Management Effects on Greenhouse Gas Dynamics in Fen Ditches

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Abstract

Globally, large areas of peatland have been drained through the digging of ditches, generally to increase agricultural production. By lowering the water table it is often assumed that drainage reduces landscape-scale emissions of methane (CH_4) into the atmosphere to negligible levels. However, drainage ditches themselves are known to be sources of CH_4 and other greenhouse gases (GHGs), but emissions data are scarce, particularly for carbon dioxide (CO_2) and nitrous oxide (N_2O), and show high spatial and temporal variability. Here, we report dissolved GHGs and diffusive fluxes of CH_4 and CO_2 from ditches at three UK lowland fens under different management; semi-natural fen, cropland, and cropland restored to low-intensity grassland. Ditches at all three fens emitted GHGs to the atmosphere, but both fluxes and dissolved GHGs showed extensive variation both seasonally and within-site. CH_4 fluxes were particularly large, with medians peaking at all three sites in August at 120-230 mg m^{-2} d^{-1}. Significant between site differences were detected between the cropland and the other two sites for CO_2 flux and all three dissolved GHGs, suggested that intensive agriculture has major effects on ditch biogeochemistry. Multiple regression models using environmental and water chemistry data were able to explain 29-59% of observed variation in dissolved GHGs. Annual CH_4 fluxes from the ditches were 37.8, 18.3 and 27.2 g CH_4 m^{-2} yr^{-1} for the semi-natural, grassland and cropland, and annual CO_2 fluxes were similar (1100 to 1440 g CO_2 m^{-2} yr^{-1}) among sites. We suggest that fen ditches are important contributors to landscape-scale GHG emissions, particularly for CH_4. Ditch emissions should be included in GHG budgets of human modified fens, particularly where drainage has removed the original terrestrial CH_4 source, e.g. agricultural peatlands.

Keywords: peatland, carbon dioxide, methane, nitrous oxide, ditch flux, restoration

1. Introduction

Northern peatlands store approximately 547 Pg of carbon (Yu *et al.*, 2010) and contribute to the global atmospheric balance of GHGs through the release and uptake of carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O). Intact peatlands are typically net sinks for CO_2, and sources of CH_4 and N_2O (Freeman *et al*., 1993, Nykänen *et al*., 1995, Smith *et al*., 2004, Kirschke *et al*., 2013). On a 100-year timescale CH_4 and N_2O have global warming potentials (GWP) of 28 and 298, respectively, relative to CO_2 (IPCC, 2013). Insights into biogeochemical cycling in peatlands are therefore important in developing understanding of global GHG dynamics and future climate change.

Globally, peatlands have been extensively drained for conversion to agriculture, forestry and peat extraction. Drained lowland fens, such as those of Eastern England, the Netherlands and the Southern Baltic coast are extremely fertile, and are therefore principally converted to intensive agricultural use (Morris *et al*., 2000). Conversion to agricultural use often includes strict hydrological management, such as the use of subsurface irrigation and, in part due to the long-term subsidence which is an inevitable consequence of peat drainage, the active pumping of water around fields (e.g. Morrison *et al*., 2013). There is now growing interest in the restoration of agricultural fens to wetlands (e.g. Höll *et al*., 2009, Peh *et al*., 2014), although there are strong commercial factors, as well as food security considerations, that favour their continued agricultural use (Glenk *et al*., 2014).

Drainage and conversion of fens to agricultural use has the capacity to alter the cycling of GHGs. It is generally considered that peatland drainage leads to a decrease in CH_4 emissions (to near-zero values), but increases in CO_2 and N_2O emissions (Glenn *et al*., 1993, Martikainen *et al*., 1995, Alm *et al.,* 1999, Haddaway *et al*., 2014). Upon draining, peatlands therefore become a diminishing carbon reservoir, releasing carbon into the atmosphere that was fixed over thousands of years.

CH_4 fluxes from drained peatlands were previously assumed to be insignificant (IPCC, 2006). However, a number of studies have shown that the ditches created during drainage can themselves be significant CH_4 sources (Best & Jacobs, 1997, Sundh *et al.,* 2000,

Minkkinen & Laine, 2006, Hendriks *et al.*, 2007, Hyvönen *et al*., 2013), contributing 60-70% of total CH_4 emissions in one study (Schrier-Uijl *et al.*, 2010), over 84% in another (Teh *et al.*, 2011) and with measured fluxes as high as 366 mg CH_4 m^{-2} hr^{-1} (Schrier-Uijl *et al.*, 2010). Where the space between ditches is small, drainage could in theory actually result in a net increase in landscape-scale CH_4 fluxes compared to undrained sites (Roulet & Moore, 1995).

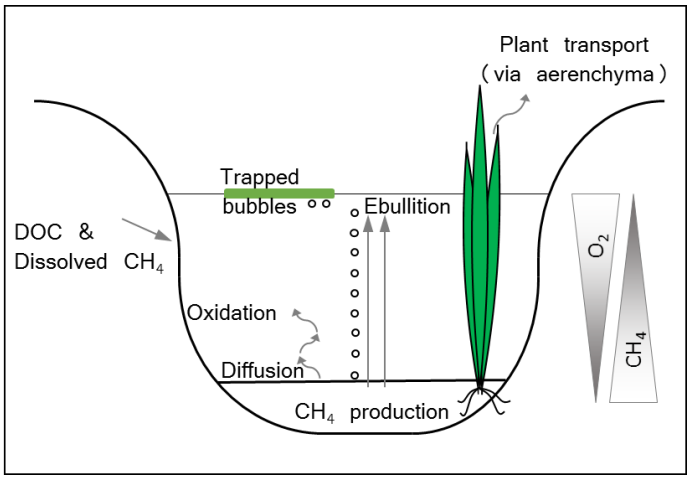


Figure 1. Schematic of methane transport pathways within ditch systems and surrounding peat.

Large ditch CH_4 fluxes are usually associated with productive, high-nutrient, sites with low water flow and high labile carbon inputs (e.g. agricultural grasslands; Best & Jacobs, 1997). Conversely, faster-flowing ditches in nutrient-poor upland bogs typically have small fluxes; Cooper *et al*. (2014) recorded an annual mean CH_4 flux of 59.7 kg CH_4 ha^{-2} y^{-1} from an open ditch in a blanket bog, and Sirin *et al*. (2012) measured a growing season flux of 9.9 mg CH_4 m^2 d^{-1} from ditches in a forested bog. A recent review found mean fluxes for different peat/land-use types varied from approximately 30 g CH_4 m^{-2} yr^{-1} in forest/semi-natural peatlands, to 200 g CH_4 m^{-2} yr^{-1} in tropical deforested peatlands (Evans et al., 2016a). It is important to recognise that methane emissions can occur via different pathways, and the rates of flux via these pathways will have different controls (fig.1). Diffusive/steady emissions result from the CH_4 concentration differential between the ditch and the atmosphere. Wetland plant aerenchyma may provide a chimney through which oxygen is transported into sediment and CH_4 escapes to the atmosphere. Finally, steady emissions may be punctuated by temporally and spatially heterogeneous ebullition, which can contribute significantly to net CH_4 fluxes (Vermaat *et al*., 2011). The importance of ditches in GHG cycling has therefore been recognised by the IPCC and incorporated into their guidelines (IPCC, 2014).

As well as CH_4, drainage ditches emit N_2O (Reay *et al*., 2003, Teh *et al.*, 2011, Hyvönen *et al*., 2013). Some ditches have been found to emit CO_2 (Best & Jacobs, 1997, Sundh *et al*., 2000, Teh *et al.*, 2011, Hyvönen *et al*., 2013), which others with emergent vegetation have sometimes been observed to fix CO_2 (e.g. Vermaat *et al*., 2011). However, whilst ditches appear to be consistent hotspots for CH_4 emissions, CO_2 and N_2O fluxes are of a considerably smaller magnitude in terms of their overall contribution to GHG emissions, and are typically more similar to fluxes from drained peat adjacent to ditches (Evans *et al*., 2016a). For example, Hyvönen *et al*. (2013) found ditches in a boreal cutaway peatland being used to cultivate *Phalaris arundinacea* contributed just 1% and 5% of total ecosystem emission of N_2O and CO_2.

Internationally, there is a lack of information on GHG emissions from drainage ditches; in a recent review of published studies, a total of just 19 studies were identified in which peatland CH_4 emissions had been reported, for a total of 69 individual peatland sites where CH_4 was measured (Evans et al., 2016a). The same analysis suggested that studies of CO_2 and N_2O are still too few to allow the data to be collated in a meaningful way. Just two studies to date have reported CH_4 fluxes from ditches in the UK. In contrast to this dearth of information on ditches, numerous studies have looked at GHG emissions associated with other freshwaters. For instance, Cole *et al*. (2007) noted that carbon emissions from lakes and rivers could be approximately 0.8 Pg C y^{-1}; enough to exert effects on regional budgets, despite these features occupying small areas. Similarly, Bastviken *et al*. (2011) suggested that CH_4 emissions from inland waters have the capacity to offset 25% of the terrestrial carbon sink, whilst Deemer et al. (2016) calculate that reservoirs emit 1.5% of global anthropogenic CO_2-equivalent emissions from CO_2, CH_4 and N_2O. Considering N_2O, rivers and estuaries could account for 20% of global anthropogenic emissions (Seitzinger & Kroeze, 1998).

To help address this knowledge gap, we carried out seasonal fieldwork for one year in ditches at three lowland fens in East Anglia, England. Each site was under a different management regime: 1) a semi-natural fen under conservation management; 2) former cropland that has been restored to extensive grassland, and; 3) intensive deep-drained cropland. We measured dissolved GHGs within ditches, diffusive fluxes of CO_2 and CH_4 from ditches, and a variety of physical ditch attributes and water chemistry determinands. Our

aim was to quantify the differences in GHGs between and within sites, and across seasons, and to attempt to elucidate the drivers behind GHG dynamics.

2. Materials and methods

2.1. Field sites

All three field sites were located in East Anglia, in Eastern England. This region was once the largest area of lowland fen peatland in the UK, covering several thousand square kilometres. Since the 17th century, drainage of the land resulted in the loss of most of the natural fenland, with only a handful of intact fragments remaining. The principal land use of the drained areas is intensive arable and horticultural agriculture. The drainage and conversion of the fens has resulted in extensive peat wastage, with much of the original deep peat area now reduced to a dense, thin intermixed organic and mineral layer (Hutchinson, 1980, Burton and Hodgson, 1987). The altitude of the land is close to (and in many areas below) sea level. Mean annual rainfall in the area is 574 mm, and mean annual temperature is 10.1 °C (data from UK Met Office station in Mepal, within 30 km of all study sites). The sites were:

1. Sedge Fen (semi-natural fen). 52.31 N, 0.28 E. Area = 61 ha. Sedge Fen is part of the Wicken Fen National Nature Reserve. Peat depth is 3.8 m, bulk density is 0.37 g cm^{-3}, C/N is 15.8 (Evans *et al*., 2016b). Vegetation comprises reedbeds dominated by *Cladium mariscus* and *Phragmites australis*, with some *Phalaris arundinacea* and *Calamagrostis canescens* (Eades, 2016), as well as areas of fen carr dominated by *Rhamnus cathartica* and *Frangula alnus* (Rowell, 1986). The fen cannot be considered to be 'pristine' as it contains numerous internal ditches, and the reedbeds are cut on a three year rotation. However, the site contains vegetation and peat that is characteristic of an intact site, and has never been converted to other land-uses.

2. Baker's Fen (extensive grassland). 52.30 N, 0.29 E. Area = 56 ha. Baker's Fen is part of the wider Wicken Fen area. Historically, the fen was drained and used for arable agriculture, resulting in extensive peat wastage and loss of organic soil. Soil depth is now less than 50 cm, bulk density is 1.06 g cm^{-3}, C/N is 19.7 (Evans *et al*., 2016b), and organic content is low (measured as 13-18 % loss on ignition by Stroh *et al*., 2013). The site was removed from

arable use and re-seeded with an unknown "grass mixture" in 1995 and 1996, and is undergoing "open-ended" restoration (Hughes *et al*., 2011); river water is pumped onto the site in autumn and winter to inundate it, and highland cattle and wild horses graze it. Much of the fen consists of species-poor, flood-plain pasture. Plant species vary across the site according to variations in hydrology and nutrient status, but include *Carex otrubae, Arrhenatherum elatius, Agrostis stolonifera, Cirsium arvense*, *Poa trivialis* and several *Juncus* species (Eades, 2016). *C. mariscus* and *P. australis* occur in some of the ditches.

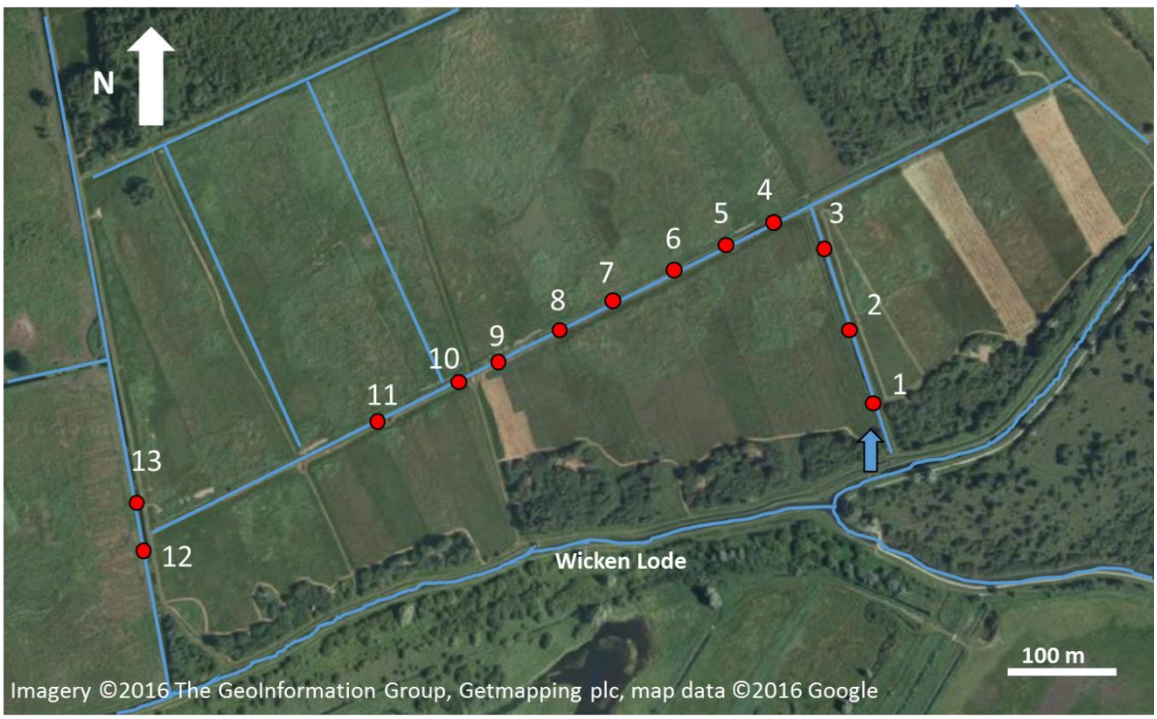
3. Rosedene (cropland). 52.52 N, 0.49 E. Field area = 8.7 ha. The cropland site consists of ditches that surround a field near Methwold Hythe. The field is part of a much larger area (~ 90 km^2) of drained fen that is now under intensive arable cultivation, and is bounded by rivers and canals. Peat depth is 1 m, bulk density is 0.32 g cm^{-3}, and C/N is 15 (Evans *et al*., 2016b). The hydrology of the site is highly managed; the fields contain subsurface pipes at 1 m depth to aid irrigation and drainage, and water is actively pumped round field perimeter ditches in order to maintain water levels within the field, removing water during wet periods and providing irrigation water during dry periods. During 2015, the study site was used to cultivate celery (*Apium graveolens*). This site is 28 km from the other two sites.

All three sites formed part of a larger study of GHG emissions from a total of fifteen lowland peatland sites at located across six regions of England and Wales, which included a broad suite of eddy covariance and static chamber gas flux, hydrological and water quality measurements. The results of this large-scale study are reported elsewhere (Evans et al., 2016b).

2.2. Sampling

The sites were visited on four occasions in 2015 in March, May, August and October. Because of proximity, the intact and restored site could be visited on the same day, whilst the agricultural site was visited within three days. The sampling dates were as follows: 11th March – semi-natural/grassland, 12th March – cropland; 5th May – semi-natural/grassland, 6th May – cropland; 17th August – cropland, 20th August – semi-natural/grassland; 12th October – cropland, 15th October – semi-natural/grassland. At each site, ditch sampling locations were selected with the aim of covering a large area, and were selected on a non-

185 random basis according to where measurements from the ditch could easily be taken. For
186 the semi-natural site, we sampled along a 910 m length of ditch network (i.e. all sampling
187 points were hydrologically connected) and then onto a ditch that bounded the edge of the
188 fen (fig. 2). Similarly, all ditch locations at the grassland site were hydrologically connected,
189 with a ditch distance of 1200 m between farthest sampling points. At the cropland site the
190 ditch ran continuously round a field, with junctions connecting to other ditches at field
191 corners. The sampling locations here ran for 1200 m. The number of sampling locations for
192 each site was: semi-natural = 13, grassland = 11, cropland = 10. The same sampling
193 locations were used for each of the four seasonal visits.
194 A range of measurements were taken at each sampling location. Environmental and
195 physical measurements were: air temperature, water temperature, atmospheric pressure,
196 and water depth. A 50 ml water sample for water chemistry analysis was collected in a
197 polypropylene vial. A sample for dissolved GHG analysis was collected using the headspace
198 method (Hope *et al*., 2004); 30 ml of ditch water was collected in a 60 ml plastic syringe and
199 equilibrated with 30 ml of ambient air by shaking for approximately 60 seconds, and 12 ml
200 of headspace was then collected in a 12 ml borosilicate glass vial. Fluxes of CH_4 and CO_2
201 were measured in real time in the field using a floating chamber (0.6 x 0.6 x 0.3 m) that was
202 shrouded to exclude light. Buoyancy for the chamber was provided by two 2 l plastic bottles
203 filled with air, and the chamber was placed carefully on the water to minimise disturbance.
204 Emergent vegetation was excluded (e.g. *P. australis*), but some sampling points contained
205 floating algae that will have contributed to fluxes. The chamber was connected to a Los
206 Gatos Ultraportable Greenhouse Gas Analyzer. The chamber was deployed until a linear
207 flux was observed, and this was typically 1-5 minutes. Whilst there has been some criticism
208 of the use of floating chambers, flow rates in the ditches we studied where either extremely
209 low or absent (i.e. chambers did not drift away) and therefore our measurements are likely
210 to be robust (see Lorke *et al*., 2015).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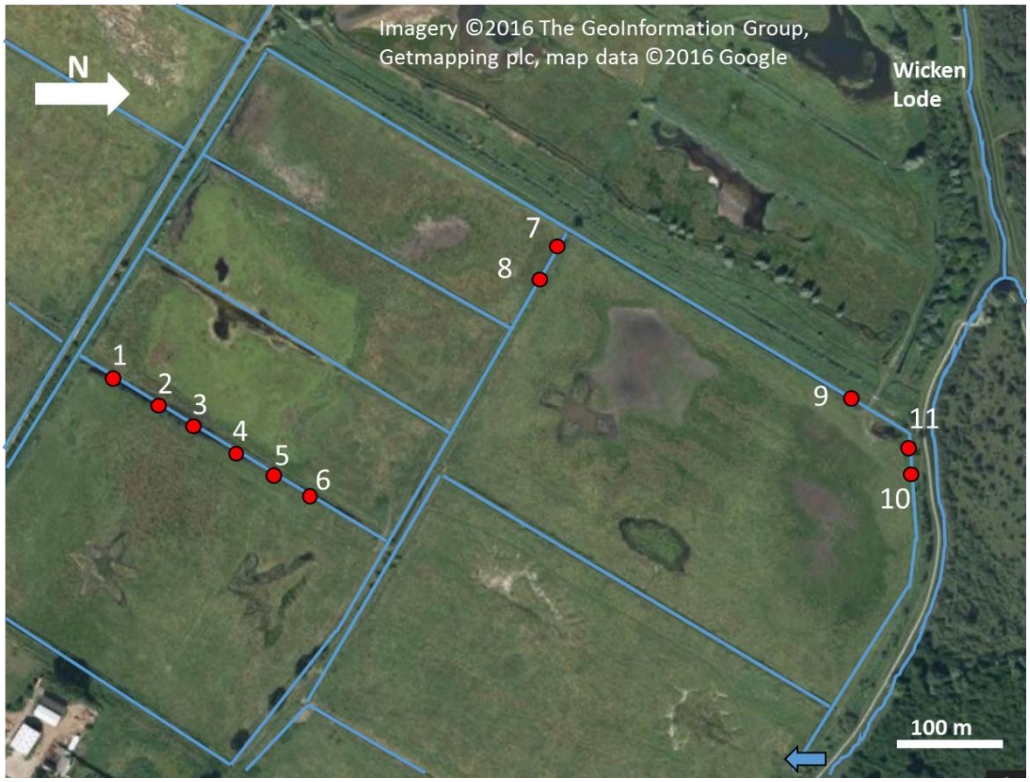
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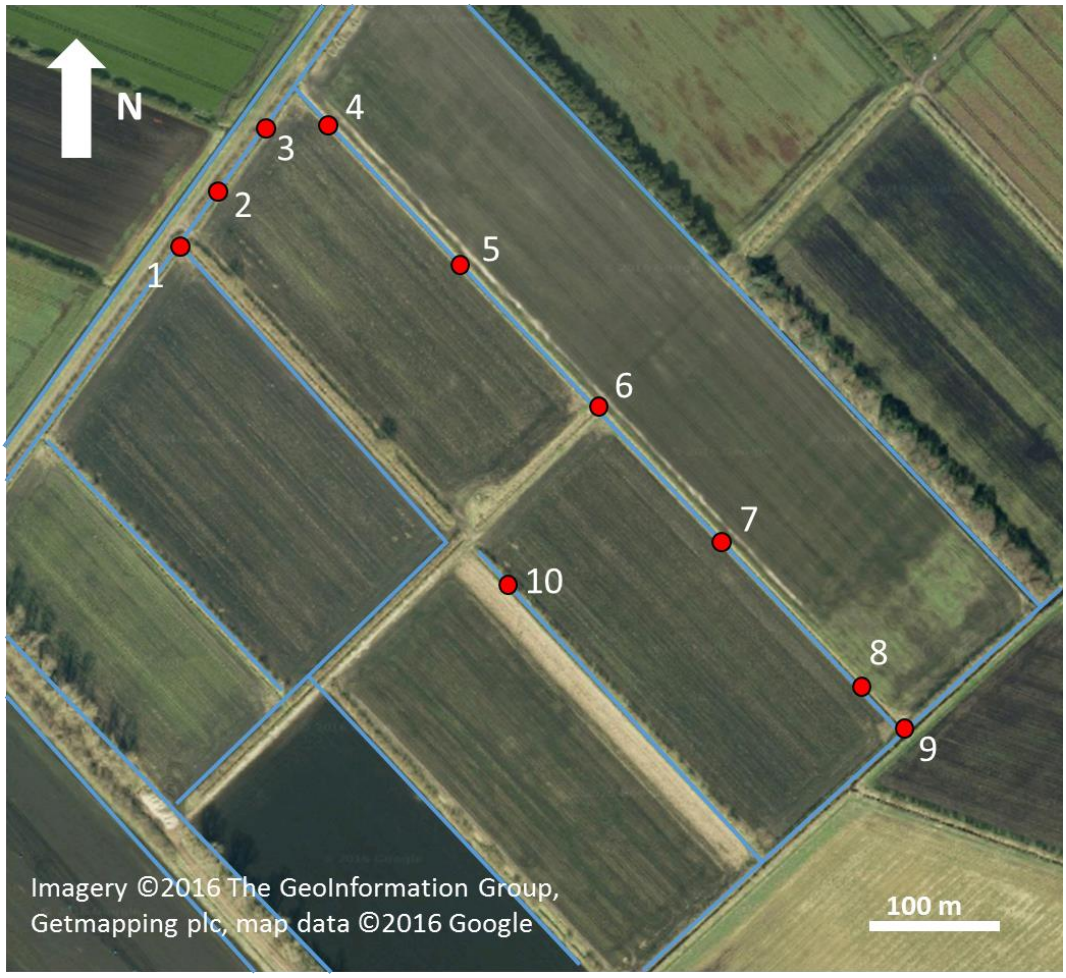


Figure 2. Maps of the semi-natural site (top panel), extensive grassland site (middle panel) and cropland site (bottom panel). Red dots mark numbered sampling locations, blue lines mark ditches/watercourses. For the semi-natural and extensive grassland sites, blue arrows mark where water is pumped onto site from Wicken Lode.

2.3 Analysis

Electrical conductivity (EC) and pH were measured on the 50 ml water sample. The sample was then passed through a nylon filter at 0.45 µm for further analysis. Dissolved organic carbon (DOC) and inorganic carbon (DIC) were analysed using a Shimadzu TOC Analyzer. DOC was measured as non-purgeable organic carbon (NPOC). Absorbance was measured at 280 nm using a Thermo Spectronic Helios Gamma Spectrophotometer. This was normalised against DOC concentration to give the specific ultraviolet absorbance (SUVA). SUVA is commonly measured at 254 nm, although high nitrate (NO_3^-) concentrations can interfere at wavelengths < 250 nm (Wang & Hsieh, 2001). Considering

the potential for high NO_3^- concentrations in surface waters in areas of intensive agriculture a SUVA wavelength of 280 nm was selected. NO_3^- was measured using a NICO 2000 ion-selective electrode and appropriate standards. Dissolved CH_4 and CO_2 were analysed using a Los Gatos Ultraportable Greenhouse Gas Analyzer equipped with a sampling loop (Baird *et al*., 2010). Dissolved N_2O was analysed on an Ai Cambridge GC94 equipped with an Electron Capture Detector (ECD).

Floating chamber fluxes of CH_4 and CO_2 fluxes were calculated according to Denmead (2008), using the modified formula:

$$F_g = \frac{1}{A}\frac{dg_m}{dt}$$

where F_g is the flux of CH_4 or CO_2 (M L^{-2} T^{-1} – mg m^{-2} day^{-1}), A is the area inside the chamber (L^2 – m^2), gm is the mass of gas in the chamber (M – mg), and t is time (T – days). Fluxes were calculated using a linear regression between time and chamber gas mass, and accepted if this regression was significant ($p \leq 0.05$). Fluxes that were not significant were assumed to be zero. Although it is usual to specify a cut-off value for the R^2 of the flux regression (below which value fluxes are rejected) we did not take this approach, because the high-frequency measurements provided by the analyser allowed detection of small but clearly non-zero (significant) fluxes despite high short-term scatter (low R^2). However, of the 253 fluxes that were significant, only 12 had an R^2 under 0.7. Fluxes were corrected for atmospheric pressure and temperature measured during each individual chamber deployment. Because of the short deployment time we assumed that pressure and temperature remained steady during flux measurement. Piston velocity was calculated using the standard formula (e.g. Gålfalk *et al*., 2013):

$$F = k\ x\ (C_{aq} - C_{eq})$$

where F is the CH_4 flux, k is the piston velocity, C_{aq} is the dissolved concentration of CH_4, and C_{eq} is the theoretical dissolved concentration if the water is in equilibrium with the air (calculated via Henry's Law). The formula was rearranged to give k.

2.4 Statistics

Statistical analysis was carried out in SPSS to determine if differences in GHGs and piston velocity (CH_4 and CO_2 flux, and dissolved concentrations of CH_4, CO_2 and N_2O) were present between sites. All six variables failed Levene's test for homogeneity of variance.

Kolmogorov-Smirnov tests were used to check for normal distributions. All six variables were not normally distributed, so transformations were sought to resolve this. CO_2 flux was transformed by square root transformation, and dissolved CO_2 was normalised by cube root transformation. Remaining variables could not be transformed to fit normal distributions. As such, a linear mixed model was used to test for differences between sites, using time as a repeated measure, and with Bonferroni correction for pairwise comparisons. Stepwise regression analysis was used as an exploratory test to look for relationships between dissolved CH_4 and CO_2 and the following variables: ditch water temperature, ditch depth, EC, absorbance at 280 nm, NO_3^-, DOC, SUVA, DIC, peat depth of the terrestrial fen, C:N, and water table in the terrestrial fen at the time of sampling (data for this was taken from Evans *et al*., 2016b). For the dissolved CO_2 model, pH was not used as an explanatory variable due to the fact that dissolved CO_2 and pH are interlinked (e.g. Abril *et al*., 2015). Differences were considered significant when $p \leq 0.05$.

3. Results

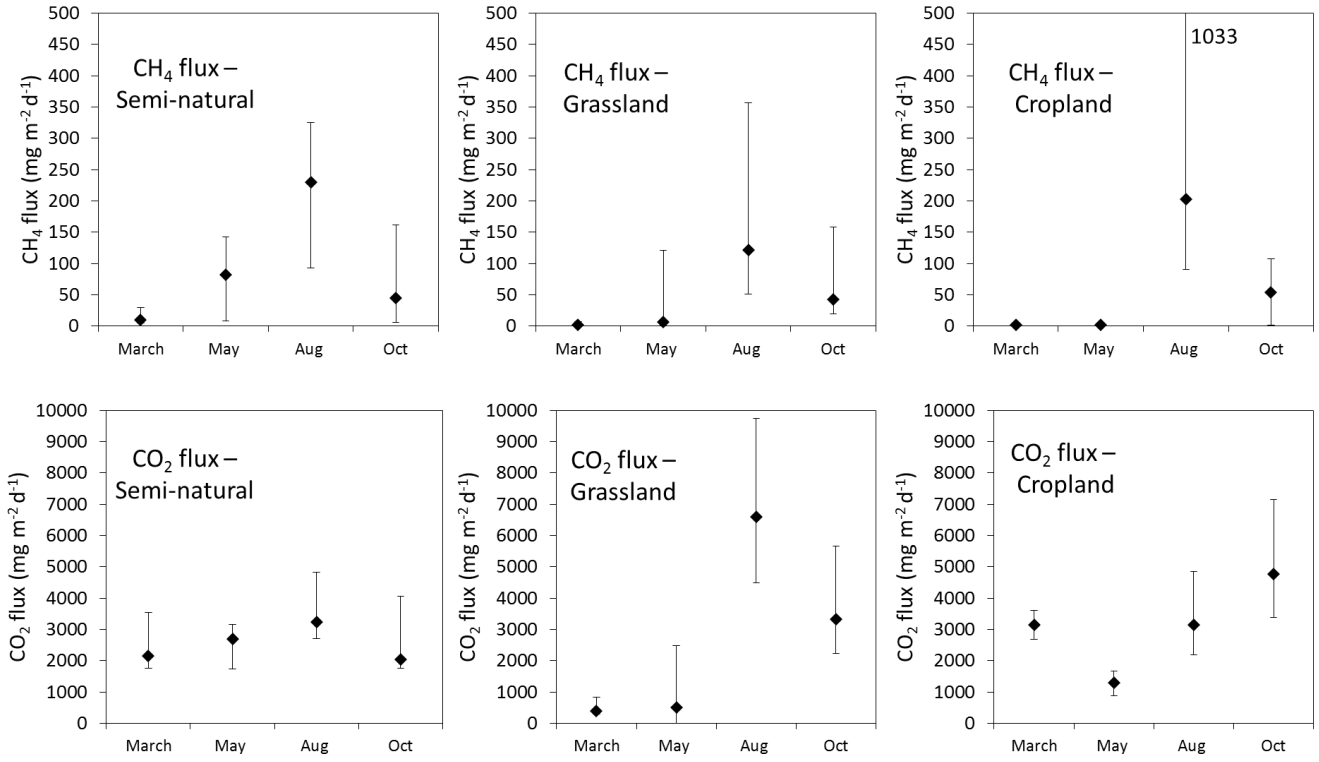
3.1. Water chemistry, ditch depths and environmental data

Table 1 displays a range of environmental and biogeochemical/physical data for the three sites through the year. Ditch water depths at the semi-natural site were consistently deep through the year (60 cm and above). Depths at the grassland site were generally shallow (~ 20cm), as were those at the cropland site, except during August when the mean was 60 cm. For all sites water and air temperature was highest during August. Mean ditch pH at all three sites was between 7.2 and 8.0, but EC was more variable both seasonally and between sites (intact < agricultural < restored). NO_3^- concentrations peaked in May at the cropland site (18 mg l^{-1}), presumably due to the use of fertilisers. At the grassland site NO_3^- was low (≤ 5 mg l^{-1}) except in March when the mean was 19 mg l^{-1}. The fen is rewetted during autumn and winter using high- NO_3^- river water, and the high concentration in March is a legacy of this rewetting. DOC concentrations were moderately high at the semi-natural and grassland sites (mean ~ 30 mg l^{-1}), but were lower by a third at the cropland.

3.2. Differences in ditch fluxes between and within sites

There was no significant difference in CH_4 flux between sites, but a significant ($p < 0.001$) difference was found for CO_2 flux between the cropland and other two sites (fig.3).

292 Median CH_4 fluxes were relatively low in March (≤ 10.5 mg m^{-2} d^{-1}). Fluxes stayed low in
293 May at the grassland and cropland fens, but were higher (80 mg m^{-2} d^{-1}) at the semi-natural
294 fen. Median CH_4 fluxes peaked in August at all three sites, at 120-230 mg m^{-2} d^{-1}. Highest
295 individual fluxes at each site were: 3650, 25400 and 7430 mg m^{-2} d^{-1} for the semi-natural
296 (May), grassland (August) and cropland (August) site respectively. CO_2 flux was relatively
297 stable at the semi-natural site at 2050-3250 mg m^{-2} d^{-1}, but fluctuated at the other two sites,
298 peaking at 6600 mg m^{-2} d^{-1} in August at the grassland site, and at 4760 mg m^{-2} d^{-1} in October
299 at the cropland site. Highest individual fluxes at each site were: 9580, 16800 and 13800 mg
300 m^{-2} d^{-1} for the semi-natural, grassland and cropland sites respectively, and were all recorded
301 in August. Differences were also apparent within sites, and median fluxes for each
302 individual sampling location are shown in fig.4.
303 There was considerable variation apparent in piston velocities between sites and
304 months, but none of these differences was significant (table 2).
305
306 *3.3. Annual ditch fluxes*
307 To calculate annual mean fluxes for 2014, a simple time-weighted median approach
308 was used, using the medians from fig.3. For CH_4, these produced estimates of 37.8, 18.3
309 and 27.2 g CH_4 $m^{-2}yr^{-1}$ for the semi-natural, grassland and cropland sites respectively, with
310 respective standard errors of 74.6, 244, and 97.3 g CH_4 $m^{-2}yr^{-1}$. For CO_2 the annual fluxes
311 were 1100, 1170 and 1440 g CO_2 m^{-2} yr^{-1} for the semi-natural, grassland and cropland sites
312 respectively, with respective SEs of 225, 340 and 312 g CO_2 m^{-2} yr^{-1}.



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315 Figure 3. Median ditch fluxes of CH_4 and CO_2 measured using floating chambers at the three sites. Error bars
316 represent first and third quartiles. . Note that the error bar for CH_4 in August at the cropland site exceeds the
317 scale. There was a significant difference ($p \leq 0.001$) for CO_2 fluxes between the cropland and other two sites.

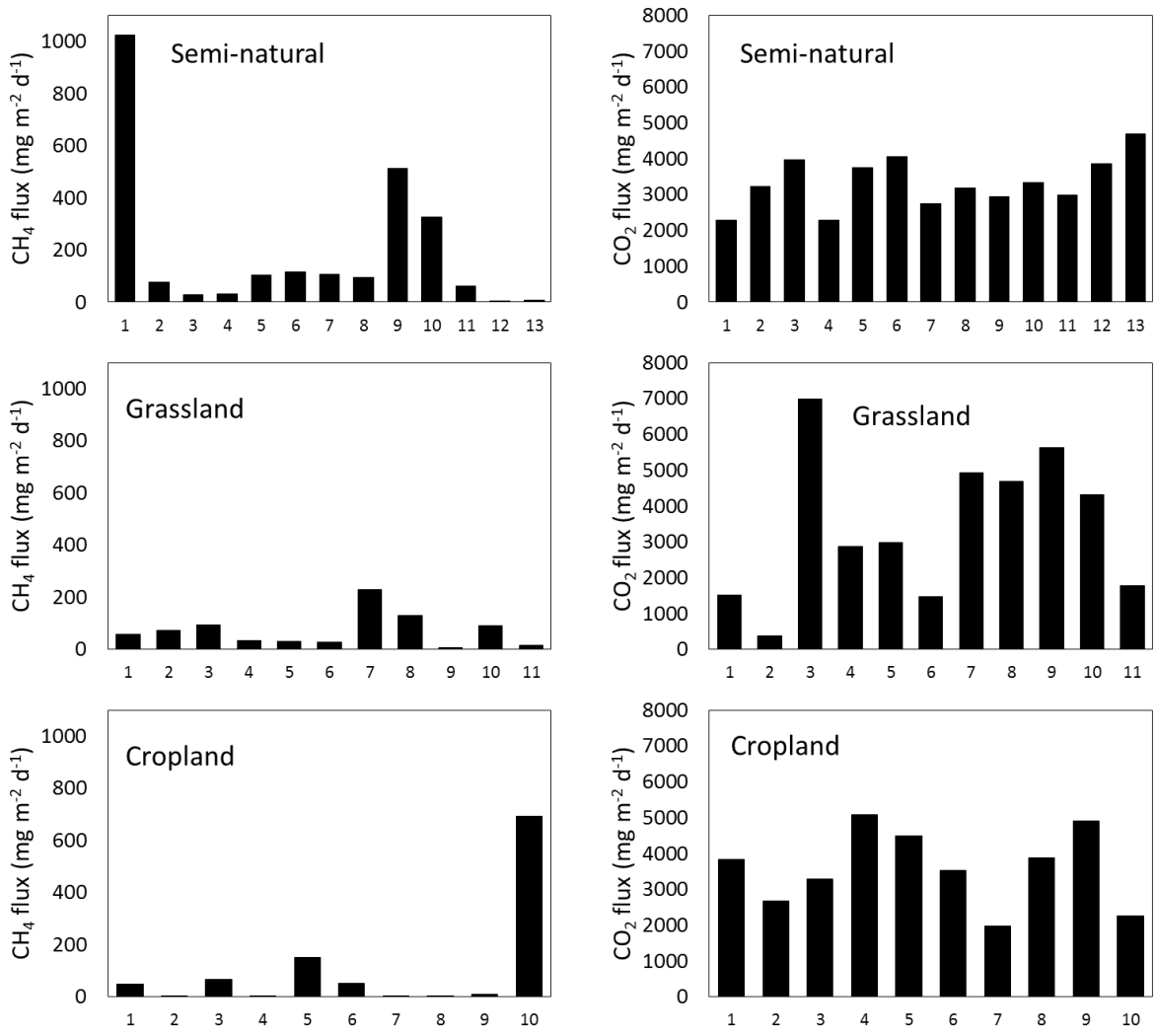

Figure 4. Median CH_4 (left) and CO_2 (right) fluxes for each individual numbered sampling point, grouped by site. Fig.2 displays numbered sampling points on site maps.

3.4. Differences in dissolved gases between sites

Significant differences were observed for dissolved CO_2 between the cropland and other two sites ($p < 0.001$). For dissolved CH_4, significant differences were found between the cropland and semi-natural fen ($p < 0.01$) and the cropland and grassland ($p < 0.05$). For N_2O, a significant difference was found between the cropland and other two sites ($p < 0.001$) (fig.5). Median CH_4 concentrations were below 0.1 mg l^{-1}, except for a spike of 0.43 mg l^{-1} at the cropland site in August. Median dissolved CO_2 at the semi-natural site showed no obvious seasonal variation (range 4.8-9.0 mg l^{-1}), whilst there was an increase through the year at the grassland site (1.7-7.5 mg l^{-1}). Dissolved CO_2 at the cropland site also peaked later in the year (15 mg l^{-1} in August and October). Median N_2O concentrations were under

1.5 µg l^{-1} at the semi-natural and grassland sites. At the cropland site N_2O concentrations were generally higher. Differences were apparent within sites, and median concentrations for each individual sampling location are shown in fig.6.

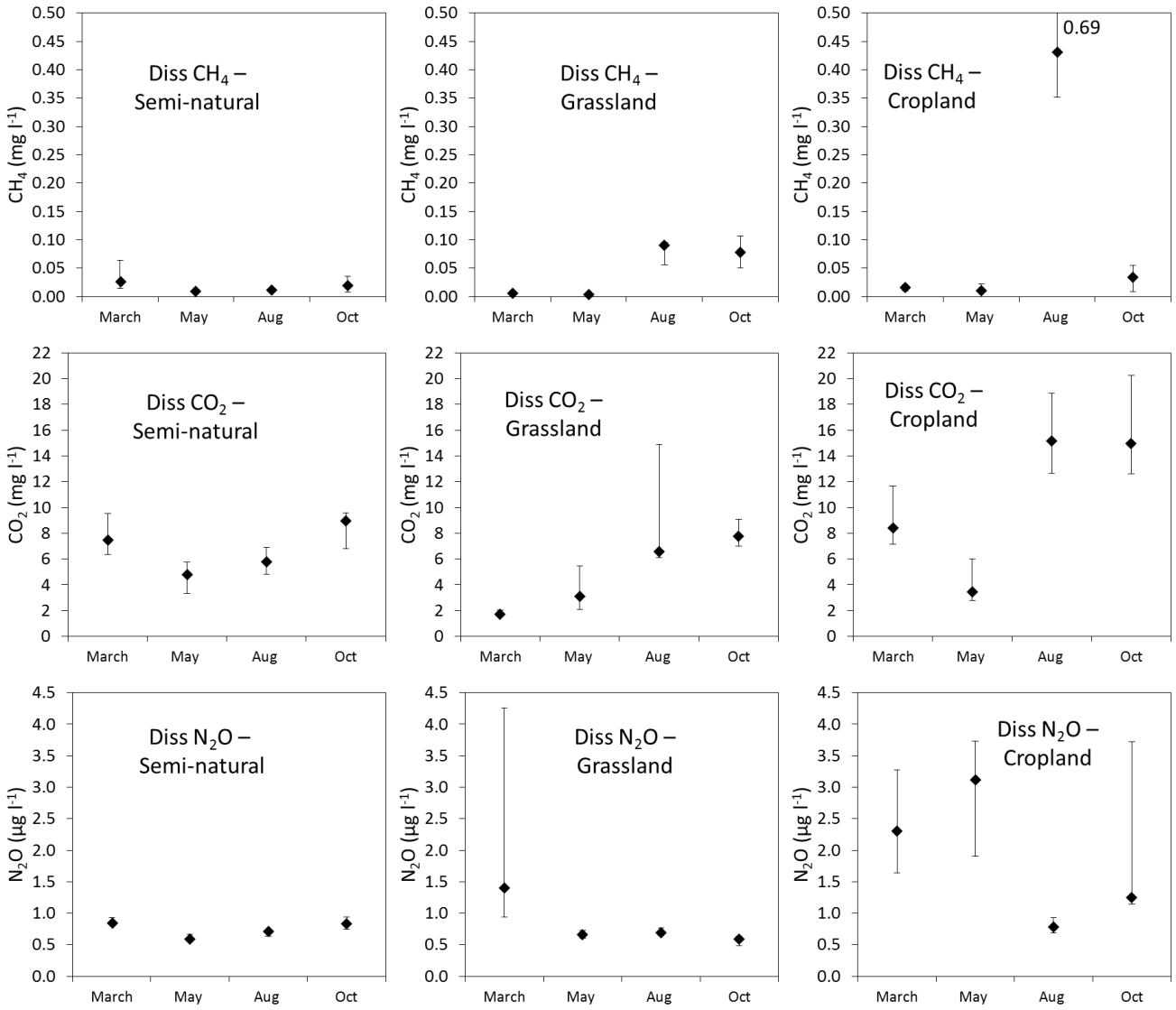

Figure 5. Median ditch dissolved concentrations of CH_4, CO_2 and N_2O at the three sites. Error bars represent first and third quartiles.. Note that the error bar for CH_4 in August at the cropland site exceeds the scale. There were significant differences between the cropland and other two sites for CO_2 ($p < 0.001$) CH_4 (cropland vs semi-natural $p < 0.01$, cropland vs grassland $p < 0.05$) and N_2O ($p < 0.001$).

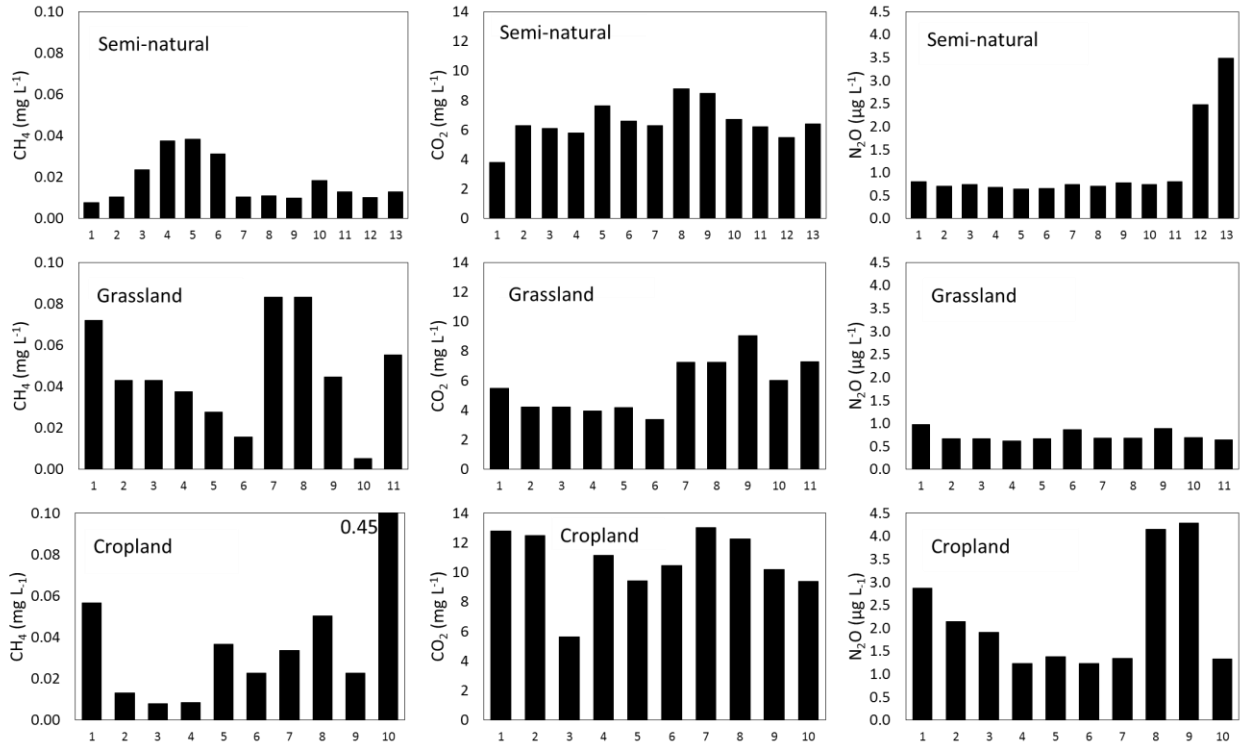

Figure 6. Median CH_4 (left) CO_2 (middle) and N_2O (right) concentrations for each individual numbered sampling point, grouped by site. Fig.2 displays numbered sampling points on site maps.

3.5. Drivers of dissolved GHGs

Significant regression models were produced for both dissolved CH_4 and CO_2, with respective R^2 values of 0.29 and 0.50. Table 3 displays the p values and slope coefficients used in each model.

There was a significant positive linear relationship between dissolved N_2O concentrations and NO_3^- ($p < 0.001$, $R^2 = 0.33$) but an improved fit was found between dissolved N_2O and the DOC:NO_3^- ratio, with N_2O concentrations increasing as the ratio decreased (fig.7). Apart from three clear outliers (which were not removed from the analysis), dissolved N_2O concentration did not rise above 1.5 µg l^{-1} until DOC:NO_3^- fell below 5. This relationship was consistent across all three sites.

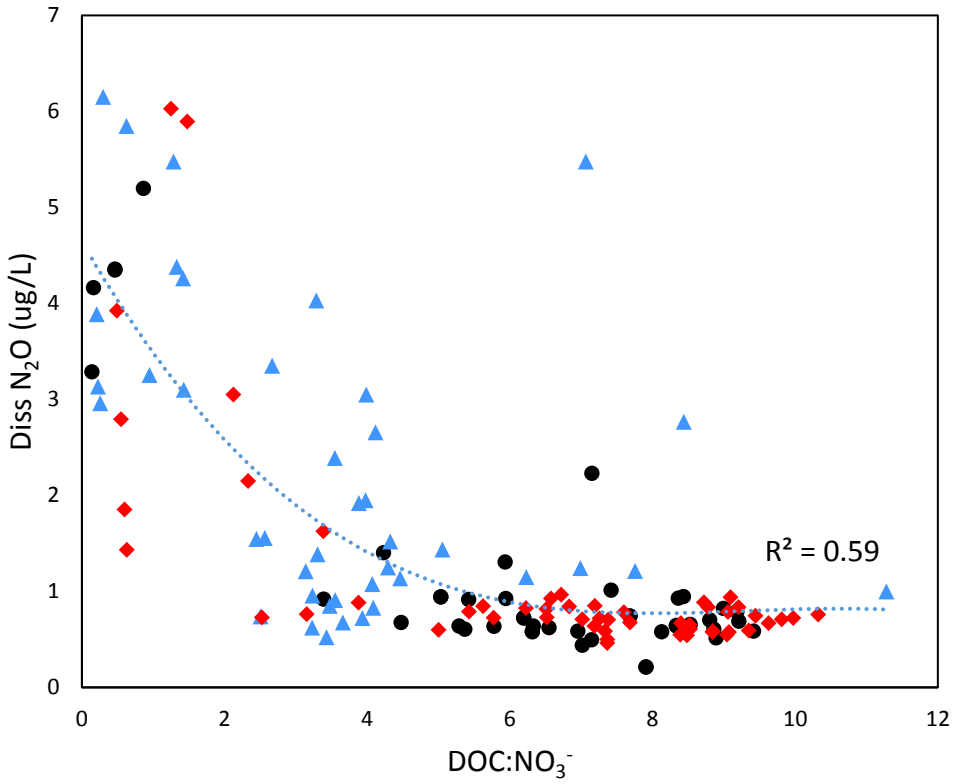

Figure 7. Relationship between dissolved N_2O and $DOC:NO_3^-$ ratio for all individual samples. Red diamonds = semi-natural fen, black circles = grassland, blue triangles = cropland. Trend line is 3rd order polynomial.

4. Discussion

4.1. Site characteristics

There were physical and biogeochemical differences in the ditches of the three fen sites. The ditches at the semi-natural site were deepest whilst those at the grassland site were shallowest, reflecting the difficulties in keeping this grassland site wet, as noted by Peh *et al*. (2014). Ditch water levels at the cropland site were also shallow, but were raised for irrigation during the peak of the growing season; this demonstrates the high degree of water management to maximise arable production (Morrison *et al*., 2013).

4.2. CO_2 and CH_4 fluxes

We no significant differences between sites for CH_4 fluxes, but fluxes were CO_2 fluxes were significantly higher at the cropland compared to the grassland and semi-natural sites.. There were seasonal patterns in fluxes of CH_4; emissions peaked at all three fens in August at which time they were not significantly different. It is likely that these high fluxes are due to the effect of summer temperatures on methanogenesis (Dunfield *et al*., 1993). There was

extensive within-site variation in gas fluxes, particularly for CH_4. For instance, at the semi-natural fen, sample point 1 had CH_4 fluxes an order of magnitude higher than the adjacent sample point 2. Sample point 1 was close to the wind pump that pumps river water onto the fen, and it could be that the mixing between low DOC/high NO_3^- river water and high DOC/low NO_3^- fen water produces a 'hotspot' of organic carbon processing resulting in CH_4 production (*sensu* Palmer *et al*., 2016). The lowest CH_4 fluxes were recorded at a ditch that bounded the edge of the fen, which displayed lower DOC concentrations and higher NO_3^- concentrations, suggesting that this ditch was connected to the river, and contained less organic substrates for methanogenesis. At the grassland site, CH_4 fluxes were highest at adjacent sample points 7 and 8 though this was not obviously related to any measured variables, e.g. EC, DIC and DOC were not elevated at these locations. At the cropland site, sample point 10 was extremely high compared to the other points. This ditch was shaded by a dense cover of trees and was near to a dead-end in the ditch system. This point had elevated levels of EC (25% higher compared to the mean of the other sample points) and DOC (114% higher), and it is likely that standing water here leads to an accumulation of organic matter and stagnation, and hence higher rates of methanogenesis (fig.1). This hypothesis is supported by the fact that dissolved CH_4 concentrations at this location were higher than any other sampling point at any site (fig. 6).

Although we only sampled four times within a year, our design featured large numbers of sampling points per site, and different sites were sampled at the same times of year. Calculated mean fluxes may not therefore be an accurate representation of the annual values, but should provide a reasonable representation of between-site differences. Mean CH_4 fluxes followed the order semi-natural>cropland>grassland. The mean flux for the semi-natural fen, 38 g CH_4 m^{-2} yr^{-1}, falls within the range of other reported fluxes from ditches in semi-natural peatlands; e.g. 12 g CH_4 m^{-2} yr^{-1} and 164 g CH_4 m^{-2} yr^{-1} from drained boreal fens (Glagolev *et al*., 2008, Minkkinen & Laine, 2006). The only reported annual ditch CH_4 flux from a temperate semi-natural site are 5.5 g CH_4 m^{-2} yr^{-1} from a UK upland blanket bog (Cooper *et al*., 2014). The considerably higher flux reported from our semi-natural site therefore shows the effect of nutrient status on ditch emissions. It has been suggested that ditch CH_4 emissions increase as land-use intensity increases (Evans *et al*., 2016a) but our data do not show this. The flux from our grassland site, 18 g CH_4 m^{-2} yr^{-1} is low compared to values such as 43, 66, 77 and 70 g CH_4 m^{-2} yr^{-1} from other low-intensity grasslands (Schrier-

Uijl *et al*., 2010, Vermaat *et al*., 2011, McNamara, 2013, Hendricks *et al*., 2007), although van den Pol-van Dasselaar *et al*. (1999) recorded an annual flux of just 11 g CH_4 m^{-2} yr^{-1}. To our knowledge, our annual ditch flux calculation of 27 g CH_4 m^{-2} yr^{-1} for the cropland is, along with the fluxes in our broader project report (Evans *et al*., 2016a), the first annual flux estimate for a temperate peatland under agriculture. There are several possible reasons for the highest annual flux being observed at the semi-natural site. Firstly, subsidence at our grassland site has resulted in the loss of the majority of peat soil, and it may be that the low organic content of the soil has led to a reduction in CH_4 production. This could be especially relevant if CH_4 is produced in the saturated peat, then transported laterally and degassed from ditches (e.g. fig.1); the grassland site dries out completely, presumably resulting in zero methanogenesis, whilst the water table remains in the peat at a deep level at the cropland, making this a plausible hypothesis. Secondly, the semi-natural site is likely to have a well-established methanogenic community compared to the other two sites where severe drainage and loss of peat (Stroh *et al*., 2013) may have disrupted the microbial communities (Jerman *et al*., 2009). Thirdly, the ditches at the semi-natural site were relatively deep, and depth fluctuations were minimal compared to the other two sites. This could lead to the formation of anoxic conditions, thus stimulating CH_4 emissions and reducing oxidation in the water column (O_2 measurements on future sampling campaigns would help to resolve this). Finally, Vermaat *et al*. (2011) recorded more ebullition in ditches sheltered by reed beds. It is therefore possible that steady ebullition contributed to the high fluxes at the semi-natural fen, as well as being responsible for the individual high fluxes (e.g. 25400 and 7430 mg m^{-2} d^{-1}) that were observed at the grassland and cropland. Ebullition from ditches in a Finnish mire measured using bubble traps was 3-37 mg m^{-2} d^{-1}, and was negligible (0.2-2.3% compared to diffusive emissions) in flowing ditches but substantial in ditches with standing water (10-22% of diffusive flux) (Minkkinen *et al*., 1997, 2006). Vermaat *et al*. (2011) calculated ditch ebullition by interpreting steep, short-term increases in CH_4 concentration in a floating chamber as evidence of bubbling, and stated that approximately 50% of total flux was due to ebullition. Other research using bubble traps has shown that ebullition in wetland and agricultural streams can equal the diffusive flux (Wilcock & Sorrell, 2008, Crawford *et al*., 2014). More measurements of ebullition in ditches are clearly needed.

Unlike CH_4, annual CO_2 fluxes did increase with land-use intensity, in the order semi-natural<grassland<cropland. Estimates of annual ditch CO_2 fluxes are lacking from the

literature, but scaling up the measurements of Vermaat *et al*. (2011) would produce annual fluxes of 1050 g CO_2 m^{-2} yr^{-1} for ditches in reed beds, and 1310 g CO_2 m^{-2} yr^{-1} for ditches in rough pasture. Our semi-natural site is therefore similar, with a flux of 1100 g CO_2 m^{-2} yr^{-1}, although our grassland annual flux was 1170 g CO_2 m^{-2} yr^{-1}. Our median CO_2 fluxes ranged from 488 mg m^{-2} d^{-1} to 8000 mg m^{-2} d^{-1}, and are therefore similar to those reported by Schrier-Uijl *et al*. (2011), Teh *et al*. (2011) and Hyvönen *et al*. (2013). CO_2 fluxes at the semi-natural site displayed less seasonality which may be a function of the deeper ditches minimising temperature increases in the basal peat, and therefore suppressing productivity (McEnroe *et al*., 2009).

4.3. Dissolved GHGs

For dissolved CH_4, CO_2 and N_2O, we found significant differences between the cropland compared to the grassland and semi-natural sites. This suggests that intensive agriculture has affected the biogeochemistry of the cropland ditches. Some sampling locations showed similar concentrations of dissolved gases, and this could be due spatial autocorrelation in dissolved GHGs (e.g. Chapra & Di Toro, 1991). This was most obvious for CH_4 at the semi-natural site, and N_2O at the semi-natural site and grassland.

Once pH had been removed as a predictive variable, we were able to account for 29% of temporal and spatial (within and between site) variability in CH_4, and 50% of variability in CO_2. For dissolved CO_2 there were positive relationships with depth to water table, DIC concentration, SUVA, and ditch depth. A deeper water table within the peat should result in increased decomposition, with CO_2 then exported laterally into ditches. A negative relationship between CO_2 and water depth has been found for pools in natural peatlands (McEnroe *et al*., 2009), and so our contrary finding could be due to the high degree of management at these fens; e.g. irrigation at the cropland reversed the natural seasonality in ditch depth and doubled the water level of the ditches in August, which coincided with the growing season increase in dissolved CO_2.

Dissolved CH_4 concentrations fell within the same range as those in agricultural streams (0.001-0.4 mg L^{-1}, Wilcock & Sorrell, 2008), and ditches in agricultural peatlands (maximum of 0.04 mg L^{-1}, Schrier-Uijl *et al*., 2011). They were of the same magnitude as 0.022 mg L^{-1} which was the calculated mean fluvial CH_4 concentration from 111 published studies (Stanley *et al*., 2016). Dissolved CH_4 correlated positively with air temperature, NO_3^-,

DIC, and depth to water table (in the fen/field), and negatively with EC. Higher temperatures could stimulate methanogenesis, leading to increased concentrations of CH_4. The positive correlation between dissolved CH_4 and depth to water table may, in part, be due to the confounding effect of seasonality; i.e. water tables were lower in the growing season when ditches become depleted in oxygen, leading to higher rate of methanogenesis. The positive correlation between NO_3^- and CH_4 is unexpected, as NO_3^- inhibits methanogenesis (Watson & Nedwell, 1998) and, as an electron acceptor, allows denitrifying bacteria to favourably out-compete methanogens (Le Mer & Roger, 2001). One possible explanation is that increased NO_3^- levels are associated with increased ammonium concentrations at the semi-natural fen (Conrad & Rothfuss, 1991), and the inhibitory effect of ammonium on methanotrophy is larger than the inhibitory effect of NO_3^- on methanogenesis. It may be that high nutrient levels associated with NO_3^- could coincide with inputs of labile organic matter, particularly at agricultural sites, thus stimulating methanogenesis when other electron acceptors have been depleted in the sediment. Alternatively, as discussed in section 4.2, it could be that CH_4 is produced in the saturated peat and then transported laterally into the ditch; i.e. methanogenesis occurs in zones distant from potential NO_3^- inhibition. Schade *et al*. (2016) did find a weak negative correlation between NO_3^- and CH_4 in a low NO_3^-/high DOC stream but found no correlation in a high NO_3^-/low DOC stream or in a high NO_3^-/high DOC stream. Similarly, Crawford *et al*. (2016) found no evidence that NO_3^- inhibited CH_4 production or emission in streams, and, in line with our hypothesis above, suggested that methanogenesis could be spatially removed from high NO_3^- concentrations. The absence of ditch depth from the CH_4 model is interesting as negative relationships between CH_4 flux and depth have been noted previously, although these are sometimes low; e.g. McEnroe *et al*. (2009) reported an R^2 value of 0.23 for pools, and Vermaat *et al*. (2011) found an R^2 of 0.15 for ditches. Pelletier *et al*. (2007) found both negative and positive relationships between CH_4 flux and depth in pools at different peatlands, and postulated that ebullition could be a confounding variable. The active water management at some sites could also be a confounding factor; as previously mentioned this management removes the natural seasonality in ditch depth. Finally, it is worth considering that wind speed may play a role in GHG dynamics. However, the ditches at our sites are predominantly sheltered by reedbeds or banks and, as previously

noted, the floating chamber did not drift, suggesting that wind speed was low on sampling days.

Dissolved N_2O was present in the ditches at all three fens, but was low at the semi-natural site. Concentrations were only high at the grassland site in March, but were high for most of the year at the cropland site, presumably due to the application of fertilisers to adjacent fields. Positive relationships between dissolved N_2O and N_2O flux have been demonstrated in rivers (Yang *et al*., 2011). Diffusive fluxes of N_2O have been shown to occur in oxygenated waters and it therefore seems highly probable that ditches at all three fens were sources of N_2O to the atmosphere. Wilcock & Sorrell (2008) measured N2O concentrations in agricultural streams between 0.26-28.5 μg l^{-1}, considerably higher than our maximum individual measurements of 6.15 μg l^{-1}, whilst concentrations in a eutrophic river have been reported as 0.66-1.14 μg l^{-1} (Silvennoinen *et al*., 2008). Sturm *et al*. (2014) recorded average concentrations of N_2O in lake surface water as 0.61 μg l^{-1} and 0.74 μg l^{-1}, similar to median concentrations at our grassland and semi-natural site, although concentrations at our cropland were higher. The authors also measured N_2O fluxes, with averages of 3.7 and 5.3 μg m^{-2} hr^{-1}. Reay *et al*. (2003) reported a relationship between N_2O fluxes and dissolved N_2O in UK agricultural ditches; applying that relationship to our data allows estimates of median flux for each fen to be calculated as 300, 210, and 1150 μg m^{-2} hr^{-1} for the semi-natural, grassland and cropland sites respectively. These fluxes at the semi-natural and grassland sites are similar to those reported by Teh *et al*. (2011) for ditches in a peatland pasture in the USA.

We found a statistically significant relationship between dissolved N_2O and NO_3^-, in agreement with others (e.g. Reay *et al*., 2003, Hinshaw & Dahlgren, 2013, Schade *et al*., 2016). However, a better fit was found between dissolved N_2O and the DOC:NO_3^- ratio. Aquatic systems generally show an inverse relationship between DOC and NO_3^- concentrations, which reflects a gradient from nitrogen limitation of microbial processes in carbon-rich systems to labile organic matter limitation in carbon-poor systems (Goodale et al., 2005; Taylor and Townsend, 2010). Our observation that dissolved N_2O only increases above ambient atmospheric concentrations when DOC/NO_3^- ratios are low suggests both that NO_3^- concentrations need to be high enough to allow denitrification to occur, and that labile organic matter concentrations need to be low enough to favour this process over other microbial processes such as NO_3^- reduction or assimilation. There were three samples

that appeared to deviate from the observed relationship, and it may be that higher concentrations of ammonium cause elevated N_2O concentrations, particularly if dissolved oxygen is not limiting (Liikanen & Martikainen, 2003).

4.4. Implications for GHG accounting and conclusions

Our data support previous studies in showing that ditches in both semi-natural and agricultural peatlands act as sources of CH_4, CO_2 and N_2O emissions. It is widely recognised that intact fens are important emitters of CH_4 (Turetsky *et al*., 2014). Although not intact, our semi-natural fen is under conservation management, and therefore the vegetation is similar to intact fens. As such, the annual terrestrial flux from our semi-natural site is 11.7 g CH_4 m^{-2} yr^{-1} (Evans *et al*., 2016b) , compared to 37.8 g CH_4 m^{-2} yr^{-1} from the ditches. When weighted by area for the entire fen, ditches would therefore be responsible for 0.53 g CH_4 m^{-2} yr^{-1}, approximately 5% of total emissions. Although ditches occupy only a fraction of the landscape, the magnitude of the fluxes observed here suggest that ditches in modified fen landscapes must be considered when calculating carbon balances, particularly for studies relying on static chamber, rather than eddy-covariance, methods, since this component of CO_2 and CH_4 emissions will otherwise be missed. In drained peatland systems, the contribution of ditches to the overall CH_4 budget is even more marked, because CH_4 fluxes from drained peat surfaces tend to be near zero (Willison, 1998, IPCC, 2014). At our cropland site, the field surface acted consistently as a small net sink for CH_4 (Evans et al., 2016b) and ditches were thus responsible for the entirety of CH_4 emissions from the system as a whole, which would give an areally-weighted flux of 0.44 g CH_4 m^{-2} yr^{-1} compared to the field sink of -0.17 g CH_4 m^{-2} yr^{-1}. This is probably true for agriculturally drained peatlands in general (IPCC, 2014, Evans et al., 2016a). Terrestrial fluxes at our grassland restoration site show that both uptake and emission of CH_4 occur, but the annual flux is approximately zero (Evans et al., 2016a). Thus, the ditches here are responsible for the majority of CH_4 emissions to the atmosphere, calculated on an areal basis as 0.31 g CH_4 m^{-2} yr^{-1}.

Our CO_2 fluxes were similar to others reported in the literature from ditches, which have often been of the same magnitude as fluxes from terrestrial fen (e.g. Schrier-Uijl *et al*., 2011, Hyvönen *et al*., 2013). Our median estimated N_2O fluxes for the semi-natural and grassland sites are slightly higher than terrestrial fluxes from a Finnish drained minerotrophic fen (Martikainen *et al*., 1995) whilst our calculated N_2O fluxes were in the

same range as mean terrestrial fluxes from a German agricultural fen (Flessa *et al*., 1998). These observations support previous suggestions that ditches do not act as hotspots for CO_2 and N_2O in the same way that they do for CH_4 (Evans *et al*., 2016a, Teh *et al*., 2011).

Future work should continue to examine the role that ditches play in releasing GHGs to the atmosphere, but a particular focus should be on CH_4. It is likely that high-frequency measurements combined with sampling replication on both small (i.e. field) and large (i.e. regional) scales would elucidate in greater detail the drivers between both dissolved GHGs concentrations and their efflux to the atmosphere. It is apparent that neglecting to consider ditches in drained peatlands will lead to significant errors when calculating landscape-scale GHG budgets.

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Bibliography

Abril, G., Bouillon, S., Darchambeau, F., Teodoru, C.R., Marwick, T.R., Tamooh, F., Ochieng Omengo, F., Geerraert, N., Deirmendjian, L., Polsenaere, P., Borges, A.V. 2015. Technical note: large overestimation of pCO_2 calculated from pH and alkalinity in acidic, organic-rich freshwaters. Biogeosciences, 12, 67-78.

Alm, J., Saarnio, S., Nykänen, H., Silvola, J., Martikainen, P.J. 1999. Winter CO2, CH4, and N2O fluxes on some natural and drained boreal peatlands. Biogeochemistry, 44, 163-186.

Baird, A.J., Stamp, I., Heppell, C.M., Green, S.M. 2010. CH_4 flux from peatlands: a new measurement method. Ecohydrology, 3, 360-367.

Bastviken, D., Tranvik, L.J., Downing, J.A., Crill, P.M., Enrich-Prast, A. 2011. Freshwater methane emissions offset the continental carbon sink. Science, 331, 6013, 50.

Best, E.P.H., Jacobs, F.H.H. 1997. The influence of raised water tables on carbon dioxide and methane production in ditch-dissected peat grasslands in the Netherlands. Ecological Engineering, 8, 129-144.

605 Billett, M.F., Moore, T.R. 2008. Supersaturation and evasion of CO_2 and CH_4 in surface waters at Mer Bleue
606 peatland, Canada. Hydrological Processes, 22, 2044-2054.
607
608 Burton, R.G.O., Hodgson, J.M. 1987. Lowland peat in England and Wales. Soil Survey Special Survey No.15. Soil
609 Survey of England and Wales, Harpenden.
610
611 Chapra, S.C., Di Toro, D.M. 1991. Delta method for estimating primary production, respiration, and reaeration
612 in streams. Journal of Environmental Engineering, 117, 640-655.
613
614 Cole, J.J., Prairie, Y.T., Caraco, N.F., McDowell, W.H., Tranvik, L.J., Striegl, R.G., Duarte, C.M., Kortelainen, P.,
615 Downing, J.A., Middelburg, J.J., Melack, J. 2007. Plumbing the global carbon cycle: integrating inland waters
616 into terrestrial carbon budgets. Ecosystems, 10, 172-185.
617
618 Cooper, M.D.A., Evans, C.D., Zielinski, P., Levy, P.E., Gray, A., Peacock, M., Norris, D., Fenner, N., Freeman, C.
619 2014. Infilled ditches are hotspots of landscape methane flux following peatland rewetting. Ecosystems, 17,
620 1227-1241.
621
622 Crawford, J.T., Stanley, E.H., Spawn, S.A., Finlay, J.C., Loken, L.C., Striegl, R.G. 2014. Ebullitive methane
623 emissions from oxygenated wetland streams. Global Change Biology, 20, 3408-3422.
624
625 Crawford, J.T., Stanley, E.H. 2016. Controls on methane concentrations and fluxes in streams draining human-
626 dominated landscapes. Ecological Applications, 26, 1581-1591.
627
628 Deemer, B.R., Harrison, J.A., Li, S., Beaulieu, J.J., DelSontro, T., Barros, N., Bezerra-Neto, J.F., Powers, S.M., dos
629 Santos, M.A., Vonk, J.A. 2016. Greenhouse gas emissions from reservoir water surfaces: a new global
630 synthesis. BioScience, doi: 10.1093/biosci/biw117.
631
632 Denmead, O.T. 2008. Approaches to measuring fluxes of methane and nitrous oxide between landscapes and
633 the atmosphere. Plant and Soil, 309, 5-24.
634
635 Dunfield, P., Knowles, R., Dumont, R., Moore, T.R. 1993. Methane production and consumption in temperate
636 and subarctic peat soils: response to temperature and pH. Soil Biology and Biochemistry, 25, 321-326.
637
638 Eades, P. 2016. Defra lowland peat project vegetation characterisation and monitoring final report 2015. In:
639 Evans, C.D., Morrison, R., Burden, A., Williamson, J., Baird, A., Brown, E., Callaghan, N., Chapman, P., Cumming,
640 A., Dean, H., Dixon, S., Dooling, G., Evans, J., Gauci, V., Grayson, R., Haddaway, N., He., Y., Heppell, K., Holden,
641 J., Hughes, S., Kaduk, J., Jones, D., Matthews, R., Menichino, N., Misselbrook, T., Page, S., Pan, G., Peacock, M.,
642 Rayment, M., Ridley, L., Robinson, I., Scowen, M., Stanley, K., Worrall. F. 2016b. Final report on project SP1210:
643 lowland peatland systems in England and Wales – evaluating greenhouse gas fluxes and carbon balances.
644 Defra.
645
646 Evans, C.D., Renou-Wilson, F., Strack, M. 2016a. The role of waterbourne carbon in the greenhouse gas
647 balance of drained and re-wetted peatlands. Aquatic Sciences, 78, 573-590.
648
649 Evans, C., Morrison, R., Burden, A., Williamson, J., Baird, A., Brown, E., Callaghan, N., Chapman, P., Cumming,
650 A., Dean, H., Dixon, S., Dooling, G., Evans, J., Gauci, V., Grayson, R., Haddaway, N., He., Y., Heppell, K., Holden,
651 J., Hughes, S., Kaduk, J., Jones, D., Matthews, R., Menichino, N., Misselbrook, T., Page, S., Pan, G., Peacock, M.,
652 Rayment, M., Ridley, L., Robinson, I., Rylett, D., Scowen, M., Stanley, K., Worrall. F. 2016b. Lowland peatland
653 systems in England and Wales – evaluating greenhouse gas fluxes and carbon balances. Final Report to Defra
654 on Project SP1210, Centre for Ecology and Hydrology, Bangor.
655
656 Flessa, H., Wild, U., Klemisch, M., Pfadenhauer, J. 1998. Nitrous oxide and methane fluxes from organic soils
657 under agriculture. European Journal of Soil Science, 49, 327-335.
658
659 Freeman, C., Lock, M.A., Reynolds, B. 1993. Fluxes of CO2, CH4, and N2O from a Welsh peatland followed
660 simulation of water table draw-down: potential feedback to climate change. Biogeochemistry, 19, 51-60.
661

662 Gålfalk, M., Bastviken, D., Fredriksson, S., Arenborg, L. 2013. Determination of the piston velocity for water-air
663 interfaces using flux chambers, acoustic Doppler velocimetry, and IR imaging of the water surface. Journal of
664 Geophysical Research Biogeosciences, 118, 770-782.
665
666 Glagolev, M.V., Chistotin, M.V., Shnyrev, N.A., Sirin, A.A. 2008. The emission of carbon dioxide and methane
667 from drained peatlands changed by economic use and from natural mires during the summer-fall period (on
668 example of a region of Tomsk oblast). Agrokhimija, 5, 46-58.
669
670 Glenk, K., Schaafsma, M., Moxey, A., Martin-Ortega, J., Hanley, N. 2014. A framework for valuing spatially
671 targeted peatland restoration. Ecosystem Services, 9, 20-33.
672
673 Glenn, S., Heyes, A., Moore, T. 1993. Carbon dioxide and methane fluxes from drained peat soils, southern
674 Quebec. Global Biogeochemical Cycles, 7, 247-257.
675
676 Goodale, C.L, Aber, J.D., Vitousek, P.M., McDowell, W.H. 2005. Long-term decreases in stream nitrate:
677 successional causes unlikely, possible links to DOC? Ecosystems, 8, 34-337.
678
679 Haddaway, N.R., Burden, A., Evans, C.D., Healey, J.R., Jones, D.L., Dalrymple, S.E., Pullin, A.S. 2014. Evaluating
680 effects of land management on greenhouse gas fluxes and carbon balances in boreo-temperate lowland
681 peatland systems. Environmental Evidence, 3:5.
682
683 Hendriks, D.M.D., Van Huissteden, J., Dolman, A.J., Van der Molen, M.K. 2007. The full greenhouse gas balance
684 of an abandoned peat meadow. Biogeosciences, 4, 411-424.
685
686 Hinshaw, S.E., Dahlgren, R.A. 2013. Dissolved nitrous oxide concentrations and fluxes from the eutrophic San
687 Joaquin River, California. Environmental Science and Technology, 47, 1313-1322.
688
689 Höll, B.S., Fiedler, S., Jungkunst, H.F., Kalbitz, K., Freibauer, A., Drösler, M., Stahr, K., 2009. Characteristics of
690 dissolved organic matter following 20 years of peatland restoration. Science of the Total Environment, 408, 78-
691 83.
692
693 Hope, D., Palmer, S.M., Billett, M.F., Dawson, J.J.C. 2004. Variations in dissolved CO_2 and CH_4 in a first-order
694 stream and catchment: an investigation of soil-stream linkages. Hydrological Processes, 18, 3255-3275.
695
696 Hughes, F.M.R., Stroh, P.A., Adams, W.M., Kirby, K.J., Mountford, O., Warrington, S. 2011. Monitoring and
697 evaluating large-scale, 'open-ended' habitat creation projects: a journey rather than a destination. Journal of
698 Nature Conservation, 19, 245-253.
699
700 Hutchinson, J.N. 1980. The record of peat wastage in the East Anglian fenland at Holme Post, 1848-1978 A.D.
701 Journal of Ecology, 68, 229-249.
702
703 Hyvönen, N.P., Huttunen, J.T., Shurpali, J., Lind, S.E., Marushchak, M.E., Heitto, L., Martikainen, P.J. 2013. The
704 role of drainage ditches in greenhouse gas emissions and surface leaching losses from a cutaway peatland
705 cultivated with a perennial bioenergy crop. Boreal Environment Research, 18, 109-126.
706
707 IPCC, 2006. Wetlands. In Eggleston, S., Buendia, L., Miwa, K., Ngara, T., Tanabe, K. (eds). 2006 IPCC Guidelines
708 for National Greenhouse Gas Inventories, Volume 4; Agriculture, Forestry and Other Land Use. IPCC National
709 Greenhouse Gas Inventories Programme.
710
711 IPCC, 2013: Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth
712 Assessment Report of the Intergovernmental Panel on Climate Change [Stocker, T.F., D. Qin, G.-K. Plattner, M.
713 Tignor, S.K. Allen, J. Boschung, A. Nauels, Y. Xia, V. Bex and P.M. Midgley (eds.)]. Cambridge University Press,
714 Cambridge, United Kingdom and New York, NY, USA, 1535 pp.
715
716 IPCC 2014, 2013 Supplement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories: Wetlands,
717 Hiraishi, T., Krug, T., Tanabe, K., Srivastava, N., Baasansuren, J., Fukuda, M. and Troxler, T.G. (eds). Published:
718 IPCC, Switzerland.

Jerman, V., Metje, M., Mandić-Mulec, I., Frenzel, P. 2009. Wetland restoration and methanogenesis: the activity of microbial populations and competition for substrates at different temperatures. Biogeosciences, 6, 1127-1138.

Kirschke, S., Bousquet, P., Ciais, P., Saunois, M., Canadell, J. G., Dlugokencky, E. J., Bergamaschi, P., Bergmann, D., Blake, D.R., Bruhwiler, L., Cameron-Smith, P., Castaldi, S., Chevallier, F., Feng, L., Fraser, A., Heimann, M., Hodson, E.L., Houweling, S., Josse, B., Fraser, P.J., Krummel, P.B., Lamarque, J-F., Langenfelds, R.L., Le Quéré, C., Naik, V., O'Doherty, S., Palmer, P.I., Pison, I., Plummer, D., Poulter, B., Prinn, R.G., Ribgy, M., Ringeval, B., Santini, M., Schmidt, M., Shindell, D.T., Simpson, I.J., Spahni, R., Steele, L.P., Strode, S.A., Sudo, K., Szopa, S., van der Werf, G.R., Voulgarakis, A., van Weele, M., Weiss, R.F., Williams, J.E., Zeng, G. 2013. Three decades of global methane sources and sinks. Nature Geoscience, 6, 813-823.

Le Mer, J., Roger, P. 2001. Production, oxidation, emission and consumption of methane by soils: a review. 2001. European Journal of Soil Biology, 37, 25-50.

Liikanen, A., Martikainen, P.J. 2003. Effect of ammonium and oxygen on methane and nitrous oxide fluxes across sediment-water interface in a eutrophic lake. Chemosphere, 52, 1287-1293.

Lorke, A., Bodmer, P., Noss, C., Alshboul, Z., Koschorreck, M., Somlai-Haase, C., Bastviken, D., Flury, S., McGinnis, D.F., Maeck, A., Müller, D., Premke, K. 2015. Technical note: drifting versus anchored flux chambers for measuring greenhouse gas emissions from running waters. Biogeosciences, 12, 7013-7024.

Martikainen, P.J., Nykänen, H., Alm, J., Silvola, J. 1995. Changes in fluxes of carbon dioxide, methane and nitrous oxide due to forest drainage of mire sites of different trophy. 1995. Plant and Soil, 168-169, 571-577.

McEnroe, N.A., Roulet, N.T., Moore, T.R., Garneau, M. 2009. Do pool surface area and depth control CO_2 and CH_4 fluxes from an ombrotrophic raised bog, James Bay, Canada? Journal of Geophysical Research, 114, G01001, doi: 10.1029/2007JG000639.

McNamara, N. 2013. CH_4 emissions from ditches in a drained lowland peat grassland, Somerset, UK. In: Emissions of greenhouse gases associated with peatland drainage waters. Report to Defra under project SP1205: Greenhouse gas emissions associated with non-gaseous losses of carbon from peatlands – fate of particulate and dissolved carbon. Report to the Department of Environment, Food and Rural Affairs, UK.

Minkkinen, K., Laine, J., Nykänen, H., Martikainen, P.J. 1997. Importance of drainage ditches in emissions of methane from mires drained for forestry. Canadian Journal of Forest Research, 27, 949-952.

Minkkinen, K., Laine, J. 2006. Vegetation heterogeneity and ditches create spatial variability in methane fluxes from peatlands drained for forestry. Plant and Soil, 285, 289-304.

Morris, J., Gowing, D.J.D., Mills, J., Dunderdale, J.A.L. 2000. Reconciling agricultural economic and environmental objectives: the case of recreating wetlands in the Fenland area of eastern England. Agriculture, Ecosystems and Environment, 79, 245-257.

Morrison, R., Cumming, A.M.J., Taft, H.E., Kaduk, J., Page, S.E., Jones, D.L., Harding, R.J., Balzter, H. 2013. Carbon dioxide fluxes at an intensively cultivated temperate lowland peatland in the East Anglia Fens, UK. Biogeosciences Discussions, 10, 4193-4223.

Nykänen, H., Alm, J., Lång, K., Silvola, J., Martikainen, P.J. 1995. Emissions of CH_4, N_2O and CO_2 from a virgin fen and a fen drained for grassland in Finland. Journal of Biogeography, 22, 351-357.

Palmer, S.M., Evans, C.D., Chapman, P.J., Burden, A., Jones, T.G., Allott, T.E.H., Evans, M.G., Moody, C.S., Worrall, F., Holden, J. 2016. Sporadic hotspots for physico-chemical retention of aquatic organic carbon: from peatland headwater source to sea. Aquatic Sciences, 78, 491-504.

Peh, K.S-H., Balmford, A., Field, R.H., Lamb, A., Birch, J.C., Bradbury, R.B., Brown, C., Butchart, S.H.M., Lester, M., Morrison, R., Sedgwick, I., Soans, C., Stattersfield, A.J., Stroh, P.A., Swetnam, R.D., Thomas, D.H.L., Walpole, M., Warrington, S., Hughes, F.M.R. 2014. Benefits and costs of ecological restoration: rapid assessment of changing ecosystem service values at a U.K. wetland. Ecology and Evolution, 20, 3875-3886.

Pelletier, L., Moore, T.R., Roulet, N.T., Garneau, M., Beaulieu-Audy, V. 2007. Methane fluxes from three peatlands in the La Grande Rivière watershed, James Bay lowland, Canada. Journal of Geophysical Research, 112, G01018, doi: 10.1029/2006JG000216.

Raymond, P.A., Bauer, J.E., Cole, J.J. 2000. Atmospheric CO_2 evasion, dissolved inorganic carbon production, and net heterotrophy in the York River estuary. Limnology and Oceanography, 45, 1707-1717.

Reay, D.S., Smith, K.A., Edwards, A.C. 2003. Nitrous oxide emission from agricultural drainage waters. Global Change Biology, 9, 195-203.

Roulet, N.T., Moore, T.R. 1995. The effect of forestry drainage practices on the emission of methane from northern peatlands. Canadian Journal of Forest Research, 25, 491-499.

Rowell, T.A. 1986. The history of drainage at Wicken Fen, Cambridgeshire, England, and its relevance to conservation. Biological Conservation, 35, 111-142.

Schade, J.D., Bailio, J., McDowell, W.H. 2016. Greenhouse gas flux from headwater streams in New Hampshire, USA: patterns and drivers. Limnology and Oceanography, DOI: 10.1002/lno.10337.

Schrier-Uijl, A.P., Kroon, P.S., Leffelaar, P.A., Van Huissteden, J.C., Berendse, F., Veenendaal, E.M. 2010. Methane emissions in two drained peat agro-ecosystems with high and low agricultural intensity. Plant and Soil, 329, 509-520.

Schrier-Uijl, A.P., Veraart, A.J., Leffelaar, P.A., Berendse, F., Veenendaal, E.M. 2011. Release of CO2 and CH4 from lakes and drainage ditches in temperate wetlands. Biogeochemistry, 102, 265-279.

Seitzinger, S.P., Kroeze, C. 1998. Global distribution of nitrous oxide production and N inputs in freshwater and coastal marine ecosystems. Global Biogeochemical Cycles, 12, 93-113.

Silvennoinen, H., Liikanen, A., Rintala, J., Martikainen, P.J. 2008. Greenhouse gas fluxes from the eutrophic Temmesjoki River and its estuary in the Liminganlahti Bay (the Baltic Sea). Biogeochemistry, 90, 193-208.

Sirin, A.A., Suvorov, G.G., Chistotin, M.V., Glagolev, M.V. 2012. Values of methane emission from drainage ditches. Environmental Dynamics and Global Climate Change, 3, 1–10.

Smith, L.C., MacDonald, G.M., Velichko, A.A., Beilman, D.W., Borisova, O.K., Frey, K.E., Kremenetski, K.V., Sheng, Y. 2004. Siberian peatlands a net carbon sink and global methane source since the early Holocene. Science, 303, 353-356.

Sobek, S., Algesten, G., Bergström, A-K., Jansson, M., Tranvik, L.J. 2003. The catchment and climate regulation of pCO_2 in boreal lakes. Global Change Biology, 9, 630-641.

Sundh, I., Nilsson, M., Mikkelä, C., Granberg, G., Svensson, B.H. 2000. Fluxes of methane and carbon dioxide on peat-mining areas in Sweden. Ambio, 29, 499-503.

Stanley, E.H., Casson, N.J., Christel, S.T., Crawford, J.T., Loken, L.C., Oliver, S.K. 2016. The ecology of methane in streams and rivers: patterns, controls, and global significance. Ecological Monographs, 86, 146-171.

Stroh, P.A., Mountford, J.O., Araya, Y.N., Hughes, F.M.R. 2013. Quantifying soil hydrology to explain the development of vegetation at an ex-arable wetland restoration site. Wetlands, 33, 311-320.

Sturm, K., Yuan, Z., Gibbes, B., Werner, U., Grinham, A. 2014. Methane and nitrous oxide sources and emissions in a subtropical freshwater reservoir, South East Queensland, Australia. Biogeosciences, 11, 5245-5258.

Taylor, P.G, Townsend, A.R. 2010. Stoichiometric control of organic carbon–nitrate relationships from soils to the sea. Nature, 464, 1178-1181.

Teh, Y.A., Silver, W.L., Sonnentag, O., Detto, M., Kelly, M., Baldocchi, D.D. 2011. Large greenhouse gas emissions from a temperate peatland pasture. Ecosystems, 14, 311-325.

Turetsky, M.R., Kotowska, A., Bubier, J., Dise, N.B., Crill, P., Hornibrook, E.R.C., Minkkinen, K., Moore, T.R., Mysers-Smith, I.H., Nykänen, H., Olefeldt, D., Rinne, J., Saarnio, S., Shurpali, N., Tuittila, E-S., Waddington, J.M., White, J.R., Wickland, K.P., Wilmking, M. 2014. A synthesis of methane emissions from 71 northern, temperate, and subtropical wetlands. Global Change Biology, 20, 2183-2197.

van den Pol-van Dasselaar, A., van Beusichem, M.L., Oenema, O. 1999. Methane emissions from wet Grasslands on peat soil in a nature preserve. Biogeochemistry, 44, 205–220

Vermaat, J.E., Hellmann, F., Dias, A.T.C., Hoorens, B., van Logtestijn, R.S.P., Aerts, R. 2011. Greenhouse gas fluxes from Dutch peatland water bodies: importance of the surrounding landscape. Wetlands, 31, 493-498.

Wang, G-S., Hsieh, S-T. 2001. Monitoring natural organic matter in water with scanning spectrophotometer. Environment International, 26, 205-212.

Watson, A., Nedwell, D.B. 1998. Methane production and emission from peat: the influence of anions (sulphate, nitrate) from acid rain. Atmospheric Environment, 32, 3239-3245.

Wilcock, R.J., Sorrell, B.K. 2008. Emissions of greenhouse gases CH4 and N2O from low-gradient streams in agriculturally developed catchments. Water, Air, and Soil Pollution, 188, 155-170.

Willison, T.W., Baker, J.C., Murphy, D.V. 1998. Methane fluxes and nitrogen dynamics from a drained fenland peat. Biology and Fertility of Soils, 27, 279-283.

Yang, L., Yan, W., Ma, P., Wang, J. 2011. Seasonal and diurnal variations in N_2O concentrations and fluxes from three eutrophic rivers in Southeast China. Journal of Geographical Sciences, 21, 820-832.

Yu, Z., Loisel, J., Brosseau, D.P., Beilman, D.W., Hunt, S.J. 2010. Global peatland dynamics since the Last Glacial Maximum. Geophysical Research Letters, 37, L13402, doi:10.1029/2010GL043584.

Table 1. Mean environmental, physical and water chemistry measurements for ditches at the three sites on the four sampling occasions. Numbers in brackets are standard error of the mean. Depth is ditch water depth.

		March	May	Aug	Oct		March	May	Aug	Oct
Air temp (°C)	Semi-natural	8	12.5	12.7	12.2	Water temp (°C)	7.3 (0.2)	15.5 (0.2)	18.6 (0.2)	10.2 (0.1)
	Grassland	8	12.5	12.7	12.2		10.7 (0.3)	15.2 (0.3)	17.6 (0.4)	10.1 (0.2)
	Cropland	6.9	13.7	15.9	9.1		9.9 (0.6)	12.6 (0.2)	15.5 (0.4)	9.8 (0.2)
Depth (cm)	Semi-natural	85.6 (4.8)	86.9 (3.5)	64.0 (6.7)	70.1 (5.5)	pH	7.7 (0.02)	7.9 (0.05)	7.6 (0.04)	7.6 (0.02)
	Grassland	23.3 (7.3)	36.4 (3.9)	16.2 (1.4)	21.1 (2.8)		8.0 (0.06)	7.8 (0.09)	7.3 (0.06)	7.6 (0.03)
	Cropland	18.4 (3.1)	32.7 (4.6)	60.6 (3.5)	20.5 (3.9)		7.5 (0.04)	7.8 (0.05)	7.2 (0.07)	7.2 (0.04)
EC ($\mu S\ cm^{-1}$)	Semi-natural	921 (62)	907 (18)	810 (73)	965 (51)	NO_3^- ($mg\ l^{-1}$)	11.6 (4.2)	6.7 (1.8)	5.2 (0.8)	6.5 (1.2)
	Grassland	994 (32)	1117 (102)	1306 (80)	1584 (99)		19.1 (5.8)	4.1 (0.1)	3.9 (0.1)	5.0 (0.2)
	Cropland	1263 (87)	968 (69)	888 (28)	1134 (65)		9.9 (1.4)	18.2 (4.6)	4.8 (0.7)	5.3 (0.4)
DOC ($mg\ l^{-1}$)	Semi-natural	28.7 (0.9)	28.9 (2.1)	27.4 (0.8)	37.4 (2.2)	SUVA	2.8 (0.1)	2.8 (0.1)	3.3 (0.1)	2.8 (0.1)
	Grassland	19.7 (2.7)	28.8 (1.1)	30.3 (1.6)	37.6 (2.4)		2.3 (0.1)	2.2 (0.0)	2.4 (0.1)	2.2 (0.2)
	Cropland	27.4 (3.3)	15.2 (4.8)	18.0 (2.8)	25.5 (3.7)		2.1 (0.1)	2.1 (0.1)	2.8 (0.1)	2.3 (0.1)
DIC ($mg\ l^{-1}$)	Semi-natural	92 (2)	80 (3)	70 (2)	97 (2)					
	Grassland	72 (2)	59 (6)	79 (4)	102 (3)					
	Cropland	69 (2)	52 (6)	68 (4)	77 (5)					

Table 2. Median piston velocities (m s^{-1}) for each site and each month, with first and third quartiles

		March	May	Aug	Oct
Semi-natural	Median	5.87E-06	1.16E-04	2.65E-04	8.58E-06
	1st	2.97E-06	1.33E-05	8.72E-05	3.96E-06
	3rd	1.33E-05	2.42E-04	4.17E-04	1.23E-04
Grassland	Median	4.89E-06	3.80E-05	9.18E-06	8.89E-06
	1st	2.23E-06	3.75E-06	7.99E-06	2.21E-06
	3rd	9.76E-06	7.87E-04	6.75E-05	2.33E-05
Cropland	Median	2.25E-06	2.26E-06	4.31E-06	8.01E-06
	1st	1.62E-06	1.67E-06	2.88E-06	4.39E-06
	3rd	3.40E-06	3.18E-06	7.20E-05	3.61E-05

Table 3. Results of the multiple linear regressions to determine the relationships between dissolved CO_2/CH_4 and other measured variables for the intact, restoration, and agricultural site. Note that depth to WT refers to the water table depth in the terrestrial part of the fen.

Diss CO_2	Slope coefficient	p
Intercept	-15.1	<0.001
Depth to WT	0.101	<0.001
DIC	0.133	<0.001
SUVA	2.594	0.001
Ditch depth	0.032	0.02

Diss CH_4	Slope coefficient	p
Intercept	-0.63	<0.001
Air temp	0.0148	<0.001
NO_3^-	0.0066	<0.001
DIC	0.0066	<0.001
Depth to WT	0.0019	<0.001
EC	-0.0002	0.02