

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

Taft, Helen; Cross, Paul; Jones, Davey L.

#### Soil Biology and Biochemistry

DOI: 10.1016/j.soilbio.2018.08.020

Published: 01/12/2018

Peer reviewed version

Cyswllt i'r cyhoeddiad / Link to publication

Dyfyniad o'r fersiwn a gyhoeddwyd / Citation for published version (APA): Taft, H., Cross, P., & Jones, D. L. (2018). Efficacy of mitigation measures for reducing greenhouse gas emissions from intensively cultivated peatlands. Soil Biology and Biochemistry, 127, 10-21. https://doi.org/10.1016/j.soilbio.2018.08.020

Hawliau Cyffredinol / General rights Copyright and moral rights for the publications made accessible in the public portal are retained by the authors and/or other copyright owners and it is a condition of accessing publications that users recognise and abide by the legal requirements associated with these rights.

• Users may download and print one copy of any publication from the public portal for the purpose of private study or research.

- You may not further distribute the material or use it for any profit-making activity or commercial gain
   You may freely distribute the URL identifying the publication in the public portal ?

Take down policy If you believe that this document breaches copyright please contact us providing details, and we will remove access to the work immediately and investigate your claim.

# Accepted Manuscript

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

Helen E. Taft, Paul A. Cross, Davey L. Jones

PII: S0038-0717(18)30274-8

DOI: 10.1016/j.soilbio.2018.08.020

Reference: SBB 7258

To appear in: Soil Biology and Biochemistry

Received Date: 14 March 2018

Revised Date: 15 August 2018

Accepted Date: 17 August 2018

Please cite this article as: Taft, H.E., Cross, P.A., Jones, D.L., Efficacy of mitigation measures for reducing greenhouse gas emissions from intensively cultivated peatlands, *Soil Biology and Biochemistry* (2018), doi: 10.1016/j.soilbio.2018.08.020.

This is a PDF file of an unedited manuscript that has been accepted for publication. As a service to our customers we are providing this early version of the manuscript. The manuscript will undergo copyediting, typesetting, and review of the resulting proof before it is published in its final form. Please note that during the production process errors may be discovered which could affect the content, and all legal disclaimers that apply to the journal pertain.



- 1 Efficacy of mitigation measures for reducing greenhouse gas emissions from intensively
- 2 cultivated peatlands
- 3
- 4 Helen E. Taft<sup>\*</sup>, Paul A. Cross, Davey L. Jones
- 5 School of Environment, Natural Resources and Geography, Bangor University, Deiniol
- 6 Road, Bangor, Gwynedd, LL57 2UW, UK
- 7
- 8 <sup>\*</sup>*Corresponding author.*
- 9 *Email address:* <u>h.taft@bangor.ac.uk</u>

#### 10 ABSTRACT

11 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 12 gas (GHG) emission. We hypothesised that GHG losses from these soils could be reduced by 13 manipulating water table depth, tillage regime, crop residue application or horticultural fleece 14 cover. Using intact soil columns from a horticultural peatland, emissions of CO<sub>2</sub>, N<sub>2</sub>O and 15 CH<sub>4</sub> were monitored over a six-month period, using a closed-chamber method. Concurrent 16 measurements of soil properties allowed identification of the key controls on GHG emissions. 17 18 Raising the water table to the soil surface provided the strongest reduction in global warming potential (*GWP*<sub>100</sub>;  $26 \pm 6$  kg CO<sub>2</sub>-e ha<sup>-1</sup> d<sup>-1</sup>), compared to a free-draining control ( $81 \pm 1$  kg 19  $CO_2$ -e ha<sup>-1</sup> d<sup>-1</sup>), but this effect was partially negated by an emission pulse when the water 20 table was subsequently lowered. The highest emissions occurred when the water table was 21 maintained 15 cm below the surface  $(172 \pm 12 \text{ kg CO}_2\text{-e ha}^{-1} \text{ d}^{-1})$ , as this stimulated N<sub>2</sub>O loss. 22 Placement of horticultural fleece over the soil surface during spring had no significant effect 23 on  $GWP_{100}$ , but prolonged fleece application exacerbated GHG emissions. Leaving lettuce 24 crop residues on the surface increased soil  $GWP_{100}$  (106 ± 4 kg CO<sub>2</sub>-e ha<sup>-1</sup> d<sup>-1</sup>) in comparison 25 to when residues were incorporated into the soil ( $85 \pm 4 \text{ kg CO}_2$ -e ha<sup>-1</sup> d<sup>-1</sup>), however, there 26 was no evidence that this promoted positive priming of native soil organic matter (SOM). For 27 maximum abatement potential, mitigation measures should be applied during the growing 28 29 season, when GHG emissions are greatest. Our results also suggest that introduction of zeroor minimum-till practices may not reduce GHG emissions. Maintaining a high water table 30 was the only option that reliably reduced GHG emissions, however, this option is impractical 31 32 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 33 explored as a means of preserving these soils for future generations. 34

*Keywords:* Carbon cycling; Food security; Greenhouse gases; Histosol; Sustainable cropping
 37

#### 38 1. Introduction

35

Approximately 14-20% of peatlands globally are used for agriculture and when 39 drained and cultivated they represent some of the world's most productive agricultural soils 40 (IPS, 2008). Their management is highly problematic, however, due to the potential for soil 41 loss, either from wind or water erosion or from microbial mineralisation of the peat substrate 42 43 (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 44 also progressively diminishes the resource base (Cannell et al., 1999). There is therefore a 45 46 clear ecosystem services trade-off between (1) preserving (and enhancing) peat carbon (C) storage for climate change mitigation, maintaining high biodiversity habitats, and improving 47 water quality, and (2) using this resource to promote food security. 48

49 In many temperate and tropical countries, agricultural peatland emissions dominate national emissions of greenhouse gas (GHGs) from peat sources (IPS, 2008). For example, it 50 has been estimated that 39% of English deep fen peats are currently under intensive 51 cultivation and classed as being at risk from severe soil loss (Natural England, 2010). Within 52 these sites, the depth of soil has been declining by 0.27-3.09 cm y<sup>-1</sup> since the onset of 53 drainage and cultivation in 1850 (Richardson and Smith, 1977; Hutchinson, 1980; Dawson et 54 al., 2010). It has been estimated that 35-100% of drained Histosol loss may be attributable to 55 microbially mediated CO<sub>2</sub> production (Leifeld et al., 2011). The small net consumption of 56 57 CH<sub>4</sub> in these soils does little to offset CO<sub>2</sub> loss, whilst N<sub>2</sub>O emissions can be substantial, forming approximately one third to one half of the total GHG budget (Taft et al., 2017). 58 Mitigating GHG emissions from these soils is therefore a priority, especially as this could 59

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 62 factors, including those associated with soil (e.g., porosity, labile C), climate (rainfall, 63 temperature), and vegetation (growth rate, rooting depth), which in turn are driven by 64 agricultural management strategy (Li, 2007). Modifying a single factor may simultaneously 65 increase emissions of one GHG and result in the reduction of another (Smith et al., 2008). 66 Therefore, mitigation studies should consider the overall effect of a measure on the total 67 68 emissions of CO<sub>2</sub>, CH<sub>4</sub> and N<sub>2</sub>O, 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 69 where measures to reduce CO<sub>2</sub> emission increase the release of the more radiatively powerful 70 71 CH<sub>4</sub> and N<sub>2</sub>O, causing a disproportionately large increase in the overall global warming potential (GWP) of the system. Given the relationship between GHG efflux and soil organic 72 C (SOC) loss (Dawson and Smith, 2007), and the importance of SOC to long-term soil 73 74 sustainability, it is also useful for mitigation studies to include an estimate of the effects of treatments on SOC retention. 75

While many reviews on GHG mitigation in arable systems exist, few contain 76 interventions specific to cultivated peatlands (e.g., Jauhiainen et al., 2016). Further, much of 77 the evidence remains inconclusive. Our aim was to evaluate whether common management 78 79 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 80 to promote SOC retention in cultivated peatlands. We hypothesised that tillage would 81 82 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 83 would increase soil temperature and moisture retention thereby promoting GHG emissions, 84

- while addition of crop residues might reduce GHG emissions through negative priming ofSOM.
- 87

#### 88 2. Methods and materials

89 2.1. Study sites

Soils (Sapric Histosols; FAO, 2006) utilised in this study originate from a 90 horticultural lowland peatland in East Anglia, UK (52°32' N, 0°29' E). The site has a mean 91 annual rainfall of < 700 mm, a mean annual temperature of 10.2 °C (ranging from mean 4.2 92 °C in winter to 17.2 °C in summer), and mean annual sunshine hours of 1550 (UK MetOffice, 93 2014). The study area comprises drained lowland fen typified by flat topography, which is 94 under intensive commercial-scale horticultural and arable production, growing primarily 95 96 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 97 graveolens L.]), sometimes in rotation with cereals (primarily wheat [Triticum aestivum L.]). 98 Soil was collected from a representative field (~70% SOM content; Taft et al., 2017), which 99 had been under a typical rotation for the previous growing season. Table 1 shows the physical 100 and chemical characteristics of the soils used in the experiments. 101

102

## 103 2.2. Field sampling

Intact soil cores were taken from a visually representative area  $(10 \text{ m}^2)$  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<sub>2</sub>,
CH<sub>4</sub> and N<sub>2</sub>O prior to experimentation (no significant differences among cores were
apparent; data not presented).

113

114 2.3. Preliminary soil and residue analysis

Five additional cores were taken from the field and a number of chemical and 115 physical analyses performed before commencement of the experiment; the same analyses 116 were conducted at the end of the experiment on all cores (Table 1). The cores were split into 117 three layers (0-10, 10-20 and 20-30 cm depth) and analyses were performed on each layer. A 118 Rhizon<sup>®</sup> suction sampler was inserted to 10 cm depth and a soil water sample obtained then 119 stored at c. -20 °C to await analysis. Next, a soil sample was taken using a bulk density ring 120  $(h_{total} = 10 \text{ cm}, V_{total} = 200 \text{ cm}^3)$  for calculation of soil gravimetric moisture content and bulk 121 density after oven drying (105 °C, 24 h). The remaining soil was homogenised and stored at 122 4°C prior to chemical analysis within 48 h. Soil samples extracts were performed in triplicate 123 for each soil layer for the determination of available  $NO_3^-$  and  $NH_4^+$  (5 g soil in 25 ml 0.5 M 124 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 125 NH<sub>4</sub>Cl). Extracts were obtained by shaking (200 rev min<sup>-1</sup>, 30 min), centrifugation (3,250  $\times$ 126 g, 10 min), filtering through a Whatman 42 filter paper and storage at -20 °C to await 127 analysis. Available soil NO<sub>3</sub>, NH<sub>4</sub><sup>+</sup> and P were determined colorimetrically on a PowerWave 128 XS microplate spectrophotometer (BioTek UK, Bedfordshire, UK) using the methods of 129 Mulvaney (1996), Miranda et al. (2001), and Murphy and Riley (1962) respectively. 130 Available K in the acetic acid extracts was determined with a Model 410 flame photometer 131 (Sherwood Scientific Ltd., Cambridge, UK). The moisture content of residue samples was 132 determined by oven drying (80 °C, 72 h), while total C and N was determined with a 133 CHN2000 analyser (Leco Corp., St Joseph, MI, USA). 134

135

#### 136 *2.4. Experimental treatments*

The cores were randomly assigned to six treatments as follows: (1) Control, (2) Water 137 table maintained at 15 cm below the surface  $(WT_{15})$ , (3) Water table maintained at the soil 138 surface  $(WT_0)$ , (4) Soil surface covered with horticultural fleece ( $C_{fleece}$ ), (5) Simulated tillage 139  $(S_{till})$ , (6) Crop residues applied to the soil surface (CR<sub>surf</sub>), and (7) Soil tilled and crop 140 residues incorporated into the soil (CR<sub>incorp</sub>) (Table 2). Each core had mesh covering the base 141 and was placed in larger plastic container to allow accurate water table control 142 (Supplementary information Appendix A, Fig. A.1). Sand surrounded the outside of the core 143 to minimise thermal gradients and holes drilled in the side of the containers to allow drainage, 144 or maintenance of the water table in the  $WT_0$  and  $WT_{15}$  treatments. The mesocosms were laid 145 146 out in a randomised block design with five replicates of each treatment, with blocks aligned to the prevailing wind direction (SW-NE) to account for differences in sheltering and 147 evapotranspiration. Water tables were established by filling the containers with artificial 148 rainwater solution (containing 96  $\mu$ mol L<sup>-1</sup> NaCl, 10  $\mu$ mol L<sup>-1</sup> K<sub>2</sub>SO<sub>4</sub>, 5  $\mu$ mol L<sup>-1</sup> 149 CaCl<sub>2</sub>.2H<sub>2</sub>O, 6 µmol L<sup>-1</sup> MgCl<sub>2</sub>.6H<sub>2</sub>O, 15 µmol L<sup>-1</sup> NH<sub>4</sub>NO<sub>3</sub>, and 0.1 µmol L<sup>-1</sup> KH<sub>2</sub>PO<sub>4</sub>, 150 reflecting average Welsh rainwater composition; Stevens et al., 1997) until the excess ran out 151 of the lateral drainage holes. Subsequently, water table height was maintained with natural or 152 artificial rain water. For the C<sub>fleece</sub> treatment, white horticultural, unwoven polypropylene 153 fleece was secured over the top of the core headspace using plastic-coated wire. Horticultural 154 fleece can be used for a variety of purposes including crop protection from frosts or pests and 155 diseases, and soil warming and protection from wind or water erosion (e.g., Olle and Bender, 156 2010). At our study site, it is used primarily for soil warming and crop protection against 157 frosts, to facilitate the production of early crops. Cultivation treatments were based on the 158 typical ploughing depth at the field site (c. 30-35 cm), and were implemented by removing 159

160 the whole volume of soil from the core, mixing in crop residues where appropriate, and 161 packing loosely back into the core. Soil residue treatments involved the addition of Iceberg 162 lettuce (*Lactuca sativa* L.) residues (c.  $5 \times 5$  cm pieces) to the soil based on rates measured in 163 the field post-harvest (52% of the total crop; 0.9 t C ha<sup>-1</sup>). The residues were pressed into the 164 soil surface to simulate post-harvest tractor traffic.

Mesocosm measurements were made for seven consecutive days following treatment 165 application (May and Aug. 2013), then twice per week for two weeks, then weekly until the 166 end of each experimental period (Aug. and Nov 2013). The experiment had two phases for 167 the water table treatments ( $WT_0$  and  $WT_{15}$ ): Phase I involved maintaining the water table at 168 the target depth for 3 months (i.e. 0 or -15 cm), while in Phase II the water table was lowered 169 (by drilling holes in the base of the container) to match the control treatment (i.e. -30 cm). 170 After 6 months, observable differences in GHG emissions among the water table treatments 171 were largely negligible. Consequently, the cores were dismantled, split into 10 cm depth 172 fractions and analysed as outlined in Section 2.3. 173

174

175 2.5. Greenhouse gas monitoring

Closed, non-vented static chambers were used to measure emissions of CH<sub>4</sub> and N<sub>2</sub>O. 176 These consisted of white opaque polypropylene cylindrical chambers (headspace 0.66 dm<sup>3</sup>) 177 with a rubber septum sampling port in the lid (Supplementary information Appendix A, Fig. 178 A.1). Each chamber was attached immediately before taking the first gas sample ( $t = t_0$ ), 179 giving a final average enclosed headspace of 1.72 dm<sup>3</sup>. Subsequent samples were taken at 180 approximately 10 min intervals ( $t = t_{10}, t_{20}$  and  $t_{30}$ ). Gas sampling and storage procedures and 181 materials followed those described in Taft et al. (2017). Sample analysis was undertaken with 182 a gas chromatograph (Varian 450-GC, Bruker UK Ltd., Coventry, UK), equipped with a 183 flame ionisation detector (FID, operated at 120-125 °C) and electron capture detector (ECD, 184

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<sub>2</sub> emissions from the cores with an EGM-4 infra-red gas analyser (PP
Systems, Hitchin, UK) equipped with an SRC-1 soil respiration chamber.

190 2.6. Soil water, climate and redox measurements

Soil temperature was measured with a Checktemp1<sup>®</sup> probe ( $\pm 0.3$  °C; Hanna 191 Instruments Ltd, Leighton Buzzard, UK) over a 0-10 cm depth. Soil solutions were recovered 192 non-destructively throughout the experiment using Rhizon<sup>®</sup> soil water samplers (Rhizosphere 193 Research Products, Wageningen, The Netherlands) inserted into the topsoil (0-10 cm depth). 194 Soil solutions were stored at -20 °C to await analysis. During experimental Phase II, soil 195 surface (1-2 cm depth) redox potential ( $E_{\rm h}$ ) was measured using an Eijkelkamp BNC glass 196 Platinum electrode with an Ag/AgCl reference electrode and 3 M KCl electrolyte 197 (Eijkelkamp Agrisearch Equipment, Giesbeek, The Netherlands) following Eijkelkamp 198 (2009). Sampling ports in the side of the core (at 10, 20 and 30 cm below the soil surface) 199 allowed additional temperature and  $E_{\rm h}$  measurements to be made. Rainwater samples were 200 collected periodically through the experiment and analysed for soluble N. Meteorological 201 data (rainfall, air temperature) were obtained from the local Met. Office monitoring station. 202 203

204 2.7. Statistical analysis

Statistical analyses were performed using SPSS v. 20 (IBM Corp., Armonk, NY), with significance being accepted at  $p \le 0.05$  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*<sub>100</sub>) CO<sub>2</sub> equivalents (CO<sub>2</sub>-e) according to IPCC (2006). Cumulative fluxes of CO<sub>2</sub>, N<sub>2</sub>O, CH<sub>4</sub> and total *GWP*<sub>100</sub>

210 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 211 significantly different treatments using Tukey's HSD, Gambrell-Howell, or Kolmogorov-212 Smirnov Z statistics (with Bonferroni correction for multiple comparisons) as appropriate. 213 Relationships among individual GHGs, temperature, rainfall, and soil N concentrations were 214 explored using Kendall's tau statistic ( $\tau$ ). 215 All statistical analyses were performed separately on the water table group of 216 treatments (Control vs. WT<sub>0</sub> vs. WT<sub>15</sub>), the fleece treatment (Control vs. C<sub>fleece</sub>), and the 217 cultivation and residue group of treatments (Control vs. Still vs. CR<sub>surf</sub> vs. CR<sub>incorp</sub>). Normality 218 was tested using the Shapiro-Wilk test (Field, 2005), and non-normal data were log<sub>10</sub>-219 transformed or square-root transformed; where transformation was ineffective, or where 220 221 heterogeneity of variances was observed (Levene's or Welch's test statistic), appropriate nonparametric tests were used to compare medians of those data groups. Soil physical and 222 chemical characteristics for each soil depth layer were compared using ANOVA or the 223 independent t-test, or Kruskall-Wallis or Kolmogorov-Smirnov Z tests for data deviating 224 greatly from normality or homogeneity of variances. Significant effects of treatment and time 225 (each treatment including the control, compared to the baseline) were tested. 226

227

#### 228 **3. Results**

#### 229 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

Soil respiration responded rapidly to raising of the water table, falling close to zero within 5 d of water table raising in the WT<sub>0</sub> treatment, and remaining lower (11 ± 1.4 mg CO<sub>2</sub>-C m<sup>-2</sup> h<sup>-1</sup>) than mean fluxes from the control and WT<sub>15</sub> treatments (76 ± 3.6 mg CO<sub>2</sub>-C m<sup>-2</sup> h<sup>-1</sup> and 78 ± 3.9 mg CO<sub>2</sub>-C m<sup>-2</sup> h<sup>-1</sup> respectively) for the remainder of the wetted period (Fig. 1c-d). Immediately after draining, there was a peak in CO<sub>2</sub> emissions from both the WT<sub>0</sub> and WT<sub>15</sub> treatments, however, these returned to values close to the control after a further 44 d.

During the wetted period, mean  $N_2O$  emissions ranged from  $5.0\pm6.0$  to  $4453\pm577$ 244  $\mu g~N_2 O\text{-}N~m^{-2}~h^{-1}$  across all treatments (Fig. 1e-f). A substantial peak (4453  $\pm$  577  $\mu g~N_2 O\text{-}N$ 245  $m^{-2} h^{-1}$ ) was observed from the WT<sub>15</sub> treatment after 14 d and emissions in this treatment 246 remained consistently higher than the WT<sub>0</sub> and control treatments during the first six weeks. 247 Over this period, N<sub>2</sub>O emissions were very similar in the control and WT<sub>0</sub> treatments. 248 Drainage resulted in a short-lived rise (c. 14 d) in N<sub>2</sub>O flux which was most pronounced in 249 the WT<sub>15</sub> treatment immediately following draining (1506  $\pm$  499 µg N<sub>2</sub>O-N m<sup>-2</sup> h<sup>-1</sup>). 250 Emissions in the WT<sub>0</sub> treatment exhibited a similar but smaller response 3 d after draining 251  $(699 \pm 277 \ \mu g \ N_2 O-N \ m^{-2} \ h^{-1})$ . Fluxes of CH<sub>4</sub> remained low throughout the experiment (Fig. 252 1g-h). 253

Cumulative GHG emissions were significantly influenced by water table depth (Table 3). In the initial wetted phase (Phase I), a significant decline in CO<sub>2</sub> emissions was apparent as the water table was raised closer to the soil surface. However, a significant difference was only observed between the control and WT<sub>0</sub> treatments (p < 0.01), although the difference between the WT<sub>15</sub> and WT<sub>0</sub> treatments was almost significant (p = 0.08). Cumulative N<sub>2</sub>O emission was significantly influenced by water table depth (p < 0.001), with the mean WT<sub>15</sub>

cumulative flux being significantly higher than both the control and  $WT_0$  treatments (both p <260 0.001). No significant treatment effects were observed for cumulative CH<sub>4</sub> emissions. 261 Cumulative  $GWP_{100}$  for water table treatments was significantly different among groups (p < p262 0.001); with a highly significant increase in the order  $WT_0 < control < WT_{15}$  (all p < 0.001). 263 In the drained period (Phase II), significant differences were recorded for median CO<sub>2</sub> 264 emissions among water table groups (p < 0.05; Table 3). However, no significant differences 265 were found among the three water table treatments for cumulative N<sub>2</sub>O, cumulative CH<sub>4</sub>, or 266  $GWP_{100}$ . 267

Over the entire experiment (Phase I and Phase II), CO<sub>2</sub> and N<sub>2</sub>O emissions were 268 highly influenced by water table depth (both p < 0.001; Table 3). There was a highly 269 significant decline in soil respiration between  $WT_{15}$  and  $WT_0$  treatments (p < 0.001), while 270 no difference was noted between the control and WT<sub>15</sub> treatments. Mean N<sub>2</sub>O emissions were 271 significantly higher from the WT<sub>15</sub> treatment compared to the control and WT<sub>0</sub> treatments 272 (both p < 0.001). There was no effect of water table depth on cumulative CH<sub>4</sub> emissions. 273 274 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<sub>0</sub> was lower than both the control 275 and WT<sub>15</sub> treatments (p < 0.05 and p < 0.001 respectively), and the control was lower than 276  $WT_{15}$  (*p* < 0.001). 277

278 Mean  $NO_3^-$  concentrations were substantially lower in the  $WT_0$  than in the control and 279 WT<sub>15</sub> treatments, both of which were similar to each other (Fig. 1i-j). Dissolved  $NH_4^+$ 280 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<sub>0</sub> treatment (369 ± 36 mV) than in the WT<sub>15</sub> and control treatments (480 ± 11 and 487 ± 10 mV, respectively; Fig. 2b). Upon

285	draining, an immediate and marked drop in $E_h$ was observed in the 20 cm soil layer in both
286	the WT $_0$ (315 $\pm$ 46 mV) and WT $_{15}$ (422 $\pm$ 42 mV) cores, compared with the control (490 $\pm$ 8
287	mV, Fig. 2c). Four days after draining, however, there were no observable differences among
288	treatments. Redox potentials in the 30 cm soil layer were the most responsive to water table
289	treatments (Fig. 2d). Both $WT_0$ and $WT_{15}$ treatments showed substantially lower mean $E_h$
290	values (218 $\pm$ 17 mV and 227 $\pm$ 19 mV, respectively) compared with the control cores (341 $\pm$
291	24 mV) for the first 38 d. By day 62, $WT_{15}$ redox values had returned to that of the control
292	values, whereas the WT <sub>0</sub> $E_{\rm h}$ took 85 d to recover to levels seen in the control.
293	

# 294 3.3. Effect of fleece application on GHG emissions and soil chemistry

Soil respiration from the C<sub>fleece</sub> and control cores followed a similar pattern 295 throughout the experiment although the fluxes were generally higher in the C<sub>fleece</sub> treatment 296 (Fig. 3b). The peak flux in the C<sub>fleece</sub> treatment  $(232 \pm 61 \text{ mg CO}_2\text{-C m}^{-2} \text{ h}^{-1})$  occurred on day 297 52, and was almost double that of the control emission  $(132 \pm 6.6 \text{ mg CO}_2\text{-C m}^{-2} \text{ h}^{-1})$ . Mean 298 N<sub>2</sub>O emissions were similar from the C<sub>fleece</sub> and control treatments throughout most of the 299 experimental period (Fig. 3c). Maximum  $N_2O$  emission from the  $C_{fleece}$  treatment (542  $\pm$  182 300  $\mu$ g N<sub>2</sub>O-N m<sup>-2</sup> h<sup>-1</sup>) occurred 7 d after fleece application, returning to control levels after 14 d. 301 Emissions of CH<sub>4</sub> were higher than in the control treatment, however, these fluxes were still 302 very low (Fig. 3d). Mean C<sub>fleece</sub> NO<sub>3</sub>-N and NH<sub>4</sub>-N concentrations were very similar to the 303 304 control treatment on all sampling dates (Figs. 3e-f).

Overall, cores with fleece had significantly higher mean cumulative CO<sub>2</sub> emissions (p306 < 0.05; Table 3) while total N<sub>2</sub>O emission was also higher than the control (p = 0.06). The 307 fleece treatment had a significantly greater cumulative *GWP*<sub>100</sub> emission than the control (p <308 0.01).

310 3.4. Effect of cultivation tillage on GHG emissions and soil chemistry

Mean CO<sub>2</sub> fluxes in the tilled soil were very similar to the control on most sampling 311 dates, ranging from  $26 \pm 4.7$  to  $135 \pm 5.2$  mg CO<sub>2</sub>-C m<sup>-2</sup> h<sup>-1</sup> (Fig. 4d). A marked peak in CO<sub>2</sub> 312 release was observed immediately after simulated ploughing, however, this was of short 313 duration. For a few days during the experiment, S<sub>till</sub> CO<sub>2</sub> emissions were lower than in the 314 control cores. Overall, mean fluxes of N<sub>2</sub>O and CH<sub>4</sub> were similar to the control (Figs. 4g and 315 4j). Ploughing had no significant effect when compared to undisturbed soil on cumulative 316 individual GHG emissions or overall  $GWP_{100}$  (Table 3). We observed no consistent effect of 317 318 tillage on soluble N concentrations relative to the control throughout the experiment.

319

#### 320 3.5. Effect of residue incorporation on GHG emissions and soil chemistry

Both residue treatments showed a marked increase in soil respiration immediately 321 following surface application or incorporation into the soil, with elevated levels persisting for 322 three weeks after application (Fig. 4e-f). The response was generally lower when residues 323 were incorporated into the soil. Emissions of N<sub>2</sub>O responded positively to residue application, 324 but with a slower response (5-6 d), and over a longer period (37 d), compared to the control 325 treatment (Fig. 4h-i). In the CR<sub>incorp</sub> treatment, both soil respiration and N<sub>2</sub>O emissions were 326 lower than from the control towards the end of the experimental period. No marked effect of 327 residue treatment was observed for CH<sub>4</sub> emissions or soil solution N relative to the control 328 329 throughout the experiment (Figs. 4k-l, 4n-o and 4q-p).



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

337

#### 338 3.6. Effect of soil and weather conditions on GHG emissions

Redox potential at depth was significantly correlated with CO<sub>2</sub> (p < 0.05) and N<sub>2</sub>O (p<0.05) emissions, but not CH<sub>4</sub> release (p > 0.05) (Table 4). At 20 cm below the soil surface,  $E_h$  was positively associated with CO<sub>2</sub> emission in the control and WT<sub>15</sub> treatments, explaining 3% of the variability in soil respiration ( $\tau = -0.176$  to -0.179). At 30 cm depth,  $E_h$ was negatively associated with CO<sub>2</sub> emission in the WT<sub>0</sub> treatment, and N<sub>2</sub>O emission in the WT<sub>0</sub> and WT<sub>15</sub> treatments, explaining 3% of CO<sub>2</sub> emission variability and 3-6% of N<sub>2</sub>O

345 emission variability ( $\tau = -0.174$  to -0.254).

346 Soil temperature, mean daily air temperature, and measured air temperature were positive, highly significant predictors of soil respiration within most treatments, accounting 347 for between 12-31%, 3-38%, and 5-18% of fluxes respectively ( $\tau = 0.341$  to 0.559, p < 0.05348 to < 0.01; Table 4). Temperature variables were less suitable for predicting N<sub>2</sub>O emissions, 349 although some highly significant correlations were still apparent. Soil temperature, mean 350 daily air temperature, and measured air temperature at the time of sampling predicted 2-10%, 351 3-7%, and 3-12% of N<sub>2</sub>O emissions respectively ( $\tau = 0.147$  to 0.313, p < 0.05 to < 0.001). 352 Daily and 5-day rainfall (cumulative rainfall from the day of measurement and the 353 four preceding days) were negative highly significant predictors of CO<sub>2</sub> emissions for most of 354 the treatments ( $\tau = -0.112$  to -0.460; p < 0.05 to < 0.001), while daily rainfall was positively 355 significantly correlated with surface-applied residue CO<sub>2</sub> efflux ( $\tau = 0.180$ , p < 0.05; Table 4). 356 Daily rainfall explained 1-8% and 5-day rainfall explained 2-21% of soil respiration. 357 Emissions of N<sub>2</sub>O and daily rainfall were highly significantly negatively correlated in all but 358 the drained control treatment, accounting for 2-34% of emissions ( $\tau = -0.136$  to -0.579, p < -0.136 to -0.579 to -0.579359

360	0.05 to < 0.001). Cumulative 5-day rainfall was a significant predictor of N <sub>2</sub> O emission in the
361	WT <sub>15</sub> treatment only, explaining 4-7% of N <sub>2</sub> O flux ( $\tau$ = -0.199 to -0.260; $p$ < 0.001).
362	Dissolved N was a significant predictor of soil respiration in most treatments.
363	Emissions of $N_2O$ and $NO_3^-$ concentration were significantly positively correlated in the
364	control (Phase I) and $WT_{15}$ (Phase II, Phase I + II) treatments, with $NO_3^-$ accounting for 3-
365	13% of variability in N <sub>2</sub> O emission ( $\tau = 0.185$ to 0.358, $p < 0.05$ to $< 0.001$ ). Concentrations
366	of $NH_4^+$ were positively associated with soil respiration in the control (Phase I), $WT_{15}$ (Phase
367	I, Phase I + II), and S <sub>till</sub> treatments (2-7% of variability, $\tau$ = 0.135 to 0.255, <i>p</i> < 0.05 to <
368	0.01), but negatively associated with soil respiration in the control (Phase II) treatment (3%
369	of variability, $\tau = -0.187$ , $p < 0.05$ ). A significant correlation between dissolved $NH_4^+$
370	concentration and $N_2O$ emission was found in only the surface-applied residue treatment (9%
371	of variability, $\tau = -0.292$ , $p < 0.01$ ), and with CH <sub>4</sub> emissions in the fleece treatment (6% of
372	variability, $\tau = -0.239$ , $p < 0.01$ ; Table 4).
373	

#### 374 4. Discussion

#### 375 *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 376 uses, raising the water table in this study reduced CO<sub>2</sub> emissions, moreover, the magnitude of 377 the reduction proved highly sensitive to water table depth (Dinsmore et al., 2009; Freeman et 378 al., 1993; Lloyd, 2006; Kechavarzi et al., 2007). Maintaining the water table at the surface 379 also reduced N<sub>2</sub>O emissions. We ascribe this to a reduction in the nitrification rate and  $NO_3^{-1}$ 380 production and the complete denitrification of any NO<sub>3</sub><sup>-</sup> present to N<sub>2</sub> (Velthof and Oenema, 381 1997). Lowering the water table to 15 cm, however, resulted in greatly elevated N<sub>2</sub>O 382 emissions. This concurs with findings from Freeman et al. (1993) who also reported N<sub>2</sub>O 383 emission to be inversely correlated with water table depth. Our highest rate of N<sub>2</sub>O emission 384

in the water table treatments (4.5 mg N<sub>2</sub>O-N m<sup>-2</sup> h<sup>-1</sup>) was two orders of magnitude higher 385 than emissions from semi-natural peatland mesocosms observed by Freeman et al. (1993) and 386 Dinsmore et al. (2009), but similar to studies of arable peatlands (Flessa et al., 1998; Taft et 387 al., 2017; Weslien et al., 2012). A large initial peak in N<sub>2</sub>O emissions was observed in the 388  $WT_{15}$  treatment after raising the water table, while only a small pulse was seen in the  $WT_0$ 389 treatment. Conversely, the WT<sub>0</sub> treatment released most N<sub>2</sub>O after draining, while the N<sub>2</sub>O 390 pulse from the WT<sub>15</sub> treatment was smaller. These relatively rapid, short-lived, strong 391 responses to wetting and draining events in peat soils are common, with their magnitude 392 typically limited by soil moisture and soluble N (Li et al., 1992). Overall, there was no 393 marked effect of water table treatment on CH<sub>4</sub> production over the wetted or drained 394 experimental periods, contrary to the general trend of water table raising increasing emissions 395 396 (Bussell et al., 2010). Strictly anaerobic conditions required for substantial CH<sub>4</sub> emissions, however, may take a long time to develop (>1 y; Oomes et al., 1997), and in infrequently 397 flooded soils are typically found at lower profile depths than those sampled in this study 398 399 (Mitsch and Gosselink, 2000). The low rates of CH<sub>4</sub> release could also be due to a lack of methanogens, or the abundance of alternative electron acceptors and/or an efficient 400 population of methanotrophs in the topsoil. This is supported by measured redox values 401 which largely fell within the range associated with CO<sub>2</sub> production and CH<sub>4</sub> consumption 402 (400 to 500 mV) and N<sub>2</sub>O production (200 to 500 mV), but not for CH<sub>4</sub> production (-100 to -403 200 mV; Le Mer and Roger, 2001; Li, 2007; Mitsch and Gosselink, 2000). 404 This study simulated raising the water table during late spring followed by draining in 405

405 Introduction of the soll surface water table to within 15 cm of the soil surface would not 406 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 between summer crops, possibly over quite short fallow periods. The relative efficacy of 411 flooding as a GHG mitigation strategy may be enhanced by additional impacts such as weed 412 growth even during relatively short fallow periods; which could further reduce net  $GWP_{100}$ 413 through elevated net primary productivity and plant removal of  $NO_3^-$  (e.g., Kløve et al., 414 2017). Conversely, both the presence of weeds and labile organic matter input from post-415 harvest crop residues could result in substantial emissions of N<sub>2</sub>O and CH<sub>4</sub> (Le Mer and 416 Roger, 2001). The net effect of vegetation therefore merits further investigation. 417 Maintaining the water table at the correct level and ensuring it drains freely post-418 flooding could be challenging. Kechavarzi et al. (2007) suggest that close spacing of sub-419 surface drainage pipes (≤10 m) would be required to maintain a consistent water table level in 420 a sub-irrigated field. Some fields are not equipped with closely spaced drainage pipes, and 421 not all peat soils are sub-irrigated. Fluctuation of the water level between 0-15 cm of the soil 422 surface, either through poor water level maintenance or slow drainage post-flooding, is likely 423 to result in large pulses of GHGs, as was observed in the WT<sub>15</sub> treatment, entirely negating 424 the beneficial effect of flooding. This effect may be minimised if draining is undertaken in 425 cooler weather. Further, flooding poses a number of difficulties both agronomically and in 426 the context of the wider landscape. Implementation would require careful timing so that after 427 flooding, soil had time to dry sufficiently before subsequent in-field machinery operations. 428 Yields of subsequent crops could be reduced after flooding, or the costs of mineral fertiliser 429 increased: our results strongly imply that much of the soil nitrate was leached from the soil 430 columns during draining. In terms of wider landscape effects, leaching of nitrate into 431 watercourses poses a severe pollution risk, with associated costs for the grower. Further, if 432 flooding were to be implemented on a widespread scale, regulation would be required to 433

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

436

#### 437 4.2. Effect of fleece application on GHG emissions

This study found that fleece application significantly increased  $GWP_{100}$ , CO<sub>2</sub> release 438 and N<sub>2</sub>O emissions from soil. Fleece application is known to stabilise variations in soil 439 temperature and to reduce soil moisture loss (Hamouz et al., 2006; 2005; Siwek et al., 2013; 440 2012). In this study, temperature was the strongest predictor of soil respiration, showing a 441 significant positive correlation in the fleece-enclosed cores. This is consistent with other 442 studies on the effect of temperature on peat soil respiration (Estop-Aragonés and Blodau, 443 2012; Maljanen et al., 2002). Soil temperature has also been shown to positively correlate 444 with N<sub>2</sub>O emissions (Maljanen et al., 2002), although in this study the relationship was not 445 strong. 446

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

N<sub>2</sub>O emissions may substantially increase when fertilised soil is subjected to the warmer soil
temperatures associated with fleece application.

- 461
- 462 *4.3. Effect of tillage on GHG emissions*

Simulated ploughing resulted in an immediate, small and short-lived peak in soil 463 respiration but a negligible response of N<sub>2</sub>O. Ploughing-induced peaks in CO<sub>2</sub> emission from 464 cultivated Histosols have been noted by Elder and Lal (2008) and Reicosky et al. (2008), 465 although the response found in our study was several-fold lower than that of Elder and Lal 466 (2008) (625 mg CO<sub>2</sub>-C m<sup>-2</sup> h<sup>-1</sup>). Mean emissions from a bare-tilled peat measured by 467 Maljanen et al. (2002) (300 mg  $CO_2$ -C m<sup>-2</sup> h<sup>-1</sup>), were also higher than the peak emission of 468 135 mg CO<sub>2</sub>-C m<sup>-2</sup> h<sup>-1</sup> recorded in this study. Production of N<sub>2</sub>O was not stimulated by a 469 ploughing event. This contrasts with the findings of Elder and Lal (2008), however Maljanen 470 et al. (2002) and Weslien et al. (2012) also reported negligible effects of ploughing on N<sub>2</sub>O 471 emissions. It is probable that the considerably lower peak of N<sub>2</sub>O emissions observed here 472 compared with those of Elder and Lal (2008) are a result of suboptimal soil moisture 473 conditions inhibiting N<sub>2</sub>O production, owing to the comparatively good drainage and lower 474 bulk density of our tilled cores (Dalal et al., 2003). Our results are in strong contrast to the 475 assertion that cultivation results in a large efflux of both CO<sub>2</sub> and N<sub>2</sub>O (Dawson and Smith, 476 2007; Kasimir-Klemedtsson et al., 1997). This suggests that adoption of minimum or zero 477 tillage practices may not help preserve soil C on sites with a long history of cultivation. 478

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 <sup>-1</sup> ,
491	73 mg N core <sup>-1</sup> ) 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 <sup>-1</sup> (CR <sub>surf</sub> ) and 0.01 t C ha <sup>-1</sup> (CR <sub>incorp</sub> ), i.e. considerably less than the
497	quantity of residue-C added to the cores (0.90 t C ha <sup>-1</sup> ). 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 $N_2$ O-N ha <sup>-1</sup> 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 <sub>2</sub> 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).
F07	While residue in componentian resulted in lower emissions relative to surface emplication

507 While residue incorporation resulted in lower emissions relative to surface application508 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).

516

#### 517 **5.** Conclusions and implications

The results of this study suggest that the relative efficacy of potential GHG mitigation 518 options will be strongly influenced by the weather and soil conditions at the time of 519 implementation, and hold the greatest potential efficacy if applied during the main growing 520 season when GHG emissions are greatest. Net GHG emissions from the horticultural peat 521 soils in this study proved sensitive to water table depth, with flooding to the soil surface 522 being highly effective in reducing GHG emissions. However, avoiding a shallow water table 523 is paramount in minimising emissions. Our study suggests that horticultural fleece should be 524 used for the shortest possible period, and in cool weather only. Contrary to expectation, 525 tillage did not significantly increase net GHG emissions. We recommend that tillage and 526 harvesting operations should be conducted during cooler or damper weather to minimise the 527 small peak in emissions. The impacts of lettuce residue treatment were somewhat 528 inconclusive, with residue incorporation reducing net emissions compared to surface 529 application, but only significantly for  $CO_2$  emissions and not for overall  $GWP_{100}$ . 530 The practical implications of implementation are dependent on synchronising 531 measures with on-going management operations. Precise management of water table height is 532

533 highly restricted from a practical perspective, and cannot be expected across large-scale

534 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 535 harvesting operations due to economic pressures. In contrast, restricting horticultural fleece 536 use to the start of the season should pose few practical difficulties as the practice already 537 aligns with current management. Our results suggest that no one single mitigation measure 538 may be effective in reducing the rate of soil loss in cultivated peatlands. This has important 539 implications for the practicalities of co-implementing individual mitigation strategies, or in 540 considering more radical changes of land use and management in future. 541

542

#### 543 Acknowledgements

This work was funded by the European Union's Knowledge Economy Skills Scholarships
programme, in partnership with Footprints4Food and the Natural Environment Research
Council under the Soil Security Programme (Ref. NE/P014097/1). We would like to thank
Prof. Sue Page at the University of Leicester and Dr. Ed Moorhouse and Martin Hammond at
G's Fresh for their scientific and technical input to the research.

549

#### 550 **References**

Bussell, J., Jones, D.L., Healey, J.R., Pullin, A.S., 2010. How do draining and re-wetting
affect carbon stores and greenhouse gas fluxes in peatland soils? CEE Review 08-012.
Centre for Evidence-Based Conservation, Bangor University, UK. Available at:
<u>http://www.environmentalevidence.org/wp-content/uploads/2014/09/SR49.pdf</u>
(accessed March 2018).
Cannell, M.G.R., Milne, R., Hargreaves, K.J., Brown, T.A.W., Cruickshank, M.M., Bradley,
R.I., Spencer, T., Hope, D., Billett, M.F., Adger, W.N., Subak, S., 1999. National

- inventories of terrestrial carbon sources and sinks: The UK experience. Clim. Change
- 559 42, 505-530. doi: https://doi.org/10.1023/A:1005425807434
- 560 Dalal, R.C., Wang, W.J., Robertson, G.P., Parton, W.J., 2003. Nitrous oxide emission from
- 561 Australian agricultural lands and mitigation options: A review. Aust. J. Soil Res. 41,
- 562 165-195. doi: https://doi.org/10.1071/SR02064
- 563 Dalal, R.C., Allen, D.E., Livesley, S.J., Richards, G., 2008. Magnitude and biophysical
- regulators of methane emission and consumption in the Australian agricultural, forest,
- and submerged landscapes: A review. Plant Soil 309, 43-76. doi:
- 566 https://doi.org/10.1007/s11104-007-9446-7
- 567 Dawson, J.J.C., Smith, P., 2007. Carbon losses from soil and its consequences for land-use
- 568 management. Sci. Total Environ. 382, 165-190.
- doi: https://doi.org/10.1016/j.scitotenv.2007.03.023.
- 570 Dawson, Q., Kechavarzi, C., Leeds-Harrison, P.B., Burton, R.G.O., 2010. Subsidence and
- 571 degradation of agricultural peatlands in the Fenlands of Norfolk, UK. Geoderma 154,
- 572 181-187. doi: https://doi.org/10.1016/j.geoderma.2009.09.017
- 573 Dinsmore, K.J., Skiba, U., Billett, M.F., Rees, R.M., 2009. Effect of water table on
- 574 greenhouse gas emissions from peatland mesocosms. Plant Soil 318, 229-242. doi:
- 575 https://doi.org/10.1007/s11104-008-9832-9
- 576 Eijkelkamp, 2009. Redox measuring equipment. Operating instructions. Eijkelkamp
- 577 Agrisearch Equipment, Giesbeek, The Netherlands.
- 578 Elder, J.W., Lal, R., 2008. Tillage effects on gaseous emissions from an intensively farmed
- 579 organic soil in North Central Ohio. Soil Tillage Res. 98, 45-55. doi:
- 580 https://doi.org/10.1016/j.still.2007.10.003

- 581 Estop-Aragonés, C., Blodau, C., 2012. Effects of experimental drying intensity and duration
- 582 on respiration and methane production recovery in fen peat incubations. Soil Biol.

583 Biochem. 47, 1-9. doi: https://doi.org/10.1016/j.soilbio.2011.12.008

- 584 FAO, 2006. World reference base for soil resources 2006: A framework for international
- classification, correlation and communication. World Soil Resources Reports 103. Food
- and Agriculture Organization of the United Nations, Rome, Italy. Available at:
- 587 <u>http://www.fao.org/3/a-a0510e.pdf</u> (accessed March 2018).
- 588 Field, A., 2005. Discovering Statistics Using SPSS, Second ed., Sage Publications Ltd.,
- 589 Thousand Oaks, CA, USA.
- 590 Flessa, H., Wild, U., Klemisch, M., Pfadenhauer, J., 1998. Nitrous oxide and methane fluxes
- from organic soils under agriculture. Eur. J. Soil Sci. 49, 327-335. doi:
- 592 http://doi.org/10.1046/j.1365-2389.1998.00156.x.
- 593 Freeman, C., Lock, M.A., Reynolds, B., 1993. Fluxes of CO<sub>2</sub>, CH<sub>4</sub>, and N<sub>2</sub>O from a Welsh
- 594peatland following simulation of water table draw-down: Potential feedback to climatic

change. Biogeochem. 19, 51-60. http://www.jstor.org/stable/1468507.

- Hamouz, K., Dvořák, P., Čepl, J., Pivec, J. 2005. The effect of polypropylene fleece covering
- 597 on the yield of early potatoes. Hortic. Sci. (Prague) 32, 56-59.
- Hamouz, K., Lachman, J., Dvořák, P., Trnková, E., 2006. Influence of non-woven fleece on
  the yield formation of early potatoes. Plant Soil Environ. 52, 289-294.
- Henault, C., Grossel, A., Mary, B., Roussel, M., Leonard, J., 2012. Nitrous oxide emission by
- agricultural soils: A review of spatial and temporal variability for mitigation.
- 602 Pedosphere 22, 426-433. doi: https://doi.org/10.1016/S1002-0160(12)60029-0.
- HDC, 2006. Initial report: The scope for bio-degradable crop covers in vegetable production.
- Report for the Horticultural Development Council, Project FV280, April 2006.

605 Available at:

- 606 <u>https://horticulture.ahdb.org.uk/sites/default/files/research\_papers/FV%20280%20interi</u>
- 607 <u>m%20report%20April%202006.pdf</u> (accessed May 2018).
- Henderson, S.L., Dandie, C.E., Patten, C.L., Zebarth, B.J., Burton, D.L., Trevors, J.T., Goyer,
- 609 C., 2010. Changes in denitrifier abundance, denitrification gene mRNA levels, nitrous
- 610 oxide emissions, and dentrification in anoxic soil microcosms amended with glucose
- and plant residues. Appl. Environ. Microbiol. 76, 2155-2164. doi:
- 612 https://doi.org/10.1128/AEM.02993-09.
- Hutchinson, J.N., 1980. The record of peat wastage in the East Anglian Fenlands at Holme
- 614 Post, 1848-1978 A. D. J. Ecol. 68, 229-249. http://www.jstor.org/stable/2259253.
- 615 IPCC, 2000. Good practice guidance and uncertainty management in national greenhouse gas
- 616 inventories. IPCC, Geneva, Switzerland. Available at:
- 617 <u>http://www.ipcc-nggip.iges.or.jp/public/gp/english/</u> (accessed March 2018).
- 618 IPS, 2008. Peatlands and Climate Change. International Peatland Society, Jyväskylä, Finland.
- 619 Available at:
- 620 http://www.peatsociety.org/sites/default/files/files/PeatlandsandClimateChangeBookIP
- 621 S2008.pdf (accessed March 2018).
- Jauhiainen, J., Page, S.E., Vasander, H., 2016. Greenhouse gas dynamics in degraded and
- restored tropical peatlands. Mires Peat 17, UNSP 06. doi:
- 624 https://doi.org/10.19189/MaP.2016.OMB.229.
- 625 Kasimir-Klemedtsson, Å., Klemedtsson, L., Berglund, K., Martikainen, P., Silvola, J.,
- 626 Oenema, O., 1997. Greenhouse gas emissions from farmer organic soils: A review. Soil
- 627 Use Manage. 13, 245-250. doi: https://doi.org/10.1111/j.1475-2743.1997.tb00595.x.
- 628 Kechavarzi, C., Dawson, Q., Leeds-Harrison, P.B., Szatyłowicz, J., Gnatowski, T., 2007.
- 629 Water-table management in lowland UK peat soils and its potential impact on CO<sub>2</sub>

emission. Soil Use Manage. 23, 359-367. doi: https://doi.org/10.1111/j.1475-

631 2743.2007.00125.x.

- 632 Kløve, B., Berglund, K., Berglund, O., Weldon, S., Maljanen, M., 2017. Future options for
- 633 cultivated Nordic peat soils: Can land management and rewetting control greenhouse
- 634 gas emissions? Environ. Sci. Policy 69, 85-93. doi:
- 635 https://doi.org/10.1016/j.envsci.2016.12.017.
- 636 Kuzyakov, Y., Friedel, J.K., Stahr, K., 2000. Review of mechanisms and quantification
- 637 of priming effects. Soil Biol. Biochem. 32, 1485–1498. doi: http://dx.doi.org/
- 638 10.1016/S0038-0717(00)00084-5.
- 639 Kuzyakov, Y., 2010. Priming effects: Interactions between living and dead organic
- 640 matter. Soil Biol. Biochem. 42, 1363–1371. doi: http://dx.doi.org/10.1016/j.
- 641 soilbio.2010.04.003.
- Le Mer, J., Roger, P., 2001. Production, oxidation, emission and consumption of methane by
- soils: A review. Eur. J. Soil Biol. 37, 25-50.
- 644 doi: https://doi.org/10.1016/S1164-5563(01)01067-6.
- Leifeld, J., Müller, M., Fuhrer, J., 2011. Peatland subsidence and carbon loss from drained
- temperate fens. Soil Use Manage. 27, 170-176.
- 647 doi: https://doi.org/10.1111/j.1475-2743.2011.00327.x.
- 648 Li, C. 2007. Quantifying greenhouse gas emissions from soils: Scientific basis and modeling
- approach. Soil Sci. Plant Nutr. 53, 344-352.
- 650 doi: https://doi.org/10.1111/j.1747-0765.2007.00133.x.
- Li, C., Frolking, S., Frolking, T.A., 1992. A model of nitrous oxide evolution from soil
- driven by rainfall events: 1. Model structure and sensitivity. J. Geophys. Res. 97, 9759-
- 653 9776. doi: https://doi.org/10.1029/92JD00509.

- Lloyd, C.R., 2006. Annual carbon balance of a managed wetland meadow in the Somerset
- 655 Levels, UK. Agric. For. Meteorol. 138, 168-179. doi:
- 656 https://doi.org/10.1016/j.agrformet.2006.04.005.
- 657 Loecke, T.D., Robertson, G.P., 2009. Soil resource heterogeneity in terms of litter
- aggregation promotes nitrous oxide fluxes and slows decomposition. Soil Biol.

659 Biochem. 41, 228-235. doi: https://doi.org/10.1016/j.soilbio.2008.10.017.

- 660 Maljanen, M., Martikainen, P.J., Aaltonen, H., Silvola, J., 2002. Short-term variation in
- fluxes carbon dioxide, nitrous oxide and methane in cultivated and forested organic
- boreal soils. Soil Biol. Biochem. 34, 577-584. doi: https://doi.org/10.1016/S0038-
- 663 0717(01)00213-9.
- Marella, V.S.S.R., Roberts, P., Hill, P.W., Jones, D.L., 2017. Different ways in which CO<sub>2</sub>
- can be released during the turnover of roots in soil. Biol. Fertil. Soils 53, 369-374. doi:
  https://doi.org/10.1007/s00374-017-1193-1.
- 667 Miranda, K.M., Espey, M.G., Wink, D.A., 2001. A rapid, simple spectrophotometric method
- for simultaneous detection of nitrate and nitrite. Nitric Oxide 5, 62-71. doi:
- 669 https://doi.org/10.1006/niox.2000.0319.
- 670 Mitsch, W.J., Gosselink, J.G., 2000. Wetlands. Third Edition. John Wiley & Sons, New
  671 York, USA.
- Mulvaney, R.L., 1996. Nitrogen inorganic forms. In: Methods of Soil Analysis, Part 3:
- 673 Chemical Methods (D.L. Sparks Ed.), pp. 1123-1184. Soil Science Society of America
  674 Madison, WI, USA.
- Murphy, J., Riley, J.P., 1962. A modified single solution method for the determination of
- 676 phosphate in natural waters. Anal. Chim. Acta 27, 31-36. doi:
- 677 https://doi.org/10.1016/S0003-2670(00)88444-5.

- Musarika, S., Atherton, C.E., Gomersall, T., Wells, M.J., Kaduk, J., Cumming, A.M.J., Page,
- 679 S.E., Oechel, W.C., Zona, D., 2017. Effect of water table management and elevated
- $CO_2$  on radish productivity and on  $CH_4$  and  $CO_2$  fluxes from peatlands converted to
- agriculture. Sci. Total Environ. 584, 665-672. doi:
- 682 https://doi.org/10.1016/j.scitotenv.2017.01.094.
- 683 Natural England, 2010. England's peatlands: carbon storage and greenhouse gases. Natural
- 684 England, Worcester, UK. Available at:
- 685 http://publications.naturalengland.org.uk/publication/30021 (accessed March 2018).
- Olle, M., Bender, I., 2010. The effect of non-woven fleece on the yield and production
- 687 characteristics of vegetables. Agraarteadus XXX(1): 24-29. Available at:
- 688 http://agrt.emu.ee/pdf/2010\_1\_olle.pdf (accessed May 2018).
- 689 Oomes, M.J.M., Kuikman, P.J., Jacobs, F.H.H., 1997. Nitrogen availability and uptake by
- grassland in mesocosms of two water levels and two water qualities. Plant Soil 192,
- 691 249-259. doi: https://doi.org/10.1023/A:1004261913464.
- Reicosky, D.C., Gesch, R.W., Wagner, S.W., Gilbert, R.A., Wente, C.D., Morris, D.R., 2008.
- Tillage and wind effects on soil CO<sub>2</sub> concentrations in muck soils. Soil Till. Res. 99,
- 694 221-231. doi: https://doi.org/10.1016/j.still.2008.02.006.
- Richardson, S.J., Smith, J., 1977. Peat wastage in the East Anglian fens. J. Soil Sci. 28, 485489. doi: https://doi.org/10.1111/j.1365-2389.1977.tb02256.x.
- 697 Siwek, P., Libik, A., Zawicka, I., 2012. The effect of biodegradable nonwovens in butterhead
- 698lettuce cultivation for early harvest. Folia Hortic. 24, 161-166. doi:
- 699 https://doi.org/10.2478/v10245-012-0020-2.
- Siwek, P., Libik, A., Kalisz, A., Zawicka, I., 2013. The effect of biodegradable nonwoven
- direct covers on yield and quality of winter leek. Folia Hortic. 25, 61-65. doi:
- 702 https://doi.org/10.2478/fhort-2013-0007.

- Smith, P., Martino, D., Cai, Z., Gwary, D., Janzen, H., Kumar, P., McCarl, B., Ogle, S.,
- 704 O'Mara, F., Rice, C., Scholes, B., Sirotenko, O., Howden, M., McAllister, T., Pan, G.,
- Romanenkov, V., Schneider, U., Towprayoon, S., Wattenbach, M., Smith, J., 2008.
- 706 Greenhouse gas mitigation in agriculture. Philos. Trans. R. Soc. B: Biol. Sci. 363, 789-
- 707 813. doi: https://doi.org/10.1098/rstb.2007.2184.
- 508 Stevens, P.A., Ormerod, S.J., Reynolds, B. 1997. Final report on the acid waters survey for
- 709 Wales: Volume I. Institute of Terrestrial Ecology (Natural Environment Research
- 710 Council) Project T07072R5 Final Report, July 1997.
- 711 Taft, H.E., Cross, P.A., Edwards-Jones, G., Moorhouse, E.R., Jones, D.L., 2017. Greenhouse
- 712 gas emissions from intensively managed peat soils in an arable production system.
- 713 Agric. Ecosyst. Environ. 237, 162-172. doi: https://doi.org/10.1016/j.agee.2016.11.015.
- 714 UK Parliament, 2008. Climate Change Act 2008. The Stationery Office Ltd., London, UK.
- 715 Available at:
- https://www.legislation.gov.uk/ukpga/2008/27/pdfs/ukpga\_20080027\_en.pdf (accessed
  March 2018).
- 718 UK Met Office. 2014. UK Climate and weather statistics. Available at:
- http://www.metoffice.gov.uk/climate/uk/summaries/actualmonthly (accessed March
  2014).
- Velthof, G.L., Kuikman, P.J., Oenema, O., 2002. Nitrous oxide emission from soils amended
  with crop residues. Nutr. Cycl. Agroecosyst. 62, 249-261. doi:
- 723 https://doi.org/10.1023/A:1021259107244.
- Velthof, G.L., Oenema, O., 1997. Nitrous oxide fluxes from grassland in the Netherlands: II.
- 725 Effects of soil type, nitrogen fertilizer application and grazing. Eur. J. Soil Sci. 46, 541-
- 726 549. doi: https://doi.org/10.1111/j.1365-2389.1995.tb01350.x.

- 727 Weslien, P., Rütting, T., Kasimir-Klemedtsson, Å., Klemedtsson, L., 2012. Carrot cropping
- on organic soil is a hotspot for nitrous oxide emissions. Nutr. Cycl. Agroecosyst. 94,
- 729 249-253. doi: https://doi.org/10.1007/s10705-012-9538-5.

**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<sub>15</sub>), water table at soil surface (WT<sub>0</sub>), 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 <sup>†</sup> for non-parametric (Kolmogorov-Smirnov Z statistic, Bonferroni corrected).

Treatment	Depth	Soil moisture	Bulk density	pН	EC	Available K	Available P	Available NO <sub>3</sub> <sup>-</sup>	Available $NH_4^+$
	(cm)	(% DW)	$(g \text{ cm}^{-3})$	$(H_2O)^a$	$(\mu S \text{ cm}^{-1})^{s}$	(g K kg <sup>-1</sup> )	$(g P kg^{-1})$	(g N kg <sup>-1</sup> )	$(g N kg^{-1})$
Initial	0-10	$152 \pm 1$	$0.68 \pm 0.01$	$6.2\pm0.08$	$598 \pm 50$	$0.96 \pm 0.21$	$0.39 \pm 0.01$	$0.15\pm0.016$	$0.05 \pm 0.024$
	10-20	$156 \pm 2$	$0.76\pm0.02$	$6.2\pm0.06$	$552 \pm 49$	$0.63\pm0.11$	$0.38 \pm 0.01$	$0.15\pm0.033$	$0.04\pm0.008$
	20-30	$163 \pm 5$	$0.75 \pm 0.02$	$6.3\pm0.06$	$401 \pm 24$	$0.56 \pm 0.11$	$0.35 \pm 0.02$	$0.13 \pm 0.033$	$0.03 \pm 0.001$
Post-experi	iment								
Control	0-10	$164 \pm 1^{\dagger}$	$0.73 \pm 0.01*$	$6.7\pm0.04^{\dagger}$	$161 \pm 13$	$0.54 \pm 0.08$	$0.27\pm0.02^{\dagger}$	$0.01\pm0.001^\dagger$	< 0.01
	10-20	$168 \pm 2^{**}$	$0.77\pm0.01$	$6.7 \pm 0.06^{***}$	$166 \pm 8$	$0.51 \pm 0.19$	$0.27 \pm 0.01$ **	$0.03\pm0.004^\dagger$	< 0.01
	20-30	$180 \pm 2$	$0.75 \pm 0.01$	$6.7 \pm 0.04*$	$220 \pm 9^{***}$	$0.58 \pm 0.15$	$0.21 \pm 0.04$	$0.06\pm0.008$	< 0.01
WT <sub>15</sub>	0-10	$170 \pm 1^{\dagger}$	$0.74 \pm 0.01$ **	$6.7\pm0.04^{\dagger}$	$136 \pm 3$	$0.63\pm0.08$	$0.29\pm0.02^{\dagger}$	$0.01\pm0.001^\dagger$	< 0.01
	10-20	$171 \pm 2^{***}$	$0.78\pm0.01$	$6.7 \pm 0.03^{***}$	$160 \pm 6$	$0.50 \pm 0.13$	$0.31\pm0.02$	$0.02\pm0.001^\dagger$	< 0.01
	20-30	$175 \pm 6$	$0.75\pm0.01$	$6.7 \pm 0.03*$	$223 \pm 11^{***}$	$0.44 \pm 0.10$	$0.26\pm0.04$	$0.03\pm0.006$	< 0.01
$WT_0$	0-10	$172\pm1^{\dagger}$	$0.74 \pm 0.01 **$	$6.7\pm0.03^{\dagger}$	$159 \pm 8$	$0.61 \pm 0.16$	$0.27\pm0.01^{\dagger}$	$0.01\pm0.001^\dagger$	< 0.01
	10-20	$169 \pm 3^{**}$	$0.78\pm0.02$	$6.8 \pm 0.07 ***$	$176 \pm 17$	$0.62 \pm 0.16$	$0.27 \pm 0.01$ **	$0.02\pm0.001^\dagger$	< 0.01
	20-30	$174 \pm 5$	$0.77\pm0.01$	$6.7 \pm 0.06^{**}$	196 ± 16***	$0.49\pm0.17$	$0.33\pm0.04$	$0.02\pm0.003^\dagger$	< 0.01
C <sub>fleece</sub>	0-10	$161\pm2^{\dagger}$	$0.73\pm0.01$	$6.6\pm0.05^{\dagger}$	$154 \pm 9^{\dagger}$	$0.42\pm0.07$	$0.35\pm0.03$	$0.01\pm0.001$	< 0.01
	10-20	$166 \pm 3^{*}$	$0.76 \pm 0.01$	$6.4 \pm 0.05*$	$205 \pm 20^{\dagger}$	$0.45\pm0.12$	$0.31 \pm 0.01$	$0.04\pm0.006$	< 0.01
	20-30	$175 \pm 5$	$0.76\pm0.01$	$6.4\pm0.05$	$321 \pm 10^{**}$	$0.42\pm0.11$	$0.31\pm0.02$	$0.10\pm0.003$	< 0.01
$\mathbf{S}_{\text{till}}$	0-10	$158 \pm 2$	$0.62 \pm 0.01^{***}$	$6.7 \pm 0.08$	$133 \pm 13^{+}$	$0.49\pm0.08$	$0.31 \pm 0.01^{\dagger}$	$0.01 \pm 0.001^{\dagger}$	< 0.01
	10-20	$166 \pm 2$	$0.65 \pm 0.02^{***}$	$6.6 \pm 0.07 ***$	$140\pm7^{\dagger}$	$0.55\pm0.09$	$0.30 \pm 0.03$	$0.02 \pm 0.002^{\dagger}$	< 0.01
	20-30	$175 \pm 2$	$0.69\pm0.02$	$6.5 \pm 0.08$	184 ± 13***	$0.61\pm0.14$	$0.33 \pm 0.02$	$0.04\pm0.006$	< 0.01
		±			Y .			±	
CR <sub>surf</sub>	0-10	$164 \pm 2^{+}$	$0.76 \pm 0.02^{***}$	$6.7 \pm 0.03^{\circ}$	$139 \pm 2^{+}$	$0.59 \pm 0.03$	$0.30 \pm 0.02$	$0.01 \pm 0.001^{+}$	< 0.01
	10-20	$164 \pm 1$	$0.76 \pm 0.01$	$6.7 \pm 0.04^{***}$	$149 \pm 6^{\dagger}$	$0.49 \pm 0.10$	$0.32 \pm 0.01$	$0.02 \pm 0.001^{\dagger}$	< 0.01
	20-30	$165 \pm 5$	$0.76 \pm 0.01$	$6.5 \pm 0.08$	$178 \pm 4^{***}$	$0.42 \pm 0.13$	$0.29 \pm 0.04$	$0.03 \pm 0.003$	< 0.01
CR <sub>incorp</sub>	0-10	$160 \pm 2$	$0.59 \pm 0.01^{***}$	$6.6 \pm 0.12$	$142 \pm 12$	$0.48 \pm 0.11$	$0.30 \pm 0.02$	$0.01 \pm 0.002$	< 0.01
	10-20	$170 \pm 2^{***}$	$0.65 \pm 0.01^{***}$	$6.7 \pm 0.08^{***}$	$159 \pm 3$	$0.62\pm0.16$	$0.35\pm0.02$	$0.02\pm0.001$	< 0.01
	20-30	$178 \pm 2$	$0.71 \pm 0.01$	$6.6 \pm 0.13$	$184 \pm 10^{***}$	$0.49 \pm 0.17$	$0.34 \pm 0.03$	$0.04\pm0.008$	< 0.01

<sup>a</sup> 1:2.5 (w/v) field moist soil:distilled H<sub>2</sub>O.

Treatment and code	Water table depth Lettuce biomass		Cultivation	Soil cover
	(cm)	$(g FW cm^{-2} / t FW ha^{-1})^{a}$	(cm)	
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 <sub>fleece</sub> )	>30 cm (free-draining)	None	None	Fleece
Soil tillage (S <sub>till</sub> )	>30 cm (free-draining)	None	To 30 cm depth	None
Crop residue, surface applied (CR <sub>surf</sub> )	>30 cm (free-draining)	$35.5 \text{ g cm}^{-2} / 29.7 \text{ t ha}^{-1}$	None	Crop residue
Crop residue, incorporated (CR <sub>incorp</sub> )	>30 cm (free-draining)	$35.5 \text{ g cm}^{-2} / 29.7 \text{ t ha}^{-1}$	To 30 cm depth	None

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

<sup>a</sup> FW, fresh weight.

**Table 3.** Cumulative fluxes of CO<sub>2</sub>, N<sub>2</sub>O and CH<sub>4</sub>, and total cumulative GHG emissions (*GWP*<sub>100</sub>) in t CO<sub>2</sub>-e ha<sup>-1</sup> period<sup>-1</sup> ( $\pm$  SEM), for control, water table at -15 cm below soil surface (WT<sub>15</sub>), water table at soil surface (WT<sub>0</sub>), fleece cover (C<sub>fleece</sub>), cultivated (S<sub>till</sub>), surface applied crop residue (CR<sub>surf</sub>), and incorporated crop residue (CR<sub>incorp</sub>) 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  $\pm$  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<sub>15</sub> and WT<sub>0</sub> (denoted a-c), (2) Control and C<sub>fleece</sub> (denoted d-e), (3) Control and S<sub>till</sub> (ns), (4) Control and CR<sub>surf</sub> (denoted f-g), (5) Control and CR<sub>incorp</sub> (ns), and CR<sub>surf</sub> and CR<sub>incorp</sub> (denoted h-i).

Treatment		Pha		Phase I + II								
	t CO <sub>2</sub> -e ha <sup>-1</sup> 80 d <sup>-1</sup>					t $CO_2$ -e ha <sup>-1</sup> 69 d <sup>-1</sup>			t $CO_2$ -e ha <sup>-1</sup> 153 d <sup>-1</sup>			
	CO <sub>2</sub>	N <sub>2</sub> O	$CH_4$	$GWP_{100}$	CO <sub>2</sub>	N <sub>2</sub> O	$CH_4$	$GWP_{100}$	CO <sub>2</sub>	$N_2O$	$CH_4$	$GWP_{100}$
Control	5.87 ± 0.06 a,d,f	0.55 ± 0.10 a,f	$0.00\pm0.01$	6.43 ± 0.11 a,d,f	$4.09 \pm 0.29$ a	$0.71\pm0.25$	$0.01\pm0.01$	$4.81\pm0.31$	$10.29 \pm 0.35$ a	$1.36 \pm 0.37$ a	$0.01\pm0.01$	$11.66 \pm 0.42$ a
WT15	$5.72\pm0.22$ ab	$7.70\pm0.92\ b$	$\textbf{-0.00} \pm 0.01$	$13.41\pm0.90~b$	$4.58\pm0.11\ ab$	$0.74\pm0.12$	$0.00\pm0.02$	$5.32\pm0.20$	$10.61 \pm 0.30$ a	$8.82\pm1.11\ b$	$0.00\pm0.02$	$19.42\pm1.14~b$
$WT_0$	$0.85\pm0.12~b$	$1.16 \pm 0.37$ a	$\textbf{-0.00} \pm 0.01$	$2.01 \pm 0.45 \text{ c}$	$5.30\pm0.23\ b$	$0.44\pm0.21$	$0.01\pm0.01$	$5.75\pm0.37$	$6.47\pm0.20~b$	$1.71 \pm 0.43$ a	$0.01\pm0.01$	$8.19\pm0.58~c$
Still	$5.63\pm0.22$	$0.50\pm0.10$	$0.01\pm0.00$	$6.14 \pm 0.27$								
C <sub>fleece</sub>	$7.83\pm0.58\;e$	$1.20\pm0.25$	$0.03\pm0.04$	$9.07 \pm 0.58 \text{ e}$								
CR <sub>surf</sub>	$7.07\pm0.26~g\text{,h}$	$1.42 \pm 0.29 \text{ g}$	$\textbf{-0.05} \pm 0.02$	$8.44 \pm 0.30$ g								
CR <sub>incorp</sub>	$5.99\pm0.18\ i$	$0.78\pm0.22$	$0.01\pm0.01$	$6.79\pm0.34$								

**Table 4.** Significant linear correlations between measured environmental variables and emissions of CO<sub>2</sub>, N<sub>2</sub>O and CH<sub>4</sub> for control, water table at -15 cm below soil surface (WT<sub>15</sub>), water table at soil surface (WT<sub>0</sub>), fleece cover (C<sub>fleece</sub>), cultivated (S<sub>till</sub>), surface applied crop residue (CR<sub>surf</sub>), and incorporated crop residue (CR<sub>incorp</sub>) 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 ( $\tau$ ), with significance levels presented as \* (p < 0.05), \*\* (p < 0.01), or \*\*\* (p < 0.001).

Treatment		Soil redox potential, $E_{\rm h}$ (mV)			Temperature			Rai	Rainfall		Nitrogen availability		
			Soil de	epth (cm)		Soil temp. <sup>a</sup>	Mean air temp. b	Air temp. <sup>c</sup>	Daily rain <sup>d</sup>	5 d rain <sup>e</sup>	NO <sub>3</sub> -N	NH <sub>4</sub> -N	Ν
		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 <sub>15</sub> , 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 <sub>15</sub> , 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 <sub>15</sub> , 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 <sub>fleece</sub>					0.539***	0.595***	0.365***		-0.153*			
	Still					0.341***	0.392***	0.365***			0.243**	0.255**	
	CR <sub>surf</sub>						0.230**		0.180*				0.216**
	CRincorp						0.166*		-0.112*		0.219*		
N <sub>2</sub> O	Control, wetted								-0.212**		0.185*		
-	WT <sub>15</sub> , wetted					0.180*			-0.579***	-0.260***			
	$WT_0$ , wetted								-0.357***				0.207*
	Control, drained												
	WT <sub>15</sub> , 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						Y		-0.136*				
	WT <sub>15</sub> , whole period					0.313***		0.204***	-0.440***	-0.199***	0.347***		0.241***
	$WT_0$ , whole period					0.153**		0.168**	-0.291***				
	Cflama					0 147*			-0 237**				
	S					0.1.17			-0.240**				
							-0.185*	-0.171*	-0.240			-0.292**	
	CR						-0.171*	0.171	-0.407***			0.272	
CH	Control wetted						-0.171		-0.407				
C114	WT., wetted												
	WT, wetted												
	Control drained					-0.170*	-0.164*	-0.179*					
	WT., drained					-0.170	-0.104	-0.177					
	WT <sub>2</sub> , drained												
	Control whole period	0, uranicu											
	W <sub>1</sub> , whole period												
	WT, whole period												
	$C_{a}$										0.179*	-0 239**	
	C tieece										0.179	-0.239	
	CP .					0.461*		0 100**					
	CR				Y	-0.401		-0.177					

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



**Fig. 1.** Daily rainfall, air temperature and soil temperature (a-b); fluxes of CO<sub>2</sub> (c-d), N<sub>2</sub>O (e-f), and CH<sub>4</sub> (g-h); and soil water NO<sub>3</sub><sup>-</sup> (i-j) and NH<sub>4</sub><sup>+</sup> (k-l); 28<sup>th</sup> May to 16<sup>th</sup> Aug. (Phase I, wetted) and 21<sup>st</sup> Aug. to 13<sup>th</sup> 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<sub>15</sub>), and white circles (water table at the soil surface, WT<sub>0</sub>). In panels (c)-(l), the control treatment is denoted by black circles with a solid line, WT<sub>15</sub> by grey circles with a dashed line, and WT<sub>0</sub> by white circles with a dotted line. Error bars represent  $\pm$  SEM.



**Fig. 2.** Redox potentials ( $E_h$ ) at soil depths of 0 cm (a), 10 cm (b), 20 cm (c), and 30 cm (d); 21<sup>st</sup> Aug. to 13<sup>th</sup> Nov. 2013 (Phase II, drained). The free-draining control treatment is denoted by black circles with a solid line, WT<sub>15</sub> (water table at 15 cm below the soil surface) by grey circles with a dashed line, and WT<sub>0</sub> (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); 28<sup>th</sup> May to 16<sup>th</sup> 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.



**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); 28<sup>th</sup> May to 16<sup>th</sup> 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  $\pm$  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.

CEP HER MAR