depth). Soil solutions were stored at -20 °C to await analysis. During experimental Phase II, soil surface (1-2 cm depth) redox potential (E_h) was measured using an Eijkelkamp BNC glass Platinum electrode with an Ag/AgCl reference electrode and 3 M KCl electrolyte (Eijkelkamp Agrisearch Equipment, Giesbeek, The Netherlands) following Eijkelkamp (2009). Sampling ports in the side of the core (at 10, 20 and 30 cm below the soil surface) allowed additional temperature and E_h measurements to be made. Rainwater samples were collected periodically through the experiment and analysed for soluble N. Meteorological data (rainfall, air temperature) were obtained from the local Met. Office monitoring station.

2.7. Statistical analysis

Statistical analyses were performed using SPSS v. 20 (IBM Corp., Armonk, NY), with significance being accepted at $p \leq 0.05$ unless otherwise stated. GHG flux calculation and data cleaning procedures were identical to those of Taft et al. (2017). Cumulative flux estimates were converted to 100-year global warming potential (GWP_{100}) CO_2 equivalents (CO_2-e) according to IPCC (2006). Cumulative fluxes of CO_2, N_2O, CH_4 and total GWP_{100}

for each treatment were compared using ANOVA, independent t-test, Kruskal-Wallis or Kolmogorov-Smirnov Z tests as appropriate. Post-Hoc tests were conducted to determine significantly different treatments using Tukey's HSD, Gambrell-Howell, or Kolmogorov-Smirnov Z statistics (with Bonferroni correction for multiple comparisons) as appropriate. Relationships among individual GHGs, temperature, rainfall, and soil N concentrations were explored using Kendall's tau statistic (τ).

All statistical analyses were performed separately on the water table group of treatments (Control vs. WT_0 vs. WT_{15}), the fleece treatment (Control vs. C_{fleece}), and the cultivation and residue group of treatments (Control vs. S_{till} vs. CR_{surf} vs. CR_{incorp}). Normality was tested using the Shapiro-Wilk test (Field, 2005), and non-normal data were log_{10}-transformed or square-root transformed; where transformation was ineffective, or where heterogeneity of variances was observed (Levene's or Welch's test statistic), appropriate non-parametric tests were used to compare medians of those data groups. Soil physical and chemical characteristics for each soil depth layer were compared using ANOVA or the independent t-test, or Kruskall-Wallis or Kolmogorov-Smirnov Z tests for data deviating greatly from normality or homogeneity of variances. Significant effects of treatment and time (each treatment including the control, compared to the baseline) were tested.

3. Results

3.1. Climate and changes in soil quality

Analysis of the soil at the end of the experiment showed that some properties had changed slightly over the 6-month period (Table 1). In most cases, however, the effect of treatment was small. The mean air temperature for Phase I and II of the experiment were 15.4 and 13.2 °C, respectively (Fig. 1a-b). During the same period, the cumulative rainfall was 191 and 229 mm, respectively.

235

236 *3.2. Effect of water table manipulation on GHG emissions and soil chemistry*

237 Soil respiration responded rapidly to raising of the water table, falling close to zero
238 within 5 d of water table raising in the WT_0 treatment, and remaining lower (11 ± 1.4 mg
239 CO_2-C m^{-2} h^{-1}) than mean fluxes from the control and WT_{15} treatments (76 ± 3.6 mg CO_2-C
240 m^{-2} h^{-1} and 78 ± 3.9 mg CO_2-C m^{-2} h^{-1} respectively) for the remainder of the wetted period
241 (Fig. 1c-d). Immediately after draining, there was a peak in CO_2 emissions from both the
242 WT_0 and WT_{15} treatments, however, these returned to values close to the control after a
243 further 44 d.

244 During the wetted period, mean N_2O emissions ranged from 5.0 ± 6.0 to 4453 ± 577
245 µg N_2O-N m^{-2} h^{-1} across all treatments (Fig. 1e-f). A substantial peak (4453 ± 577 µg N_2O-N
246 m^{-2} h^{-1}) was observed from the WT_{15} treatment after 14 d and emissions in this treatment
247 remained consistently higher than the WT_0 and control treatments during the first six weeks.
248 Over this period, N_2O emissions were very similar in the control and WT_0 treatments.
249 Drainage resulted in a short-lived rise (c. 14 d) in N_2O flux which was most pronounced in
250 the WT_{15} treatment immediately following draining (1506 ± 499 µg N_2O-N m^{-2} h^{-1}).
251 Emissions in the WT_0 treatment exhibited a similar but smaller response 3 d after draining
252 (699 ± 277 µg N_2O-N m^{-2} h^{-1}). Fluxes of CH_4 remained low throughout the experiment (Fig.
253 1g-h).

254 Cumulative GHG emissions were significantly influenced by water table depth (Table
255 3). In the initial wetted phase (Phase I), a significant decline in CO_2 emissions was apparent
256 as the water table was raised closer to the soil surface. However, a significant difference was
257 only observed between the control and WT_0 treatments ($p < 0.01$), although the difference
258 between the WT_{15} and WT_0 treatments was almost significant ($p = 0.08$). Cumulative N_2O
259 emission was significantly influenced by water table depth ($p < 0.001$), with the mean WT_{15}

cumulative flux being significantly higher than both the control and WT_0 treatments (both $p < 0.001$). No significant treatment effects were observed for cumulative CH_4 emissions. Cumulative GWP_{100} for water table treatments was significantly different among groups ($p < 0.001$); with a highly significant increase in the order WT_0 < control < WT_{15} (all $p < 0.001$).

In the drained period (Phase II), significant differences were recorded for median CO_2 emissions among water table groups ($p < 0.05$; Table 3). However, no significant differences were found among the three water table treatments for cumulative N_2O, cumulative CH_4, or GWP_{100}.

Over the entire experiment (Phase I and Phase II), CO_2 and N_2O emissions were highly influenced by water table depth (both $p < 0.001$; Table 3). There was a highly significant decline in soil respiration between WT_{15} and WT_0 treatments ($p < 0.001$), while no difference was noted between the control and WT_{15} treatments. Mean N_2O emissions were significantly higher from the WT_{15} treatment compared to the control and WT_0 treatments (both $p < 0.001$). There was no effect of water table depth on cumulative CH_4 emissions. Water table treatment had a highly significant effect on GWP_{100} ($p < 0.001$; Table 3), and all treatments were significantly different to each other: WT_0 was lower than both the control and WT_{15} treatments ($p < 0.05$ and $p < 0.001$ respectively), and the control was lower than WT_{15} ($p < 0.001$).

Mean NO_3^- concentrations were substantially lower in the WT_0 than in the control and WT_{15} treatments, both of which were similar to each other (Fig. 1i-j). Dissolved NH_4^+ remained consistently low at all measurement times (Fig. 1k-l).

Redox (E_h) values in the upper soil layer was similar across all treatments remaining > 400 mV for most of the monitoring period (Fig. 2a). On the day on which the cores were drained, the E_h was notably lower in the 10 cm soil layer WT_0 treatment (369 ± 36 mV) than in the WT_{15} and control treatments (480 ± 11 and 487 ± 10 mV, respectively; Fig. 2b). Upon

draining, an immediate and marked drop in E_h was observed in the 20 cm soil layer in both the WT_0 (315 ± 46 mV) and WT_{15} (422 ± 42 mV) cores, compared with the control (490 ± 8 mV, Fig. 2c). Four days after draining, however, there were no observable differences among treatments. Redox potentials in the 30 cm soil layer were the most responsive to water table treatments (Fig. 2d). Both WT_0 and WT_{15} treatments showed substantially lower mean E_h values (218 ± 17 mV and 227 ± 19 mV, respectively) compared with the control cores (341 ± 24 mV) for the first 38 d. By day 62, WT_{15} redox values had returned to that of the control values, whereas the WT_0 E_h took 85 d to recover to levels seen in the control.

3.3. Effect of fleece application on GHG emissions and soil chemistry

Soil respiration from the C_{fleece} and control cores followed a similar pattern throughout the experiment although the fluxes were generally higher in the C_{fleece} treatment (Fig. 3b). The peak flux in the C_{fleece} treatment (232 ± 61 mg CO_2-C m^{-2} h^{-1}) occurred on day 52, and was almost double that of the control emission (132 ± 6.6 mg CO_2-C m^{-2} h^{-1}). Mean N_2O emissions were similar from the C_{fleece} and control treatments throughout most of the experimental period (Fig. 3c). Maximum N_2O emission from the C_{fleece} treatment (542 ± 182 µg N_2O-N m^{-2} h^{-1}) occurred 7 d after fleece application, returning to control levels after 14 d. Emissions of CH_4 were higher than in the control treatment, however, these fluxes were still very low (Fig. 3d). Mean C_{fleece} NO_3-N and NH_4-N concentrations were very similar to the control treatment on all sampling dates (Figs. 3e-f).

Overall, cores with fleece had significantly higher mean cumulative CO_2 emissions ($p < 0.05$; Table 3) while total N_2O emission was also higher than the control ($p = 0.06$). The fleece treatment had a significantly greater cumulative GWP_{100} emission than the control ($p < 0.01$).

3.4. Effect of cultivation tillage on GHG emissions and soil chemistry

Mean CO_2 fluxes in the tilled soil were very similar to the control on most sampling dates, ranging from 26 ± 4.7 to 135 ± 5.2 mg CO_2-C m^{-2} h^{-1} (Fig. 4d). A marked peak in CO_2 release was observed immediately after simulated ploughing, however, this was of short duration. For a few days during the experiment, S_{till} CO_2 emissions were lower than in the control cores. Overall, mean fluxes of N_2O and CH_4 were similar to the control (Figs. 4g and 4j). Ploughing had no significant effect when compared to undisturbed soil on cumulative individual GHG emissions or overall GWP_{100} (Table 3). We observed no consistent effect of tillage on soluble N concentrations relative to the control throughout the experiment.

3.5. Effect of residue incorporation on GHG emissions and soil chemistry

Both residue treatments showed a marked increase in soil respiration immediately following surface application or incorporation into the soil, with elevated levels persisting for three weeks after application (Fig. 4e-f). The response was generally lower when residues were incorporated into the soil. Emissions of N_2O responded positively to residue application, but with a slower response (5-6 d), and over a longer period (37 d), compared to the control treatment (Fig. 4h-i). In the CR_{incorp} treatment, both soil respiration and N_2O emissions were lower than from the control towards the end of the experimental period. No marked effect of residue treatment was observed for CH_4 emissions or soil solution N relative to the control throughout the experiment (Figs. 4k-l, 4n-o and 4q-p).

The surface-applied residue treatment yielded a significantly higher mean cumulative soil respiration ($p < 0.01$), mean cumulative N_2O emission ($p < 0.05$), and median cumulative GWP_{100} ($p < 0.01$) than the control treatment (Table 3). In contrast, no significant differences were apparent in any of the individual cumulative GHG emissions or overall GWP_{100} between the control and residue incorporation treatment (Table 3). Compared to the surface-

residue application treatment, cumulative emissions from the incorporated residue treatment were only significantly lower for CO_2 ($p < 0.05$).

3.6. Effect of soil and weather conditions on GHG emissions

Redox potential at depth was significantly correlated with CO_2 ($p < 0.05$) and N_2O ($p < 0.05$) emissions, but not CH_4 release ($p > 0.05$) (Table 4). At 20 cm below the soil surface, E_h was positively associated with CO_2 emission in the control and WT_{15} treatments, explaining 3% of the variability in soil respiration (τ = -0.176 to -0.179). At 30 cm depth, E_h was negatively associated with CO_2 emission in the WT_0 treatment, and N_2O emission in the WT_0 and WT_{15} treatments, explaining 3% of CO_2 emission variability and 3-6% of N_2O emission variability (τ = -0.174 to -0.254).

Soil temperature, mean daily air temperature, and measured air temperature were positive, highly significant predictors of soil respiration within most treatments, accounting for between 12-31%, 3-38%, and 5-18% of fluxes respectively (τ = 0.341 to 0.559, $p < 0.05$ to < 0.01; Table 4). Temperature variables were less suitable for predicting N_2O emissions, although some highly significant correlations were still apparent. Soil temperature, mean daily air temperature, and measured air temperature at the time of sampling predicted 2-10%, 3-7%, and 3-12% of N_2O emissions respectively (τ = 0.147 to 0.313, $p < 0.05$ to < 0.001).

Daily and 5-day rainfall (cumulative rainfall from the day of measurement and the four preceding days) were negative highly significant predictors of CO_2 emissions for most of the treatments (τ = -0.112 to -0.460; $p < 0.05$ to < 0.001), while daily rainfall was positively significantly correlated with surface-applied residue CO_2 efflux (τ = 0.180, $p < 0.05$; Table 4). Daily rainfall explained 1-8% and 5-day rainfall explained 2-21% of soil respiration. Emissions of N_2O and daily rainfall were highly significantly negatively correlated in all but the drained control treatment, accounting for 2-34% of emissions (τ = -0.136 to -0.579, $p <$

0.05 to < 0.001). Cumulative 5-day rainfall was a significant predictor of N_2O emission in the WT_{15} treatment only, explaining 4-7% of N_2O flux (τ = -0.199 to -0.260; $p < 0.001$).

Dissolved N was a significant predictor of soil respiration in most treatments. Emissions of N_2O and NO_3^- concentration were significantly positively correlated in the control (Phase I) and WT_{15} (Phase II, Phase I + II) treatments, with NO_3^- accounting for 3-13% of variability in N_2O emission (τ = 0.185 to 0.358, $p < 0.05$ to < 0.001). Concentrations of NH_4^+ were positively associated with soil respiration in the control (Phase I), WT_{15} (Phase I, Phase I + II), and S_{till} treatments (2-7% of variability, τ = 0.135 to 0.255, $p < 0.05$ to < 0.01), but negatively associated with soil respiration in the control (Phase II) treatment (3% of variability, τ = -0.187, $p < 0.05$). A significant correlation between dissolved NH_4^+ concentration and N_2O emission was found in only the surface-applied residue treatment (9% of variability, τ = -0.292, $p < 0.01$), and with CH_4 emissions in the fleece treatment (6% of variability, τ = -0.239, $p < 0.01$; Table 4).

4. Discussion

4.1. Effect of water table manipulation on GHG emissions

In agreement with previous studies of fen and blanket peats under a range of land uses, raising the water table in this study reduced CO_2 emissions, moreover, the magnitude of the reduction proved highly sensitive to water table depth (Dinsmore et al., 2009; Freeman et al., 1993; Lloyd, 2006; Kechavarzi et al., 2007). Maintaining the water table at the surface also reduced N_2O emissions. We ascribe this to a reduction in the nitrification rate and NO_3^- production and the complete denitrification of any NO_3^- present to N_2 (Velthof and Oenema, 1997). Lowering the water table to 15 cm, however, resulted in greatly elevated N_2O emissions. This concurs with findings from Freeman et al. (1993) who also reported N_2O emission to be inversely correlated with water table depth. Our highest rate of N_2O emission

in the water table treatments (4.5 mg N_2O-N m^{-2} h^{-1}) was two orders of magnitude higher than emissions from semi-natural peatland mesocosms observed by Freeman et al. (1993) and Dinsmore et al. (2009), but similar to studies of arable peatlands (Flessa et al., 1998; Taft et al., 2017; Weslien et al., 2012). A large initial peak in N_2O emissions was observed in the WT_{15} treatment after raising the water table, while only a small pulse was seen in the WT_0 treatment. Conversely, the WT_0 treatment released most N_2O after draining, while the N_2O pulse from the WT_{15} treatment was smaller. These relatively rapid, short-lived, strong responses to wetting and draining events in peat soils are common, with their magnitude typically limited by soil moisture and soluble N (Li et al., 1992). Overall, there was no marked effect of water table treatment on CH_4 production over the wetted or drained experimental periods, contrary to the general trend of water table raising increasing emissions (Bussell et al., 2010). Strictly anaerobic conditions required for substantial CH_4 emissions, however, may take a long time to develop (>1 y; Oomes et al., 1997), and in infrequently flooded soils are typically found at lower profile depths than those sampled in this study (Mitsch and Gosselink, 2000). The low rates of CH_4 release could also be due to a lack of methanogens, or the abundance of alternative electron acceptors and/or an efficient population of methanotrophs in the topsoil. This is supported by measured redox values which largely fell within the range associated with CO_2 production and CH_4 consumption (400 to 500 mV) and N_2O production (200 to 500 mV), but not for CH_4 production (-100 to -200 mV; Le Mer and Roger, 2001; Li, 2007; Mitsch and Gosselink, 2000).

This study simulated raising the water table during late spring followed by draining in late summer, mimicking the water management regime commonly employed by farms in the study area to enable sub-surface irrigation and minimise peat loss via wind erosion (Dawson et al., 2010). In practice, raising the water table to within 15 cm of the soil surface would not be implemented while a crop was in place, as it would likely result in high crop mortality and

410 be unsuitable for field traffic. Instead, this intervention would probably be implemented
411 between summer crops, possibly over quite short fallow periods. The relative efficacy of
412 flooding as a GHG mitigation strategy may be enhanced by additional impacts such as weed
413 growth even during relatively short fallow periods; which could further reduce net GWP_{100}
414 through elevated net primary productivity and plant removal of NO_3^- (e.g., Kløve et al.,
415 2017). Conversely, both the presence of weeds and labile organic matter input from post-
416 harvest crop residues could result in substantial emissions of N_2O and CH_4 (Le Mer and
417 Roger, 2001). The net effect of vegetation therefore merits further investigation.

418 Maintaining the water table at the correct level and ensuring it drains freely post-
419 flooding could be challenging. Kechavarzi et al. (2007) suggest that close spacing of sub-
420 surface drainage pipes (≤10 m) would be required to maintain a consistent water table level in
421 a sub-irrigated field. Some fields are not equipped with closely spaced drainage pipes, and
422 not all peat soils are sub-irrigated. Fluctuation of the water level between 0-15 cm of the soil
423 surface, either through poor water level maintenance or slow drainage post-flooding, is likely
424 to result in large pulses of GHGs, as was observed in the WT_{15} treatment, entirely negating
425 the beneficial effect of flooding. This effect may be minimised if draining is undertaken in
426 cooler weather. Further, flooding poses a number of difficulties both agronomically and in
427 the context of the wider landscape. Implementation would require careful timing so that after
428 flooding, soil had time to dry sufficiently before subsequent in-field machinery operations.
429 Yields of subsequent crops could be reduced after flooding, or the costs of mineral fertiliser
430 increased: our results strongly imply that much of the soil nitrate was leached from the soil
431 columns during draining. In terms of wider landscape effects, leaching of nitrate into
432 watercourses poses a severe pollution risk, with associated costs for the grower. Further, if
433 flooding were to be implemented on a widespread scale, regulation would be required to

ensure that it did not adversely impact on flood risk and response across the region, which would be challenging across areas of flat topography.

4.2. Effect of fleece application on GHG emissions

This study found that fleece application significantly increased GWP_{100}, CO_2 release and N_2O emissions from soil. Fleece application is known to stabilise variations in soil temperature and to reduce soil moisture loss (Hamouz et al., 2006; 2005; Siwek et al., 2013; 2012). In this study, temperature was the strongest predictor of soil respiration, showing a significant positive correlation in the fleece-enclosed cores. This is consistent with other studies on the effect of temperature on peat soil respiration (Estop-Aragonés and Blodau, 2012; Maljanen et al., 2002). Soil temperature has also been shown to positively correlate with N_2O emissions (Maljanen et al., 2002), although in this study the relationship was not strong.

The greatest emissions from the fleece treatment were observed when the air temperature was highest. In practice, fleece would usually only be applied to early crops, to minimise the risk of frost damage and encourage early crop development (Hamouz et al., 2006). However, the presence of fleece did increase net emissions under cooler as well as warmer temperatures, albeit at a reduced rate. It is important therefore, to restrict fleece application to as short a period as possible during cooler weather, as is common under current practice (G's Fresh, *pers. comm.*; HDC, 2006).

As with the water table treatments, the effect of prolonged fleece application in the presence of a crop should be investigated at the field scale, to compare crop growth and associated net ecosystem exchange between fleece and control treatments, as this may further reduce the difference in emissions. It would also be of interest to consider the effect on net emissions when fleece is applied over recently-fertilised peat, since the results suggest that

459 N_2O emissions may substantially increase when fertilised soil is subjected to the warmer soil
460 temperatures associated with fleece application.
461
462 *4.3. Effect of tillage on GHG emissions*
463 Simulated ploughing resulted in an immediate, small and short-lived peak in soil
464 respiration but a negligible response of N_2O. Ploughing-induced peaks in CO_2 emission from
465 cultivated Histosols have been noted by Elder and Lal (2008) and Reicosky et al. (2008),
466 although the response found in our study was several-fold lower than that of Elder and Lal
467 (2008) (625 mg CO_2-C m^{-2} h^{-1}). Mean emissions from a bare-tilled peat measured by
468 Maljanen et al. (2002) (300 mg CO_2-C m^{-2} h^{-1}), were also higher than the peak emission of
469 135 mg CO_2-C m^{-2} h^{-1} recorded in this study. Production of N_2O was not stimulated by a
470 ploughing event. This contrasts with the findings of Elder and Lal (2008), however Maljanen
471 et al. (2002) and Weslien et al. (2012) also reported negligible effects of ploughing on N_2O
472 emissions. It is probable that the considerably lower peak of N_2O emissions observed here
473 compared with those of Elder and Lal (2008) are a result of suboptimal soil moisture
474 conditions inhibiting N_2O production, owing to the comparatively good drainage and lower
475 bulk density of our tilled cores (Dalal et al., 2003). Our results are in strong contrast to the
476 assertion that cultivation results in a large efflux of both CO_2 and N_2O (Dawson and Smith,
477 2007; Kasimir-Klemedtsson et al., 1997). This suggests that adoption of minimum or zero
478 tillage practices may not help preserve soil C on sites with a long history of cultivation.
479
480 *4.4. Effect of residue application on GHG emissions*
481 The pattern and magnitude of CO_2 and N_2O fluxes observed after residue application
482 may be attributed in part to the characteristics and amount of, and mechanism by which, the
483 residues were added. In a study comparing emissions from soils amended with crop residues

484 with differing compositions, Velthof et al. (2002) observed a rapid response and pronounced
485 peak in N_2O and CO_2 emissions from crops which, similarly to this study, had a low C/N
486 ratio (c. 10-20) and high moisture content (>80%). Other studies support the theory that the
487 application of crop residues with low C/N ratios tends to induce greater CO_2 and N_2O
488 emissions (Loecke and Robertson, 2009), as well as biodegrading faster (Henderson et al.,
489 2010). The emissions observed in our study were lower than expected, and may be explained
490 by the relatively low total quantity of residue C and N added to each core (746 mg C $core^{-1}$,
491 73 mg N $core^{-1}$) in comparison with other studies (e.g., Velthof et al., 2002).

492 Residue application increased cumulative net emissions. This could be attributable to
493 the positive priming of soil microbial activity and loss of native SOM (Kuzyakov et al., 2000;
494 Kuzyakov, 2010). Although we cannot discount this mechanism, our data does not support it
495 for the following reasons: (1) Compared to the control, the extra loss of CO_2 was only
496 equivalent to 0.32 t C ha^{-1} (CR_{surf}) and 0.01 t C ha^{-1} (CR_{incorp}), i.e. considerably less than the
497 quantity of residue-C added to the cores (0.90 t C ha^{-1}). This suggests that negative priming
498 may actually be occurring, particularly when residues are incorporated into the soil, although
499 further work would be needed to confirm this; (2) The equivalent of 88 kg N was added to the
500 residue cores, but only 2.1 and 0.7 kg N_2O-N ha^{-1} more than the control was lost in the
501 surface applied and incorporation treatments respectively. It should be noted, however, that
502 we cannot account for denitrification losses of N_2; (3) We had expected that if positive
503 priming was occurring the effects would be greater when the residues were incorporated into
504 the soil; and (4) Recent research suggests that much of the CO_2 released from plant residues
505 applied to soil originates from the residue itself (e.g., cell autolysis) rather than from a soil
506 microbial-induced breakdown of the residues (Marella et al., 2017).

507 While residue incorporation resulted in lower emissions relative to surface application
508 in our study, our experiment was limited to a single crop (lettuce). Characteristics such as

crop dry matter content, C/N ratio, availability of labile C and N, and the total quantity of residue applied and its particle size distribution across or within the soil can significantly impact net emissions associated with residue application of different crops (Loecke and Robertson, 2009; Velthof et al., 2002; Webb et al., 2014). Further research might therefore focus on relative emissions from surface applied and incorporated residues of a range of crops at the field scale, and at a variety of points in the growing season (to account for the common practice of multiple cropping on these soils; Taft et al., 2017).

5. Conclusions and implications

The results of this study suggest that the relative efficacy of potential GHG mitigation options will be strongly influenced by the weather and soil conditions at the time of implementation, and hold the greatest potential efficacy if applied during the main growing season when GHG emissions are greatest. Net GHG emissions from the horticultural peat soils in this study proved sensitive to water table depth, with flooding to the soil surface being highly effective in reducing GHG emissions. However, avoiding a shallow water table is paramount in minimising emissions. Our study suggests that horticultural fleece should be used for the shortest possible period, and in cool weather only. Contrary to expectation, tillage did not significantly increase net GHG emissions. We recommend that tillage and harvesting operations should be conducted during cooler or damper weather to minimise the small peak in emissions. The impacts of lettuce residue treatment were somewhat inconclusive, with residue incorporation reducing net emissions compared to surface application, but only significantly for CO_2 emissions and not for overall GWP_{100}.

The practical implications of implementation are dependent on synchronising measures with on-going management operations. Precise management of water table height is highly restricted from a practical perspective, and cannot be expected across large-scale

areas, as this type of mitigation risks creating within-field emission hotspots. Conducting tillage operations during cooler weather is likely to be somewhat impractical in relation to harvesting operations due to economic pressures. In contrast, restricting horticultural fleece use to the start of the season should pose few practical difficulties as the practice already aligns with current management. Our results suggest that no one single mitigation measure may be effective in reducing the rate of soil loss in cultivated peatlands. This has important implications for the practicalities of co-implementing individual mitigation strategies, or in considering more radical changes of land use and management in future.

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Table 1. Major soil characteristics in the soil cores sampled at the start and end of the experimental period and for the control, water table at -15 cm below soil surface (WT_{15}), water table at soil surface (WT_0), fleece cover (C_{fleece}), simulated till (S_{till}), surface applied crop residue (CR_{surf}), and incorporated crop residue (CR_{incorp}) treatments. Values are presented as mean ± SEM. Significant differences between initial core values and post-experiment values for each treatment (within each soil layer) are marked with * for $p < 0.05$, ** for $p < 0.01$, *** for $p < 0.001$, and † for non-parametric (Kolmogorov-Smirnov Z statistic, Bonferroni corrected).

Treatment	Depth (cm)	Soil moisture (% DW)	Bulk density (g cm^{-3})	pH (H_2O) [a]	EC (µS cm^{-1}) [s]	Available K (g K kg^{-1})	Available P (g P kg^{-1})	Available NO_3^- (g N kg^{-1})	Available NH_4^+ (g N kg^{-1})
Initial	0-10	152 ± 1	0.68 ± 0.01	6.2 ± 0.08	598 ± 50	0.96 ± 0.21	0.39 ± 0.01	0.15 ± 0.016	0.05 ± 0.024
	10-20	156 ± 2	0.76 ± 0.02	6.2 ± 0.06	552 ± 49	0.63 ± 0.11	0.38 ± 0.01	0.15 ± 0.033	0.04 ± 0.008
	20-30	163 ± 5	0.75 ± 0.02	6.3 ± 0.06	401 ± 24	0.56 ± 0.11	0.35 ± 0.02	0.13 ± 0.033	0.03 ± 0.001
Post-experiment									
Control	0-10	164 ± 1†	0.73 ± 0.01*	6.7 ± 0.04†	161 ± 13	0.54 ± 0.08	0.27 ± 0.02†	0.01 ± 0.001†	<0.01
	10-20	168 ± 2**	0.77 ± 0.01	6.7 ± 0.06***	166 ± 8	0.51 ± 0.19	0.27 ± 0.01**	0.03 ± 0.004†	<0.01
	20-30	180 ± 2	0.75 ± 0.01	6.7 ± 0.04*	220 ± 9***	0.58 ± 0.15	0.21 ± 0.04	0.06 ± 0.008	<0.01
WT_{15}	0-10	170 ± 1†	0.74 ± 0.01**	6.7 ± 0.04†	136 ± 3	0.63 ± 0.08	0.29 ± 0.02†	0.01 ± 0.001†	<0.01
	10-20	171 ± 2***	0.78 ± 0.01	6.7 ± 0.03***	160 ± 6	0.50 ± 0.13	0.31 ± 0.02	0.02 ± 0.001†	<0.01
	20-30	175 ± 6	0.75 ± 0.01	6.7 ± 0.03*	223 ± 11***	0.44 ± 0.10	0.26 ± 0.04	0.03 ± 0.006	<0.01
WT_0	0-10	172 ± 1†	0.74 ± 0.01**	6.7 ± 0.03†	159 ± 8	0.61 ± 0.16	0.27 ± 0.01†	0.01 ± 0.001†	<0.01
	10-20	169 ± 3**	0.78 ± 0.02	6.8 ± 0.07***	176 ± 17	0.62 ± 0.16	0.27 ± 0.01**	0.02 ± 0.001†	<0.01
	20-30	174 ± 5	0.77 ± 0.01	6.7 ± 0.06**	196 ± 16***	0.49 ± 0.17	0.33 ± 0.04	0.02 ± 0.003†	<0.01
C_{fleece}	0-10	161 ± 2†	0.73 ± 0.01	6.6 ± 0.05†	154 ± 9†	0.42 ± 0.07	0.35 ± 0.03	0.01 ± 0.001	<0.01
	10-20	166 ± 3*	0.76 ± 0.01	6.4 ± 0.05*	205 ± 20†	0.45 ± 0.12	0.31 ± 0.01	0.04 ± 0.006	<0.01
	20-30	175 ± 5	0.76 ± 0.01	6.4 ± 0.05	321 ± 10**	0.42 ± 0.11	0.31 ± 0.02	0.10 ± 0.003	<0.01
S_{till}	0-10	158 ± 2	0.62 ± 0.01***	6.7 ± 0.08	133 ± 13†	0.49 ± 0.08	0.31 ± 0.01†	0.01 ± 0.001†	<0.01
	10-20	166 ± 2	0.65 ± 0.02***	6.6 ± 0.07***	140 ± 7†	0.55 ± 0.09	0.30 ± 0.03	0.02 ± 0.002†	<0.01
	20-30	175 ± 2	0.69 ± 0.02	6.5 ± 0.08	184 ± 13***	0.61 ± 0.14	0.33 ± 0.02	0.04 ± 0.006	<0.01
CR_{surf}	0-10	164 ± 2†	0.76 ± 0.02***	6.7 ± 0.03†	139 ± 2†	0.59 ± 0.03	0.30 ± 0.02	0.01 ± 0.001†	<0.01
	10-20	164 ± 1	0.76 ± 0.01	6.7 ± 0.04***	149 ± 6†	0.49 ± 0.10	0.32 ± 0.01	0.02 ± 0.001†	<0.01
	20-30	165 ± 5	0.76 ± 0.01	6.5 ± 0.08	178 ± 4***	0.42 ± 0.13	0.29 ± 0.04	0.03 ± 0.003	<0.01
CR_{incorp}	0-10	160 ± 2	0.59 ± 0.01***	6.6 ± 0.12	142 ± 12	0.48 ± 0.11	0.30 ± 0.02	0.01 ± 0.002	<0.01
	10-20	170 ± 2***	0.65 ± 0.01***	6.7 ± 0.08***	159 ± 3	0.62 ± 0.16	0.35 ± 0.02	0.02 ± 0.001	<0.01
	20-30	178 ± 2	0.71 ± 0.01	6.6 ± 0.13	184 ± 10***	0.49 ± 0.17	0.34 ± 0.03	0.04 ± 0.008	<0.01

[a] 1:2.5 (w/v) field moist soil:distilled H_2O.

Table 2. Summary of the control, water table, fleece, cultivation, and residue treatment characteristics used in the experiment.

Treatment and code	Water table depth (cm)	Lettuce biomass (g FW cm^{-2} / t FW ha^{-1}) [a]	Cultivation (cm)	Soil cover
Control	>30 cm (free-draining)	None	None	None
Low water table (WT_{15})	15 cm below soil surface	None	None	None
High water table (WT_0)	0 cm (at soil surface)	None	None	None
Fleece (C_{fleece})	>30 cm (free-draining)	None	None	Fleece
Soil tillage (S_{till})	>30 cm (free-draining)	None	To 30 cm depth	None
Crop residue, surface applied (CR_{surf})	>30 cm (free-draining)	35.5 g cm^{-2} / 29.7 t ha^{-1}	None	Crop residue
Crop residue, incorporated (CR_{incorp})	>30 cm (free-draining)	35.5 g cm^{-2} / 29.7 t ha^{-1}	To 30 cm depth	None

[a] FW, fresh weight.

Table 3. Cumulative fluxes of CO_2, N_2O and CH_4, and total cumulative GHG emissions (GWP_{100}) in t CO_2-e ha^{-1} $period^{-1}$ (± SEM), for control, water table at -15 cm below soil surface (WT_{15}), water table at soil surface (WT_0), fleece cover (C_{fleece}), cultivated (S_{till}), surface applied crop residue (CR_{surf}), and incorporated crop residue (CR_{incorp}) treatments. For the water table treatments, totals are reported separately for the wetted (Phase I; months 0-3), drained (Phase II; months 4-6), and whole measurement period (Phase I + II; 0-6 months). Values are presented as mean ± SEM. Significant differences among values for each treatment (within each column) at the $p < 0.05$ level are marked with different letters, with separate comparisons made between (1) Control, WT_{15} and WT_0 (denoted a-c), (2) Control and C_{fleece} (denoted d-e), (3) Control and S_{till} (ns), (4) Control and CR_{surf} (denoted f-g), (5) Control and CR_{incorp} (ns), and CR_{surf} and CR_{incorp} (denoted h-i).

Treatment	Phase I t CO_2-e ha^{-1} 80 d^{-1}				Phase II t CO_2-e ha^{-1} 69 d^{-1}				Phase I + II t CO_2-e ha^{-1} 153 d^{-1}			
	CO_2	N_2O	CH_4	GWP_{100}	CO_2	N_2O	CH_4	GWP_{100}	CO_2	N_2O	CH_4	GWP_{100}
Control	5.87 ± 0.06 a,d,f	0.55 ± 0.10 a,f	0.00 ± 0.01	6.43 ± 0.11 a,d,f	4.09 ± 0.29 a	0.71 ± 0.25	0.01 ± 0.01	4.81 ± 0.31	10.29 ± 0.35 a	1.36 ± 0.37 a	0.01 ± 0.01	11.66 ± 0.42 a
WT_{15}	5.72 ± 0.22 ab	7.70 ± 0.92 b	-0.00 ± 0.01	13.41 ± 0.90 b	4.58 ± 0.11 ab	0.74 ± 0.12	0.00 ± 0.02	5.32 ± 0.20	10.61 ± 0.30 a	8.82 ± 1.11 b	0.00 ± 0.02	19.42 ± 1.14 b
WT_0	0.85 ± 0.12 b	1.16 ± 0.37 a	-0.00 ± 0.01	2.01 ± 0.45 c	5.30 ± 0.23 b	0.44 ± 0.21	0.01 ± 0.01	5.75 ± 0.37	6.47 ± 0.20 b	1.71 ± 0.43 a	0.01 ± 0.01	8.19 ± 0.58 c
S_{till}	5.63 ± 0.22	0.50 ± 0.10	0.01 ± 0.00	6.14 ± 0.27								
C_{fleece}	7.83 ± 0.58 e	1.20 ± 0.25	0.03 ± 0.04	9.07 ± 0.58 e								
CR_{surf}	7.07 ± 0.26 g,h	1.42 ± 0.29 g	-0.05 ± 0.02	8.44 ± 0.30 g								
CR_{incorp}	5.99 ± 0.18 i	0.78 ± 0.22	0.01 ± 0.01	6.79 ± 0.34								

Table 4. Significant linear correlations between measured environmental variables and emissions of CO_2, N_2O and CH_4 for control, water table at -15 cm below soil surface (WT_{15}), water table at soil surface (WT_0), fleece cover (C_{fleece}), cultivated (S_{till}), surface applied crop residue (CR_{surf}), and incorporated crop residue (CR_{incorp}) treatments. The values are reported separately for comparison against the water table treatments for the wetted (Phase I; months 0-3), drained (Phase II; months 4-6), and whole measurement period (Phase I + II; 0-6 months). Values are presented as Kendall's tau statistic (τ), with significance levels presented as * ($p < 0.05$), ** ($p < 0.01$), or *** ($p < 0.001$).

Treatment		Soil redox potential, E_h (mV)				Temperature			Rainfall		Nitrogen availability		
		Soil depth (cm)				Soil temp. [a]	Mean air temp. [b]	Air temp. [c]	Daily rain [d]	5 d rain [e]	NO_3-N	NH_4-N	N
		0 cm	10 cm	20 cm	30 cm	(°C)	(°C)	(°C)	(mm)	(mm)	(mg l^{-1})	(mg l^{-1})	(mg l^{-1})
CO_2	Control, wetted					0.539***	0.617***	0.322***		-0.174*		0.254**	
	WT_{15}, wetted					0.559***	0.538***	0.420***	-0.238**	-0.360***	-0.152*	0.254**	-0.199*
	WT_0, wetted								-0.169*				
	Control, drained			0.176*		0.345***	0.384***	0.231**		-0.219**	0.182*	-0.187*	
	WT_{15}, drained			0.179*		0.443***	0.442***	0.357***	-0.279***	-0.460***	0.445***		
	WT_0, drained				-0.174*	0.474***	0.481***	0.395***	-0.289***	-0.404***			
	Control, whole period					0.381***	0.528***	0.279***		-0.212***			
	WT_{15}, whole period					0.353***	0.523***	0.359***	-0.236***	-0.407***	-0.111*	0.135*	
	WT_0, whole period						0.162**		-0.236***	-0.130***	-0.298***		-0.191**
	C_{fleece}					0.539***	0.595***	0.365***		-0.153*			
	S_{till}					0.341***	0.392***	0.365***			0.243**	0.255**	
	CR_{surf}						0.230**		0.180*				0.216**
	CR_{incorp}						0.166*		-0.112*		0.219*		
N_2O	Control, wetted								-0.212**		0.185*		
	WT_{15}, wetted					0.180*			-0.579***	-0.260***			
	WT_0, wetted								-0.357***				0.207*
	Control, drained												
	WT_{15}, drained				-0.174*	0.283***	0.258**	0.345***	-0.271**		0.358***		0.254**
	WT_0, drained				-0.254*	0.285**	0.160*	0.302**	-0.216*				
	Control, whole period								-0.136*				
	WT_{15}, whole period					0.313***		0.204***	-0.440***	-0.199***	0.347***		0.241***
	WT_0, whole period					0.153**		0.168**	-0.291***				
	C_{fleece}					0.147*			-0.237**				
	S_{till}								-0.240**				
	CR_{surf}						-0.185*	-0.171*	-0.186*			-0.292**	
	CR_{incorp}						-0.171*		-0.407***				
CH_4	Control, wetted												
	WT_{15}, wetted												
	WT_0, wetted												
	Control, drained					-0.170*	-0.164*	-0.179*					
	WT_{15}, drained												
	WT_0, drained												
	Control, whole period												
	W_{15}, whole period												
	WT_0, whole period												
	C_{fleece}										0.179*	-0.239**	
	S_{till}												
	CR_{surf}					-0.461*		-0.199**					
	CR_{incorp}												

[a] Soil temp., soil temperature at the time of GHG measurement; [b] Mean air temp., mean daily air temperature on the day of the GHG measurement; [c] Air temp., temperature at the time the GHG measurement was made; [d] Daily rain, rainfall on the day of GHG measurement; [e] 5 d rain, cumulative rainfall in the 5 d preceding the GHG measurement.

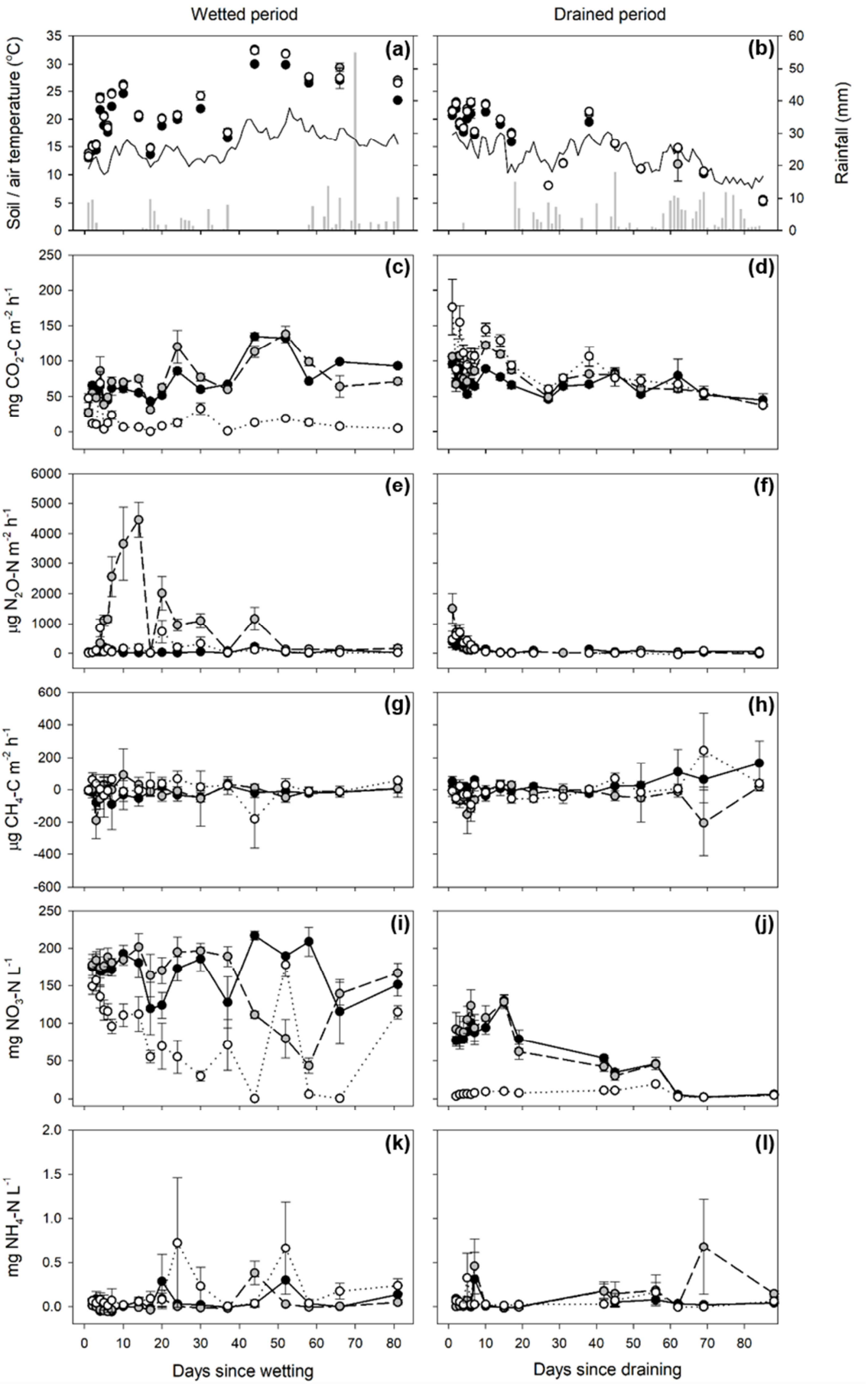
Wetted period
Drained period
(a)
(b)
(c)
(d)
(e)
(f)
(g)
(h)
(i)
(j)
(k)
(l)
Soil / air temperature (°C)
Rainfall (mm)
mg CO2-C m-2 h-1
µg N2O-N m-2 h-1
µg CH4-C m-2 h-1
mg NO3-N L-1
mg NH4-N L-1
Days since wetting
Days since draining

Fig. 1. Daily rainfall, air temperature and soil temperature (a-b); fluxes of CO_2 (c-d), N_2O (e-f), and CH_4 (g-h); and soil water NO_3^- (i-j) and NH_4^+ (k-l); 28th May to 16th Aug. (Phase I, wetted) and 21st Aug. to 13th Nov. 2013 (Phase II, drained). In panels (a)-(b), mean daily air temperature (°C) is denoted by a solid black line, rainfall (mm) by grey bars, and mean soil temperature by solid black circles (free-draining control), grey circles (water table at 15 cm below the soil surface, WT_{15}), and white circles (water table at the soil surface, WT_0). In panels (c)-(l), the control treatment is denoted by black circles with a solid line, WT_{15} by grey circles with a dashed line, and WT_0 by white circles with a dotted line. Error bars represent ± SEM.

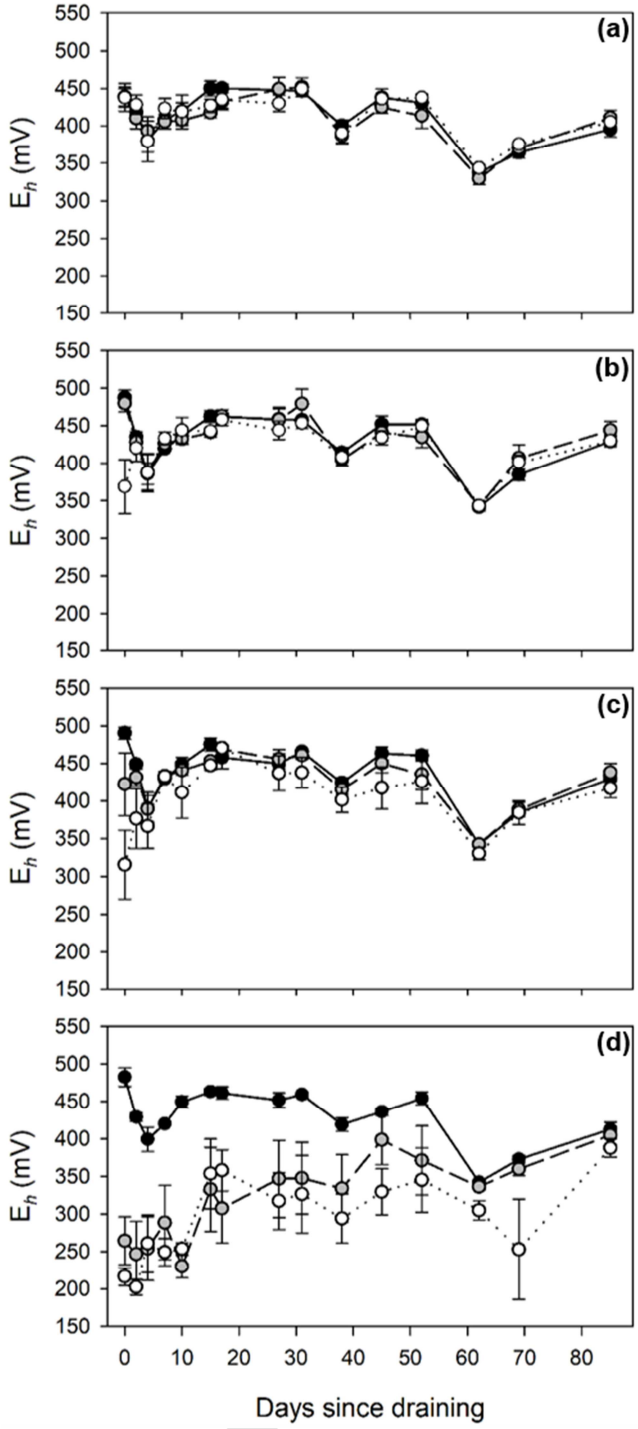

Fig. 2. Redox potentials (E_h) at soil depths of 0 cm (a), 10 cm (b), 20 cm (c), and 30 cm (d); 21st Aug. to 13th Nov. 2013 (Phase II, drained). The free-draining control treatment is denoted by black circles with a solid line, WT_{15} (water table at 15 cm below the soil surface) by grey circles with a dashed line, and WT_0 (water table at the soil surface) by white circles with a dotted line. Error bars represent ± SEM.

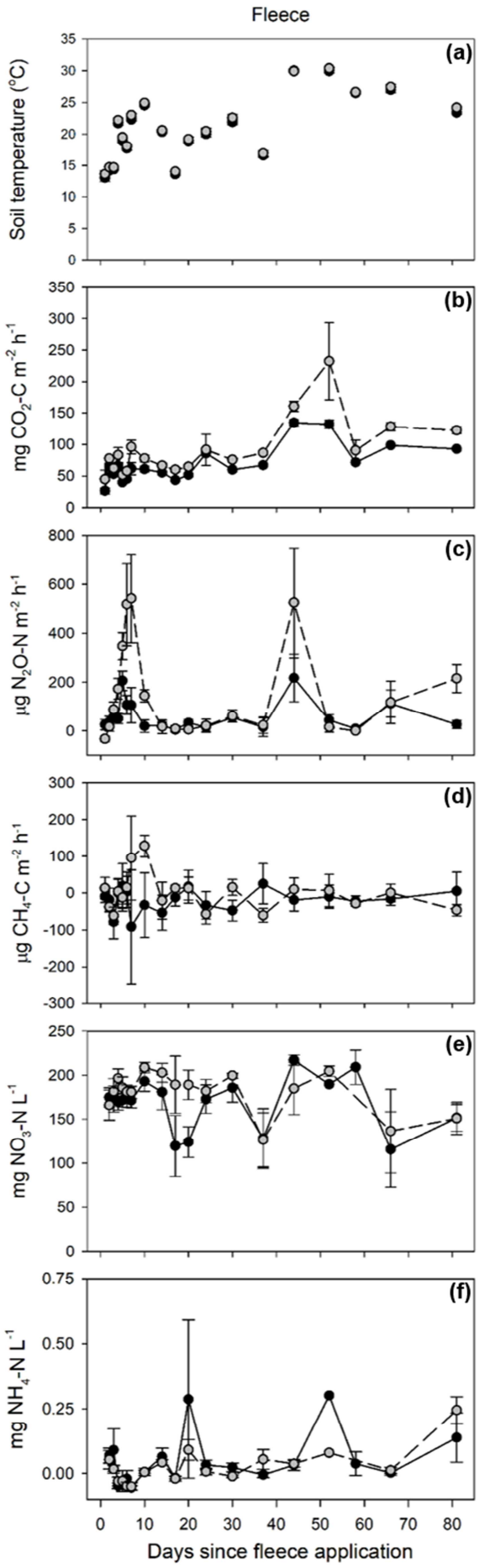

Fig. 3. Soil temperature (a); fluxes of CO_2 (b), N_2O (c), and CH_4 (d); and soil water NO_3^- (e) and NH_4^+ (f); 28th May to 16th Aug. 2013. In panel (a), mean soil temperature is denoted by solid black circles (uncovered control), and grey circles (fleece applied, C_{fleece}). In panels (b)-(f), the control treatment is denoted by black circles with a solid line, and C_{fleece} by grey circles with a dashed line. Error bars represent ± SEM.

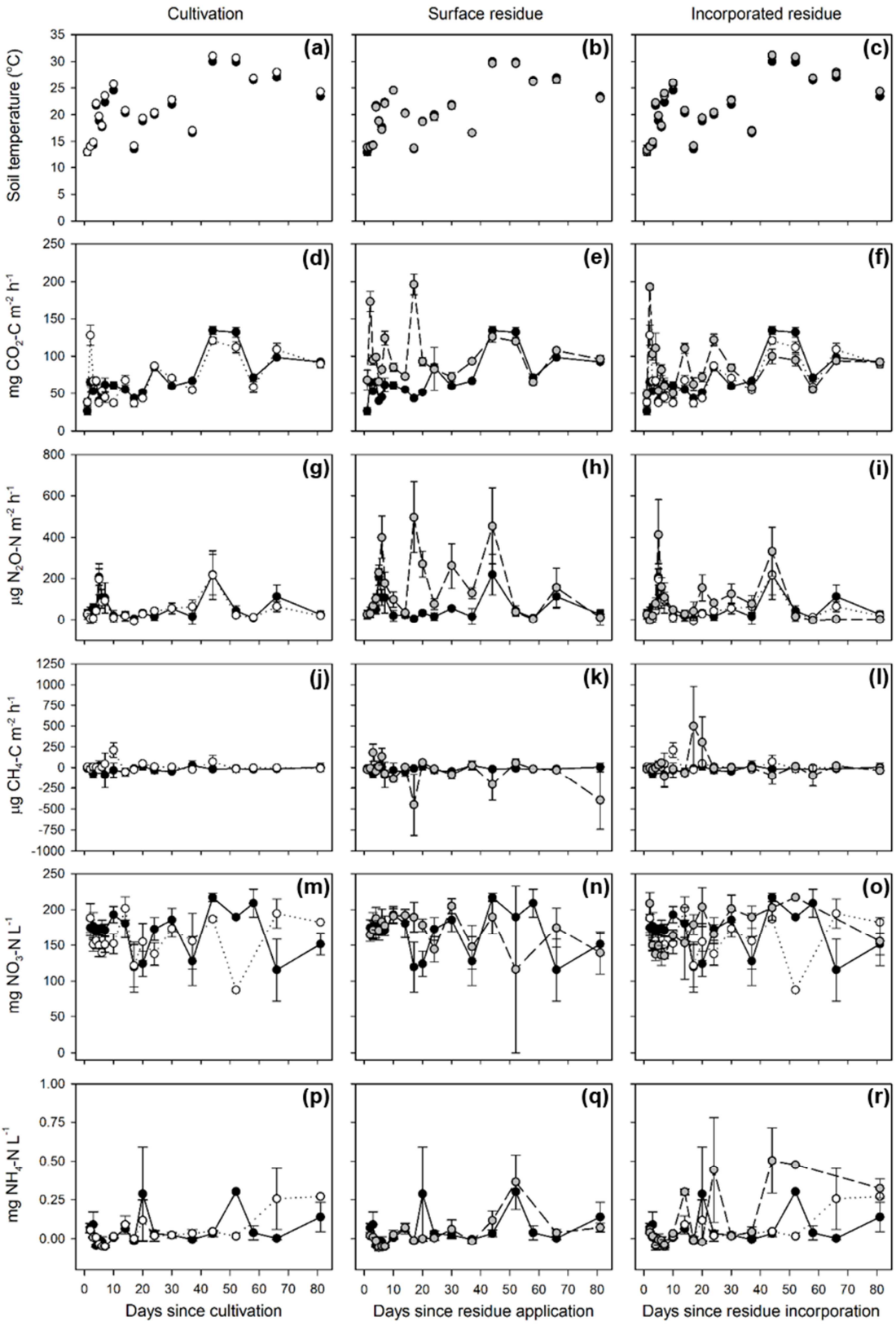
Cultivation
Surface residue
Incorporated residue
(a)
(b)
(c)
Soil temperature (°C)
(d)
(e)
(f)
mg CO2-C m-2 h-1
(g)
(h)
(i)
µg N2O-N m-2 h-1
(j)
(k)
(l)
µg CH4-C m-2 h-1
(m)
(n)
(o)
mg NO3-N L-1
(p)
(q)
(r)
mg NH4-N L-1
Days since cultivation
Days since residue application
Days since residue incorporation

Fig. 4. Soil temperature (a-c); fluxes of CO_2 (d-f), N_2O (g-i), and CH_4 (j-l); and soil water NO_3^- (m-o) and NH_4^+ (p-r); 28th May to 16th Aug. In panels (a)-(c), mean soil temperature is denoted by solid black circles (control without cultivation or residue), solid grey circles (surface applied residue, CR_{surf}, or incorporated residue, CR_{incorp}), and white circles (simulated tillage, S_{till}). In panels (d)-(r), the control treatment is denoted by black circles with a solid line, CR_{surf} and CR_{incorp} by grey circles with a dashed line, and S_{till} by white circles with a dotted line. Error bars represent ± SEM.

RESEARCH HIGHLIGHTS

- Greenhouse gas (GHG) emissions were measured in a horticultural fen peat soil.
- CO_2 and N_2O emissions were highly sensitive to water table depth changes.
- Tillage and horticultural fleece had no appreciable impact on GHG emissions.
- Crop residue addition did not appear to induce positive SOM priming.
- Alternative land uses are likely required to preserve these soils in the long-term.