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Author: Helen E. Taft Paul A. Cross Gareth Edwards-Jones
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Greenhouse gas emissions from intensively managed peat soils in an arable production system

Helen E. Taft [a, *], Paul A. Cross [a], Gareth Edwards-Jones [a, †], Edwin R. Moorhouse [b], Davey L. Jones [a]

[a] *School of Environment, Natural Resources and Geography, Deiniol Road, Bangor, Gwynedd, LL57 2UW, UK*

[b] *G's Fresh, Barway, Cambs., CB7 5TZ, UK*

[*] *Corresponding author. Email address:* h.taft@bangor.ac.uk

[†] *Deceased*

Highlights

- We monitored annual CO_2, N_2O and CH_4 emissions from arable peat soils.
- Emissions were dominated by CO_2 from SOM mineralization.
- Cumulative N_2O emissions were important, and CH_4 emissions negligible.
- Total emissions generally increased with an increase in SOM content.
- Intensive cultivation promotes loss of SOM and depletes the soil resource.

ABSTRACT

Organic-rich, eutrophic peat soils (Histosols) represent a major store of carbon (C) within the terrestrial biosphere. However, these soils are also highly susceptible to damage, particularly when used for intensive agricultural production. Sustainable management of such soils is contingent upon improved understanding of the impact of their management on the environment. In this context, we report the first annual budget of greenhouse gas emissions from temperate peat soils under intensive horticultural production. Fluxes of CO_2, N_2O and CH_4 were measured using static chambers on three farms along an organic matter loss gradient (~20%, ~35%, and ~70% soil organic matter (SOM) content respectively), under a number of commercially important crops in similar rotations. Cumulative annual fluxes of CO_2 in fallow and cropped soils were large and ranged from 13.0 ± 2.4 to 30.9 ± 2.5 t CO_2-e ha^{-1} y^{-1}, showing a general increase with SOM, and on cropped compared to bare soils. Annual emissions of N_2O varied from 5.0 ± 0.7 to 13.9 ± 1.9 t CO_2-e ha^{-1} y^{-1}, and CH_4 from -0.02 ± 0.08 to 0.04 ± 0.02 t CO_2-e ha^{-1} y^{-1}; neither showed a significant relationship with either SOM content or cropping. Distinct seasonal patterns of CO_2 and N_2O fluxes were observed, corresponding to significant correlations between emissions and soil and air temperature, soil moisture content, water table depth, and soil nitrate on some soil types. No discernible seasonal pattern in CH_4 fluxes was observed, and very few significant correlations with soil environmental variables were found. Compared to emissions estimates suggested in IPCC inventory guidelines for cultivated peat soils, the observed emissions in this study were relatively high, and net annual fluxes of CO_2 and CH_4 are equivalent to a loss of soil depth of 0.33 to 0.75 cm y^{-1}. We conclude that arable farming is promoting extreme mineralization of the soil's organic carbon reserves and that a change in land use or management regime is needed to protect and preserve this natural capital.

1. Introduction

Peat soils (Histosols) represent a major store of carbon (C) within the terrestrial biosphere (Limpens et al., 2008). While most studies of C loss have focused on non-agricultural peat soils, current evidence suggests that intensively cultivated lowland peats may also represent hotspots for greenhouse gas (GHG) emissions and related soil organic C (SOC) losses (Cannell et al., 1999; Dawson et al., 2010; Smith et al., 2007). Work on these arable peat systems has mainly focussed on quantifying CO_2 emission and changes in SOC storage, largely neglecting emissions of CH_4 and N_2O, and complete GHG budgets for organic soils under continuous arable management are extremely scarce (Evans et al., 2011; IPCC, 2014; Worrall et al., 2011). As GHG Emission Factors (EF) for arable mineral soils, or peat under managed grassland, are not likely to reflect emissions from these arable peats (due to differences in management regime and soil properties, for example), there is an urgent need to develop accurate and robust EFs for these agroecosystems. This is reinforced by their economic importance in terms of food security in many countries (Parish et al., 2008).

Improving emissions estimates facilitates accurate inventorying at the national level, a legal requirement for emissions reduction target compliance in many countries, and an important step in identifying mitigation priorities (IPCC, 2006). Mitigating agricultural emissions could contribute substantially to overall reduction targets. For example, in 2013, agriculture was estimated to be the second-largest sector contributor to emissions in Europe (9.9% CO_2-e of total EU_{28} emissions), with direct agricultural soil emissions of N_2O accounting for approximately 40% of this figure (EEA, 2015). Agricultural peat emissions at the national level are currently calculated using a default EF averaged over all temperate zones, with little recognition of regional differences in climate, peat soil characteristics and agricultural management practices when compared to the temperate-zone average. Consequently, soil N_2O EFs in particular have been identified as a priority for refinement,

dominating overall sources of uncertainty in EU GHG estimates since 1990 (EEA, 2015; Leip, 2010).

The sustainability of cropping on peat soils is an important consideration for long-term food security, with soil loss rates in drained and intensively cropped regions indicating that these practices are detrimental to the conservation of soil C stocks. Rates of soil loss from temperate lowland peats have been reported at 0.2 to 7.0 cm y^{-1} (Richardson and Smith, 1977; Ewing and Vepreskas, 2006), with a recent estimate from UK East Anglian arable fens of 1.10 to 1.48 cm y^{-1} between 1982 and 2004 (Dawson et al., 2010). Emissions of CO_2 may constitute between 35% and 100% of peat subsidence C losses (Leifeld et al., 2011), but the literature remains unclear regarding the proportion of total SOC loss that can be attributed to other routes (principally, wind and water erosion, leaching, and crop adherence). Quantifying GHG emissions from arable peats under different management regimes can therefore contribute to estimates of future soil losses, and enable prioritisation of soil loss mitigation measures via the different routes of loss.

Factors influencing emissions from agricultural soils are numerous and interact in often complex ways; they include soil (e.g. moisture, temperature, porosity, substrate availability), climate (rainfall, temperature), and vegetation (yield, water uptake), which in turn are driven by human activities such as farm operations (Li, 2007). Often, a change in a single variable may simultaneously increase the emission of one GHG and result in the reduction of another (Smith et al., 2008). Whilst individual studies have been conducted and models created that identify the relative importance of these factors in driving agricultural soil emissions (e.g. Giltrap et al., 2010), quantification of emissions drivers requires further attention with regard to intensively managed agricultural peat soils.

The primary aim of this study was to quantify and compare emissions of CO_2, CH_4 and N_2O from soils of comparatively high organic matter content (~70% SOM, ~35% SOM,

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and ~20% SOM respectively, to 1 m depth), under a number of commercially important horticultural/arable crops. We also aimed to determine which soil and crop factors most strongly influenced GHG fluxes from these soils.

2. Methods and materials

2.1. Study sites

The study area was located in East Anglia, UK, and comprised drained lowland fen typified by flat topography (0-1% slope) with long-term (1980-2013) mean annual rainfall of 621 mm, mean annual temperature of 10.2°C (winter mean 4.4°C, summer mean 16.4°C), and mean annual sunshine of 1280 h (UK MetOffice, 2014). All sites have been under long-term horticultural/arable production in rotation since c. 1940, growing primarily vegetables (e.g. celery, leeks, lettuces, potatoes, red beet) and wheat. Details of management practices implemented during the monitoring period are provided in the supplementary material (Supplementary tables S.1-S.3).

Three farms (sites) were identified for monitoring on the basis of their contrasting soil organic matter content to 1 m depth: (1) Site 1 comprised the low SOC farm site (SOC_{LOW}) where the soils had a SOM content of ~20%; (2) Site 2 comprised the medium SOC farm site (SOC_{MED}) where the soils had a SOM content of ~35%; (3) Site 3 comprised the high-SOC farm site (SOC_{HIGH}) where the soils had a SOM content of ~70%. Experimental monitoring sites were selected from each farm using farm records to identify fields with typical commercial cropping rotations. Crops selected for study included: celery (*Apium graveolens* L.), red beet (*Beta vulgaris* L., grown in tandem with a cover crop of barley, *Hordeum vulgare* L.), lettuce (*Lactuca sativa* L.), radish (*Raphanus sativus* L.), and potato (*Solanum tuberosum* L.). Six fields (three at the SOC_{MED} farm site, and three at the SOC_{HIGH} farm site)

were sampled monthly from April 2011 until June 2012, with a seventh field at the SOC_{LOW}

farm site added in June 2011.

One experimental sampling block (6×30 m) was randomly positioned within each

field (Fig. S.1). Blocks were located at least 10 m from field margins and areas of heavy

vehicle trafficking were avoided. Each sampling block contained five randomised pairs of

cropped (C) and bare (B) fallow plots 6×6 m in size with the long axis of the block running

parallel to the crop planting line. Inclusion of both cropped and bare plots enabled estimation

of autotrophic vs. heterotrophic respiration (after Koerber et al., 2010). During the growing

season the bare plots were covered with black geo-textile ground cover to suppress weed

growth. Each 6×6 m plot enclosed a 1 m buffer around its inner boundary to reduce edge

effects of adjacent plots and field areas, leaving a 4×4 m monitoring area containing one

randomly positioned GHG monitoring collar.

2.2. Seasonal greenhouse gas fluxes

Monthly greenhouse gas measurements were undertaken at all sites. Closed, non-vented static

chambers were used to monitor soil emissions of N_2O and CH_4. Cylindrical black

polyethylene collars (internal dimensions $d = 26.3$ cm, $h = 19.8$ cm; PBSL, Colchester, UK)

were inserted 12 cm into the soil and left *in situ* unless removed to allow tillage operations to

take place. All vegetation was removed from within and surrounding the collar at installation

at least 24 h before each sampling event. The static chambers fitting onto the collars consisted

of white opaque polypropylene closed cylinders (internal dimensions $d_1 = 22.0$ cm, $d_2 =$

25.0 cm, $h = 26.3$ cm; CJK Packaging, Derbyshire, UK), with a rubber septum sampling port

7 cm from the top of the chamber, and an internal battery powered 25 mm 12 V fan (typical

flow rate = 54 l min^{-1}; CPC Ltd., Leeds, UK). Chambers were inserted 4.5 cm into the top lip

of the soil collar so that the flexible seal around the chamber rim formed an air-tight seal with

the collar, giving a final enclosed headspace volume of 19.8 dm^3 (Fig. S.2). All chambers were vented for > 5 min prior to collar attachment and GHG sampling.

The chamber headspace was sampled four times at approximately 10 min intervals, with the first gas sample taken immediately after chamber enclosure. Glass sample vials (20 ml) fitted with butyl rubber septa (QUMA Electronik & Analytik GmbH, Wuppertal, Germany) were manually evacuated (60 ml) prior to sampling. Gas samples were removed from the headspace of the static chambers using a 30 ml syringe and a 21G, 5 cm needle. On insertion of the needle into the septum, the syringe was flushed twice then the sample taken and injected into the vial.

Gas samples were stored at room temperature in the dark until analysis. Sample analysis was within six weeks of collection using a gas chromatograph (Varian 450-GC, Bruker UK Ltd., Coventry, UK), equipped with a flame ionisation detector (FID, operated at 120-125°C) and electron capture detector (ECD, operated at 300°C), and attached to a QUMA QHSS®-40 Headspace Sampler (QUMA Electronik & Analytik GmbH, Wuppertal, Germany), which injected 2 ml of sample into the GC. Gas standards with certified concentrations to within 2-10% of their specification (STG Ltd., UK) were analysed concurrently with field samples. Sample concentrations of GHGs were calculated according to Levy et al. (2011).

Immediately following chamber measurements, soil CO_2 emissions were measured within 10-30 cm of the outside of chamber collars using an EGM-4 portable infra-red gas analyser (IRGA; PP Systems Ltd., Hitchin, UK) equipped with an automatic SRC-1 soil respiration chamber (internal dimensions $d = 10.35$ cm, $h = 6.50$ cm; total volume = 1694 ml), as described in Alm et al. (2007). Soil and air temperature (0-10 cm depth) were recorded at the time of sampling using a Checktemp1® temperature probe (accurate to ± 0.3°C; Hanna, Bedfordshire, UK).

All GHG samples from a single field were taken within a 3 h period during daylight hours with all seven fields sampled over a 96 h period.

2.3. Measurement of environmental variables

A range of soil and crop samples were taken within 24 h of chamber and IRGA measurements. In each plot, a randomly placed bulk density core ($V = 100$ cm^3) was used to collect soil from 0-5 cm and 0-10 cm depths. Soils were stored at 4°C then homogenised before analysis. Soil moisture and bulk density were calculated after drying (105°C, 24 h). A sub-sample of field-moist soil (0-5 cm depth) was used to extract plant available NH_4^+ and NO_3^- with 0.5 M K_2SO_4 or 1 M KCl for 1 h (1:5 w/v), then soluble N concentrations determined with a Powerwave XS Microplate Spectrophotometer (BioTek UK, Bedfordshire, UK) using the colorimetric methods outlined in Mulvaney (1996) and Miranda et al. (2001) respectively. Soil pH was measured on field-moist soil (0-10 cm depth) in 0.01 M $CaCl_2$ (1:1 w/v) following the method of Doran and Jones (1996). To estimate above-ground crop biomass during the growing season, five randomly selected plants were taken from near the sampling block in each field, their roots removed, and shoot fresh and dry (80°C, > 48 h) weight determined. At harvest, shoots were additionally separated into 'harvested' and 'residue' portions, which were weighed and dried separately. Water table depth (to within 10 cm) was measured at the field margin ditches perpendicular to the sampling block.

Additional soil samples were taken at each site in January 2012 to provide estimates of soil C and N stocks to 1 m depth. A bulk density core ($V = 100$ cm^3) was used to collect soil from the top 5 cm of each 10 cm depth, from 3 excavations at each site at 10 m intervals along a 20 m transect. Soil bulk density was calculated after drying (105°C, 24 h), and CN analysis performed on dried samples (< 2 mm fraction) using a LECO TruSpec CN analyzer (LECO Corp., St. Joseph, MI, USA).

2.4. Data cleaning and statistical analysis

Measured concentrations of N_2O and CH_4 were converted to hourly gas fluxes (µg N_2O-N m^{-2} h^{-1} or µg CH_4-C m^{-2} h^{-1}) using linear interpolation and the method described in Burden et al. (2013). Fluxes were visually examined and accepted for further analysis if the adjusted R^2 (R^2_{adj}) was $\geq$ 0.70 (after Ford et al., 2012, and Waddington et al., 2010), including the use of cleaned data where clear outliers were observed. Low fluxes were included in the analysis even if they had an R^2_{adj} value of < 0.70, to avoid a high flux bias in the data (Alm et al., 2007; Ford et al., 2012). Data cleaning resulted in 94% of N_2O and 92% of CH_4 individual flux curves being accepted for analysis. Soil respiration values were adjusted for diurnal temperature variation after Koerber et al. (2010), using the procedure described in Parkin and Kaspar (2003). We adjusted fluxes using Met Office Integrated Data Archive System (MIDAS) air temperature averaged for available stations within the local vicinity of each field site, and a Q_{10} of 2.2 for peat soils (York, 2012). No information could be found for appropriate adjustment for diurnal flux variation of N_2O or CH_4, so the raw data values were used.

Mean cumulative fluxes over the whole measurement period were calculated separately for each SOM level (SOC_{LOW}, SOC_{MED}, SOC_{HIGH}) and cropping combination by multiplying hourly values by 24 to give a daily flux, followed by linear stepwise interpolation of the flux values on known dates, then summing the resulting values over the required period (IPCC, 2000). Cumulative standard errors for each treatment were taken as the standard error of the cumulative means of individual chambers within that SOM-cropping (cropped vs. bare soil) type. Cumulative fluxes were calculated for the period 10th June 2011 to 9th June 2012 to allow inclusion of all sites for comparison. All cumulative flux estimates were converted to 100-year global warming potential (GWP_{100}) CO_2 equivalents (CO_2-e) according to the

methodology outlined in IPCC (2000), allowing comparison between SOM-cropping types for total GWP and individual GHG GWPs, and comparison with UK GHG Inventory EFs (Webb et al., 2014).

Statistical analyses were performed using SPSS v. 20 (IBM, Inc.), with significance being accepted at $p \leq 0.05$ except where stated otherwise. Statistical analyses were performed separately on each SOM-cropping (cropped vs. bare soil) combination. Normality was tested using the Shapiro-Wilk test (Field, 2005), and non-normal data were log-transformed or square-root transformed; homogeneity of variances were tested using Levene's or Welch's test statistic. Cumulative fluxes of CO_2, N_2O, and total GWP_{100} were compared using two-way ANOVA and Bonferroni Post-Hoc tests, using SOM% and cropping as fixed factors. Cumulative CH_4 fluxes showed heterogeneity of variances, so SOM% and cropping effects were assessed separately using the Kruskall-Wallis test and Mann-Whitney U test respectively. Relationships between individual GHGs and environmental variables (soil temperature, MIDAS mean air temperature, measured air temperature, daily and 5-day cumulative rainfall, soil N concentrations, soil bulk density, soil pH, and crop aerial biomass) were explored using Kendall's tau statistic (τ).

3. Results

3.1. Weather conditions

The mean annual air temperatures for the SOC_{LOW}, SOC_{MED} and SOC_{HIGH} sites were similar at 11.3, 11.2 and 11.0°C, respectively during the monitoring period (Fig. 1a). Recorded mean annual soil temperatures were 13.2, 13.0 and 13.5°C, respectively. The maximum recorded air temperature across the sites was 23.4 to 24.7°C (Jun. 2011) while the lowest was -7.2 to -4.6°C (Feb.). The mean annual rainfall at the SOC_{LOW}, SOC_{MED} and SOC_{HIGH} sites was 588, 588 and 612 mm, respectively. Over the whole monitoring period, SOC_{HIGH} was the coolest

and wettest site; while SOC_{MED} and SOC_{LOW} had similar higher daily average temperature and lower cumulative rainfall. Peak rainfall events over the measurement period were moderate and similar across sites, with peak daily rainfall events of 20 to 23 mm observed.

3.2. Seasonal patterns of greenhouse gas fluxes

A pronounced seasonal pattern of soil respiration was observed at all three sites (Fig. 1b). The highest fluxes were recorded in the summer (Jun.-Sept.; 157 to 201 mg CO_2-C m^{-2} h^{-1}) on cropped soils and 128 to 201 mg CO_2-C m^{-2} h^{-1} on bare soils) with the lowest fluxes seen in winter (Oct.-Feb.; 6 to 22 mg CO_2-C m^{-2} h^{-1} on cropped soils and -2.2 to 20 mg CO_2-C m^{-2} h^{-1} on bare soils). Net CO_2 uptake was only observed briefly, at the SOC_{LOW} site during October.

During most months, mean N_2O emissions were considerably smaller in magnitude than CO_2 emissions, even when corrected for GWP, but showed greater variability (Fig. 1c). Similarly to soil respiration, N_2O emissions tended to be lower in the winter and higher during the summer. Peak mean fluxes were observed in spring and early summer (Apr.-Jun.; 601 to 1116 µg N_2O-N m^{-2} h^{-1} on cropped soils and 489 to 1661 µg N_2O-N m^{-2} h^{-1} on bare soils) with the exception of a peak of N_2O from cropped soils at the SOC_{HIGH} site in October. Generally, minimum mean fluxes were recorded in winter (Oct.-Feb.; -6.7 to 27.7 µg N_2O-N m^{-2} h^{-1} on cropped soils and -75.2 to 24.0 µg N_2O-N m^{-2} h^{-1} on bare soils). Mean negative fluxes were recorded on only a few occasions, at the SOC_{LOW} site (Jul. and Oct.) and SOC_{HIGH} site (Feb.).

Mean CH_4 fluxes were very small when compared to the other GHGs, both in absolute terms and when corrected for GWP_{100}. Fluxes oscillated around zero for the whole of the measurement period, with no apparent seasonal trend in emissions (Fig. 1d). Net CH_4 uptake was observed on 39% of sampling occasions.

3.3. Seasonal patterns of soil N availability

At most sampling dates, mean soil concentrations of nitrate were generally higher than for

ammonium (Fig. 1e, f). The pattern of soil nitrate concentration over time differed between

sites, with peak levels observed in August at the SOC_{LOW} site (322 ± 56 mg NO_3-N kg dry

soil^{-1}), March at the SOC_{MED} site (227 ± 53 mg NO_3-N kg dry soil^{-1}), and October at the

SOC_{HIGH} site (219 ± 41 mg NO_3-N kg dry soil^{-1}). A peak in available nitrate was recorded at

all sites from March to May. Differences in soil nitrate between cropped and bare soil plots

were only observed on a few sampling dates. Where differences were seen these were mainly

in the summer, when mean NO_3-N tended to be lower in the cropped soil in comparison to

the bare soil plots. Mean soil nitrate over the whole observation period was highest at the

SOC_{LOW} site (87 ± 10 mg NO_3-N kg dry soil^{-1}), followed by the SOC_{HIGH} site (74 ± 4 mg

NO_3-N kg dry soil^{-1}), and lowest at the SOC_{MED} site (72 ± 4 mg NO_3-N kg dry soil^{-1}).

Soil available NH_4-N remained at a relatively low level throughout the observation

period (Fig. 1f). Generally, the pattern of available NH_4-N concentration was very similar at

all sites, with no discernible differences between cropped and bare soils, apart from at the

SOC_{MED} site where cropped NH_4-N was slightly lower during June in both years. Overall,

mean soil ammonium concentrations were almost identical at the SOC_{LOW} and SOC_{HIGH} sites

(19 ± 6 and 24 ± 1 mg NH_4-N kg dry soil^{-1} respectively), and higher at the SOC_{MED} site ($32 \pm$

5 mg NH_4-N kg dry soil^{-1}).

3.4. Effect of environmental variables on GHG emissions

Soil and air temperature both showed highly significantly positive correlations with soil CO_2

flux at both the SOC_{MED} and SOC_{HIGH} sites, but showed no significant correlation at the

SOC_{LOW} site (Table 1). In terms of the amount of variability explained by temperature, soil

temperature was the best predictor of CO_2 emission (15-27% of variability), followed by mean daily air temperature (15-22% of variability), with air temperature taken at the time of the flux measurement being the weakest predictor (13-19% of variability). This relationship was consistent at both the SOC_{MED} and SOC_{HIGH} sites on both cropped and bare soils, and was generally stronger on cropped than bare soils.

Indicators of soil moisture were generally negatively associated with CO_2 emission, with water table depth explaining the greatest proportion of variability in fluxes, but only at the cropped SOC_{MED} and the SOC_{HIGH} sites (8% and 10-11% of variability respectively; Table 1; Fig. 2). Soil moisture content (H_2O_{DW}) accounted for a further 3-5% of variability in soil respiration on cropped SOC_{MED} and SOC_{HIGH} soils and 5-8% of variability on SOC_{MED} and SOC_{HIGH} soils without crops. Overall, daily rainfall was a poor predictor of CO_2 emissions.

Soil nitrate, ammonium and total N were less consistent predictors of soil respiration, with the relationship varying between sites. At the SOC_{MED} site, soil N variables were negatively associated with soil respiration but only explained $< 3\%$ of the variability in CO_2 flux. In contrast, soil N variables were positively associated with CO_2 emission at the SOC_{LOW} and SOC_{HIGH} sites. A significant correlation on the SOC_{LOW} site was only found between NH_4-N and bare soil emission, but explained the highest proportion of variability (7%). Soil bulk density and pH were poor predictors of CO_2 emissions, accounting for $< 5\%$ of the variability in below-ground CO_2 fluxes. Crop aerial biomass was only significantly correlated with soil respiration at the SOC_{HIGH} site, but accounted for a large proportion (30%) of variability in CO_2 emissions.

Temperature variables were significantly positively correlated with soil N_2O emissions, although more weakly than for soil respiration. Soil temperature significantly predicted N_2O emission at all sites, explaining 4%, 2-3%, and 5-9% of emissions from

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SOC_{LOW}, SOC_{MED} and SOC_{HIGH} sites respectively. Daily air temperature only significantly correlated with N_2O emission at the SOC_{HIGH} site, accounting for 3-5% of variability in fluxes, however, this was a better predictor than air temperature recorded during the GHG monitoring period.

Soil moisture variables were negatively associated with N_2O emission, but only in a few categories, and only weakly compared to CO_2 emission predictors. Water table depth accounted for 4-5% of variability at the SOC_{HIGH} site, while soil moisture content explained 1-2% of variability in fluxes on the SOC_{MED} and SOC_{HIGH} bare soils, and daily rainfall explained $\leq$ 1% of variability across all sites. Soil N variables were also weaker predictors of N_2O than CO_2 emission, explaining only 1-2% of variability in fluxes. Soil bulk density and pH were both very weak predictors of N_2O emission. Crop aerial biomass was only significantly associated with fluxes at the SOC_{LOW} site, but accounted for 44% of variability in emissions.

Only a small number of environmental variables measured here were significantly associated with methane flux, and only weakly so. Of these, the most significant was soil nitrate which was positively correlated with CH_4 efflux on bare SOC_{LOW} soils, explaining 5% of the flux variability.

3.5. Cumulative GHG emissions

Table 2 shows annual cumulative emissions of CO_2, N_2O, CH_4 and overall GWP_{100}. Similarly to seasonal emissions, CO_2 represented the largest annual flux from all soil types on both cropped and bare soil plots, ranging from 74 to 80% of total GWP_{100} on cropped soils to 61 to 80% of total GWP_{100} on soils without crops. N_2O represented a smaller but still substantial annual emission, of between 20 and 26% of GWP_{100} on cropped soils and

20 and 39% of emissions on bare soils. On all sites, cumulative methane emission was negligible, comprising $< 0.5\%$ of annual emissions.

On both cropped and bare soils, cumulative GWP_{100} was lowest at the SOC_{LOW} site, highest at the SOC_{MED} site, and intermediate at the SOC_{HIGH} site (Table 2). At all sites, total cropped emission was greater than bare soil emission. Two-way ANOVA showed a significant main effect of both SOM content ($F = 12.254$, $p < 0.001$) and cropping ($F = 4.882$, $p < 0.05$) on cumulative GWP_{100} (Table 3). No interaction effects between SOM% and cropping were evident. Bonferroni Post-Hoc tests identified a highly significant increase in GWP_{100} between the SOC_{LOW} and SOC_{MED} sites ($p < 0.001$), but the decline in GWP_{100} between SOC_{MED} and SOC_{HIGH} sites proved non-significant.

Patterns of annual CO_2 emission broadly followed that of annual GWP_{100} (Table 2). Cropping had a significant main effect on cumulative CO_2 emissions, with higher mean fluxes from cropped plots at all three sites ($p < 0.01$; Table 3). A significant main effect was also observed between annual CO_2 emission and SOM% ($p < 0.001$). No significant interaction effect on cumulative CO_2 emission was found between cropping and SOM content. However, the response of cumulative emissions to increasing SOM content differed according to cropping: on bare soils, total CO_2 emission increased from SOC_{LOW} to SOC_{HIGH} sites; but on cropped soils, total annual emission increased from SOC_{LOW} to SOC_{MED} sites, then declined slightly at the SOC_{HIGH} site (Table 2). Bonferroni Post-Hoc tests identified the increase in annual CO_2 emission between SOC_{LOW} and SOC_{MED} sites as highly significant ($p < 0.01$), but not the difference between SOC_{MED} and SOC_{HIGH} sites.

On both bare and cropped soils, cumulate annual N_2O emission was highest at the SOC_{MED} site, lowest at the SOC_{LOW} site, and intermediate at the SOC_{HIGH} site, while no distinct pattern in annual emission relative to cropping was evident (Table 2). Results of the two-way ANOVA supported these observations: SOM% significantly influenced cumulative

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N_2O emission ($p < 0.01$), but cropping did not (Table 3). No significant interaction effects were observed between SOM% and cropping. In contrast to annual CO_2 emission, Bonferroni Post-Hoc tests identified a significant difference between annual N_2O emission at the SOC_{MED} and SOC_{HIGH} sites ($p = 0.01$), but not between the SOC_{LOW} and SOC_{HIGH} sites.

The effects of SOM content and cropping on annual CH_4 fluxes were assessed using the non-parametric Kruskall Wallis test and Mann-Whitney U test, respectively. No significant effects of either variable on annual CH_4 emission were identified.

3.6 Soil C and N stocks

Estimates of soil C and N stocks to 1 m depth for each of the study sites are shown in Table 4.

Soil C content at the SOC_{LOW} site varied between 16.7 and 19.1% in the upper 40 cm of soil, below which it sharply declined to between 3.8 and 8.7%. Soil N content at the SOC_{LOW} site followed a similar pattern (1.2 to 1.4% in the upper 40 cm, declining to 0.3 to 0.7% below the 40 cm layer). C:N ratios of individual 10 cm soil layers varied between 11.8 to 15.4 at this site, averaging 14.0 in the top 40 cm soil and 13.1 in the 40-100 cm layer.

The vertical profiles of soil C and N content at the SOC_{MED} and SOC_{HIGH} sites were comparable, but differed distinctly in the upper profile from those of the SOC_{LOW} site. Both soil C and N content were highest at c. 20-70 cm depth, lowest at below c. 70 cm, and intermediate in the top 20 cm soil (Table 4). Soil C and N content (0-20 cm depth) were only marginally higher at SOC_{HIGH} that at SOC_{MED} (34.9 and 33.4% C, and 2.1 and 2.0% N respectively). However, C and N% (20-70 cm depth) were on average notably higher at SOC_{HIGH} than SOC_{MED} (47.9 and 38.6% C, and 2.7 and 2.2% N respectively), as were C and N% below this depth (17.3 and 1.6% C, and 1.0 and 0.1% N respectively). C:N ratios of

individual 10 cm soil layers were more variable than at SOM_{LOW}, at between 9.7 and 24.6 at the SOC_{MED} site and 15.1 and 19.5 at the SOC_{HIGH} site.

Vertical bulk density profiles also differed between the SOC_{LOW} site (increasing from 0.76 to 0.99 g cm^{-3} in the top 50 cm soil, then falling to 0.61 g cm^{-3} at 90 cm depth) and the SOC_{MED} and SOC_{HIGH} sites (both showing similar profiles which declined from c. 0.37 g cm^{-3} at 0 cm depth to c. 0.18 g cm^{-3} at 50 cm depth, then increased to a maximum of c. 1.36 g cm^{-3} at 90-100 cm depth).

The combined effect of vertical bulk density profiles and vertical C and N % distributions, resulted in soil C and N stock profiles that generally declined with depth at all sites, although this occurred at different rates between sites. Total soil C stock (0-100 cm) was 787.5 t ha^{-1} at the SOC_{LOW} site, 907.0 t ha^{-1} at the SOC_{MED} site, and 953.7 t ha^{-1} at the SOC_{HIGH} site. Total soil N stock (0-100 cm) was 57.0 t ha^{-1} (SOC_{LOW} site), 53.1 t ha^{-1} (SOC_{MED} site) and 57.4 t ha^{-1} (SOC_{HIGH}) respectively.

4. Discussion

4.1. Seasonal and cumulative annual GHG emissions

Despite the importance in horticultural production of eutrophic Histosols, we are not aware of any studies that have monitored a full annual cycle of GHG emissions from Histosols under continuous intensive horticultural management. Annual emissions in this study were dominated by CO_2, followed by N_2O, whilst CH_4 emissions were small. This is similar to the pattern of GHG emissions found in other studies of disturbed peat soils (Elder and Lal, 2008; Maljanen et al., 2004). Cumulative annual soil respiration rates from this study were high in comparison to some previous studies (6.9 to 20.0 t CO_2-e ha^{-1} y^{-1}, Kasimir-Klemedtsson et al., 1997; Maljanen et al., 2004), but considerably lower than those found by other authors (44 to 122 t CO_2-e ha^{-1} y^{-1}, Elder and Lal, 2008; Elsgaard et al., 2012). Differences may be

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attributable to post-harvest microbial respiration responses to plant residue breakdown, with a large quantity of available residue in the Elder and Lal (2008) study (maize crop), compared to those in the former studies (barley crops) and this study (various relatively low-growing crops). Site sensitivity to changes in soil temperature can also be a powerful driver of respiration, and has been found to differ depending on local soil and other environmental factors (e.g. Elsgaard et al., 2012).

Mean cropped CO_2 emissions were greater than mean bare soil emissions, but only by a relatively small amount, with the majority of soil respiration (58 to 96% on a per-field basis) a result of microbial rather than root respiration. The proportion of respiration, as root respiration, found here showed greater variability than the 35-45% reported by Kasimir-Klemedtsson et al. (1997), but was of similar magnitude on all but the SOC_{HIGH} site. This can perhaps be attributed to high rates of microbial respiration at the SOC_{HIGH} site minimising the relative importance of root respiration within the total soil respiration budget. However, we were unable to account for possible priming effects in this study (of previous crops on the bare soil plots, or of present crops on cropped plots), limiting the degree to which this interpretation may be applied to these sites (Kuzyakov et al., 2000; Kuzyakov, 2010). It would be useful for future studies of these soils to include an estimate of the magnitude of any priming effects on relative soil decomposition rates, to aid differentiation between autotrophic vs. heterotrophic respiration contributions to total CO_2 emission, particularly in relation to different individual crops, and cropping rotations. Discerning interaction effects between cropping (cropped vs. bare soils) and SOM content relative to CO_2 and other GHG emissions was difficult in this study. Although cropping rotations were similar on all soil types, thus allowing categorisation as horticultural or intensive arable sites, cropping rotations were not identical (Supplementary tables S.1-S.3). Ideally, future work would aim to compare identically timed cropping rotations on soils of differing OM content,

to give a more robust comparison of emission factors. The present study rather contributes a general indication of the magnitude and patterns of emissions from these sites.

This study found a significant positive association between CO_2 emission and temperature and a negative association with water table depth at the cropped SOC_{MED} and the SOC_{HIGH} sites. The relationship between temperature and SOM oxidation is well documented (Dawson and Smith, 2007), and many studies have found soil and air temperature to be among the strongest predictors of soil respiration rate (e.g. Elsgaard et al., 2012; Estop-Aragonés and Blodau, 2012; Maljanen et al., 2001, 2002). Lowering the water table is also commonly associated with higher CO_2 flux from peat soils (Kechavarzi et al., 2007; Maljanen et al., 2001). It is therefore possible that raising the height of the water table closer to the soil surface may help reduce CO_2 emissions from these soils (e.g. Hatala et al., 2012). While this is unlikely to significantly impact on reducing CH_4 emissions, it may increase N_2O emissions, particularly if N derived from SOM mineralization or fertilisers are leached into this zone (e.g. Boon et al., 2014).

The maximum rate of N_2O emissions observed here were generally lower than previous reports for vegetable crops, although our minimum fluxes were similar. For example, previous studies from temperate and boreal arable Histosols have measured N_2O fluxes up to 7083 µg N_2O-N m^{-2} h^{-1}, although the mean flux tends to fall within the range of 30 to 3000 µg N_2O-N m^{-2} h^{-1} (Elder and Lal, 2008; Flessa et al., 1998; Maljanen et al., 2002; Regina et al., 2004; Rochette et al., 2010; Weslien et al., 2012). The pattern of seasonal N_2O emissions observed (peaks in spring and late summer/early autumn; lower emissions during winter) is similar to other studies on peat soils, where N_2O peaks have been observed in response to mineral N application, cultivation, or post-harvest residue input (Elder and Lal, 2008; Rochette et al., 2010). At our study sites, crop establishment operations (tillage, mineral N application, irrigation, planting) were often practiced together over several days

prior to measuring emissions (Supplementary tables S.1-S.3), so it is difficult to disaggregate the relative importance of individual management practices to peaks of N_2O. Field operations varied in their intensity or application rate (e.g. ploughing depth, fertiliser application rate) and timing relative to the timing of emissions measurements. Given that the response of N_2O peaks to different field operations can vary in duration and intensity, particularly when interacting factors such as rainfall events and soil temperature are considered (e.g. De Klein and Harvey, 2013), it was not possible in this study to reliably disentangle individual effects (e.g. assigning individual peaks to particular fertiliser events). It is also possible that differing fertiliser rates at the three sites due to crop requirements influenced N_2O emissions more than SOM%, although this is dependent on priming effects and whether N supply is in excess of crop demand (e.g. Ye et al., 2016). Therefore, similarly to CO_2 emissions, N_2O emissions should be interpreted as a general indication of magnitude and seasonal patterns at similar sites. To aid refinement of N_2O EFs, we recommend that future research should focus on capturing variability in emissions at a finer temporal scale across a range of SOM contents and cropping regimes, preferably over a multi-year monitoring campaign. This approach reduces the uncertainty associated with missing emissions peaks (resulting in underestimation of cumulative fluxes and soil N loss %), and that associated with linear interpolation of high and low fluxes at a lower resolution temporal scale (resulting in overestimation of cumulative fluxes and soil N loss %). Combining field measurements of key management events and environmental variables with process-based modelling (e.g. Brown *et al*., 2002; Chen *et al*., 2008; Smith *et al*., 2010), or finding suitable proxies for annual emissions (e.g. Ye et al., 2016), offers a further route to improving emissions estimates for intensively managed peat soils.

Methane emissions from temperate and boreal cropped and bare peat soils tend to be low (-0.15 to 0.25 t CO_2-e ha^{-1} y^{-1}, e.g. Elder and Lal, 2008; Kasimir-Klemedtsson et al.,

1997; Maljanen et al., 2002). Whilst these emissions are from cereal crops rather than vegetable crops, the emissions in this study (-0.11 ± 0.04 to 0.10 ± 0.10 t CO_2-e ha^{-1} y^{-1}) are of a comparable magnitude. Such low emissions are typically found at sites such as ours, where top-soils are well-mixed, and moist but unsaturated, indicating predominantly aerobic conditions which favour methanotrophic rather than methanogenic microorganism activity (Le Mer and Roger, 2001).

4.2. Comparison with IPCC default EFs

The most recently published UK National Greenhouse Gas Inventory (Webb et al., 2014) uses the IPCC (2000) default emission factors (EFs) with some UK-specific modifications, to estimate CO_2, N_2O and CH_4 emissions from cultivated Histosols, providing annual estimates equivalent to 3.99 to 46.90 t CO_2-e ha^{-1} y^{-1} carbon dioxide, 3.90 t CO_2-e ha^{-1} y^{-1} nitrous oxide, and 'negligible' methane emissions.

Mean cumulative annual CO_2 emissions in this study were within the upper range estimated using the IPCC method. The soils at the study sites have been under cultivation for at least 50 years, so based on the premise that well-aerated, homogenised peats tend to be less reactive than relatively undisturbed Histosols, lower rates of emission might be expected. However, the characteristics of the peats in this study are intermediate between the description given for the lower IPCC emission rate (< 1 m depth, < 12% SOC) and the higher emission rate (> 1 m depth, > 12% SOC), so our intermediate emissions fit well with the predicted emission factors.

Estimated annual N_2O emissions in this study varied from 1.51 to 15.79 t CO_2-e ha^{-1} y^{-1} on a per-field basis, and in all but one field were two to four times greater than the IPCC-estimated default value. At a national scale, the IPCC (2000) method sums the emissions from peat cultivation with emissions from mineral fertiliser application and crop residue

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application (each 1.25% of applied N). While insufficient crop residue data is available to allow calculation of accurate residue EFs from the study sites, mineral N EFs at the study sites after deducting the IPCC (2000) Histosol EF were estimated at between 0 and 27% of N applied during the annual emission calculation period, or between 0 and 14%, if N applied immediately prior to the first sampling date in April 2011 is included (Supplementary tables S.1-S.3). This assumes that all N_2O emissions were directly caused by mineral N application, but does not take account of crop residue input influences. Given the typically low total N content of the salad crops that are commonly grown on these sites, it is unlikely that residue N would account for a substantial portion of annual emissions, although residue input could substantially contribute to priming effects (Kuzyakov et al., 2000). It is possible therefore that the IPCC default EF of 1.25% of applied N may underestimate emissions at some SOM-rich sites; this requires further clarification.

Comparison of N_2O fluxes at the study sites with estimates of N stocks to 1 m depth (Table 4) suggest that only a small percentage of soil N is lost annually as direct N_2O emission as measured here ($< 0.1\%$ of the total N stock at all sites, or 3.1 to 32.4 kg N ha^{-1}), but that the potential for nitrogen loss via other direct and indirect routes is substantial, with an estimated 7.9 to 10.1 t N ha^{-1} stored in the top 10 cm of soil alone. Total theoretical N loss accompanying C loss during SOM mineralization can be estimated from CO_2 emissions and soil C:N ratios (Tables 2 and 4 respectively). If we assume that CO_2 emissions are generated in the top soil (0-10 cm) only, theoretical N loss from the top soil would be between 245.7 and 363.8 kg ha^{-1} y^{-1} at the SOM_{LOW} site, 311.5 to 617.8 kg ha^{-1} y^{-1} at the SOM_{MED} site, and 326.4 and 661.6 kg ha^{-1} y^{-1} at the SOM_{HIGH} site. Conversely, if we assume that CO_2 is generated from the entire cultivated layer (0-40 cm), theoretical N loss from the top soil would be slightly higher at the SOM_{LOW} site (253.2 to 374.9 kg ha^{-1} y^{-1}), but slightly lower at SOM_{MED} and SOM_{HIGH} sites (308.5 to 611.9 kg ha^{-1} y^{-1}, and 312.6 and 633.6 kg ha^{-1} y^{-1}

respectively). Further, fertiliser N input can be added to these figures to compute theoretical total available N (excluding any residue N inputs). If N_2O EFs are then calculated as a proportion of total available N on a per-field basis, gaseous N_2O losses are estimated at between 0.5% and 5.4% of total available N, i.e. still a relatively low proportion of total available N.

Observed CH_4 emissions (-0.11 to 0.10 t CO_2-e ha^{-1} y^{-1} on a per-field basis) can be considered within the 'negligible' range described by the IPCC (2000). Further modification of methane EFs from the soil surface is therefore unnecessary when predicting emissions from these sites.

4.3. Soil loss rates

Using annual C budget calculations, mean soil bulk density and soil C values from our three study sites (Table 4), cumulative net gaseous C losses (Table 2) represent an average annual soil loss rate of 0.41 to 0.61 cm at the SOC_{LOW} site, 0.33 to 0.60 cm at the SOC_{MED} site, and 0.37 to 0.75 cm at the SOC_{HIGH} site. While these values are low compared with the estimates of 1.10-1.48 $cm-y^{-1}$ between 1982 and 2004 made by Dawson et al. (2010), they do fall within the range of 0.27-3.09 cm y^{-1} presented in long-term monitoring programmes (Hutchinson, 1980; Richardson and Smith, 1977). It is also possible to make a 'back of the envelope' estimation of the time until SOC exhaustion in the region under current management and cropping rotations, from our estimated C stocks (Table 4) and current gaseous CO_2 loss rates (Table 2). The most recent definition of Histosols in emissions inventorying (IPCC, 2006) gives a minimum threshold of 11.6% SOC in more than half of the top 80 cm soil. SOC exhaustion might therefore be thought of as the point at which the soil can no longer be defined as a Histosol, through continual incorporation of SOM from the underlying soil layers into the upper cultivated later (Couwenberg and Hooijer, 2013), and

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subsequent depletion to organo-mineral then mineral soil status. At our sites, SOC% falls below the 11.6% threshold at 40 cm, 70 cm and 90 cm depths for the SOC_{LOW}, SOC_{MED} and SOC_{HIGH} sites respectively (Table 4). Dividing the remaining soil C stock within these profile sections by the current annual CO_2 emission from these sites allows for between c. 80-160 years before SOC depletion (assuming an average 1 m soil depth across the region; Dawson et al., 2010). However it should be noted that this estimate does not account for inputs of crop residues and plug plant compost which may partially restore a small quantity of OM to the soil each year. Conversely, allowing for losses via other routes (wind and water erosion, leaching of DOC, crop adherence, etc.), inter-annual variation in CO_2 emissions and crop and soil management, and the commonly observed decline in soil quality as the soil profile is depleted and underlying mineral material becomes incorporated into the peat substrate (e.g. Hooijer et al., 2012; Kasimir-Klemedtsson et al., 1997), this may be an optimistic estimate. Taking into account the need for increasing fertiliser application and tillage rates to maintain crop quality as soil quality declines over time, the continuing commercial viability of intensively cultivated Histosols of 1-2 m depth may be as little as 50 years.

During SOM mineralization processes, depletion of soil C is also accompanied by N loss, calculated at our study sites as c. 250 to 625 kg N ha^{-1} y^{-1} (Section 4.2). While fluxes of gaseous N_2O to the atmosphere merit attention for mitigation in their own right, the small proportion of theoretical N loss at our sites as gaseous N_2O emissions suggests the potential for substantial losses via additional routes not accounted for in this study, which represent further potential sources of environmental pollution and commercial costs. Reliable estimates of N losses via leaching and erosion from arable peat soils are scarce, and studies accounting for losses from peat adherence to crops are unknown (e.g. Evans et al., 2011). Mulholland et al. (2008) estimated that 20-25% of all N added to terrestrial ecosystems may be lost via rivers, but the proportion of N lost to waterways that is eventually transformed to indirect

N_2O emission is subject to a great deal of uncertainty (IPCC, 2007). Drainage channels may account for the majority of indirect arable N_2O emissions, with a recent study of a catchment in East Anglia comprising mineral and peat soils, estimating that 86% of indirect N_2O emissions originated from drainage channels (Outram and Hiscock, 2012). Applying our top soil C:N ratios (Table 4) to estimated peak soil C losses from wind erosion in the East of England (> 3 t ha^{-1} y^{-1}; Dawson and Smith, 2007) places wind-eroded N at c. 180-210 t ha^{-1} y^{-1}. Each of these losses exhibit 'downstream' effects resulting from soil nutrient loss, water and air pollution, impacting on aquatic and terrestrial ecosystems and human health (e.g. Berglund and Berglund, 2015; Parish et al., 2008).

4.4. Conclusions

The results of this study suggest that the default IPCC (2000) EFs for annual N_2O emissions for cultivated Histosols may underestimate fluxes from intensively produced vegetable cropping systems, while CO_2 and CH_4 emissions were within a comparable range to the IPCC default values. When calculating losses of soil organic matter via GHG efflux to the atmosphere, attention should be given to the variability in emissions experienced under different weather conditions and different cropping regimes, which both vary inter-annually and geographically. Using longer-term studies as the basis of Tier 2 emissions estimates may aid progress towards more accurate quantification of the variability between sites and years. The lower rates of GHG emission-related soil loss estimated from soils of lower organic matter content present a potential problem for policy-makers in incentivising growers to reduce GHG emissions from peat soils of higher organic matter content, although assessing the time-frame of negative commercial impacts of declining soil quality may aid the mitigation process. Determining suitable cost-neutral or cost-negative mitigation measures,

and investigating grower motivation to implement changes to farming practices or alternative land uses which reduce net emissions, merit further attention.

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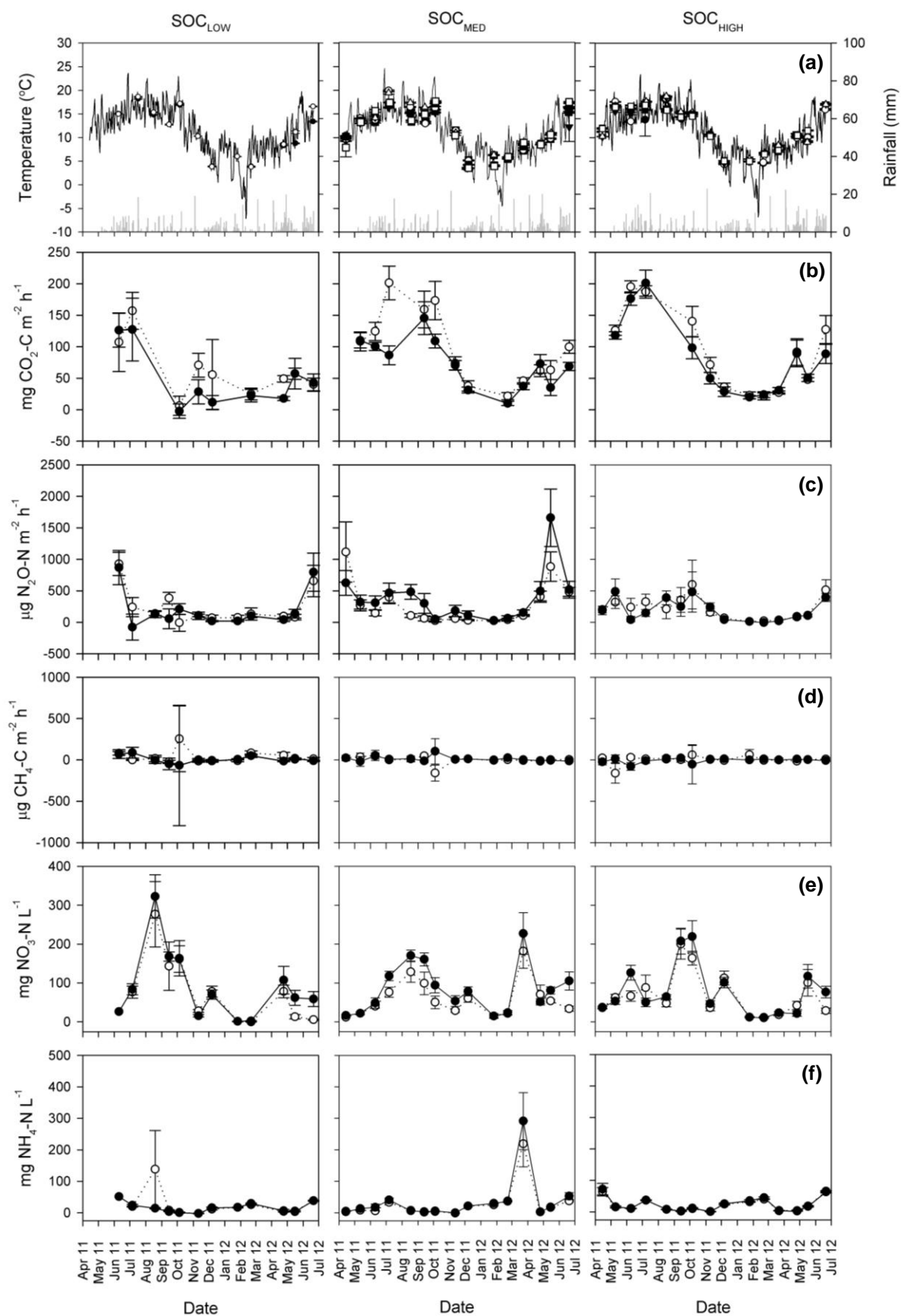
Figure 1

Daily rainfall, air temperature and soil temperature (a); fluxes of CO_2 (b), N_2O (c), and CH_4 (d); and soil NO_3^- (e) and NH_4^+ (f); 14th April 2011 to 21st June 2012: cropped and bare soils at SOC_{LOW} site (~20% SOM content), SOC_{MED} site (~35% SOM content), and SOC_{HIGH} site (~70% SOM content).

In panel (a), MIDAS mean air temperature (°C) is denoted by a solid black line, rainfall (mm) by grey bars; and mean soil temperature by solid black circles (cropped soil, fields 1, 4 and 7), open circles (bare soil, fields 1, 4 and 7), black triangles (cropped soil, fields 2 and 5), open triangles (bare soil, fields 2 and 5), black squares (cropped soil, fields 3 and 6), and open squares (bare soil, fields 3 and 6). In panels (b) to (f), cropped soil values are denoted by a black circle and bare soil values by an open circle.

Figure 2

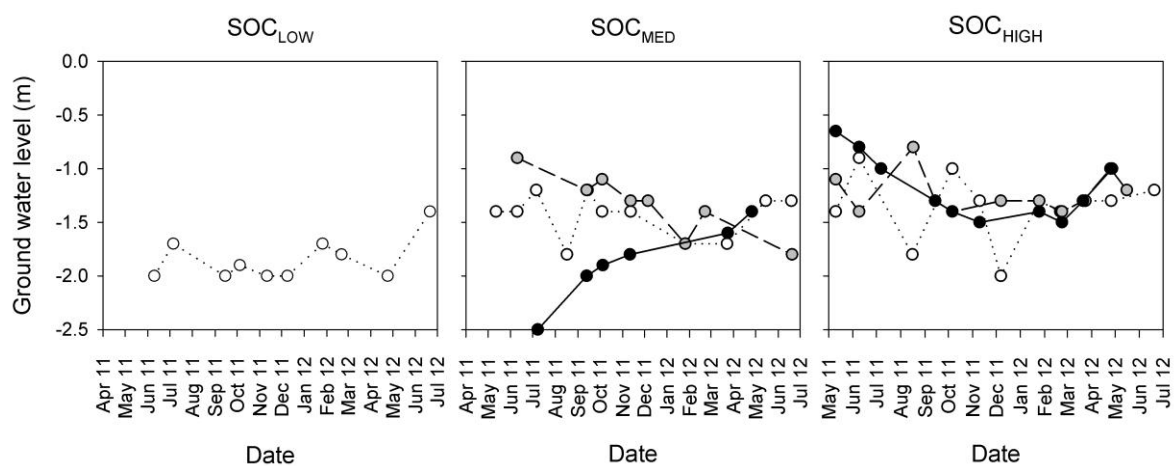Ground water levels measured at drainage ditches bordering study fields; 14th April 2011 to 21st June 2012: SOC_{LOW} site (~20% SOM content), SOC_{MED} site (~35% SOM content), and SOC_{HIGH} site (~70% SOM content).

Separate fields are denoted: at SOC_{LOW} site, field 1 by open circles; at SOC_{MED} and SOC_{HIGH} sites, field 1 by open circles, field 2 by grey circles, and field 3 by black circles.

Table 1

Summary of the significant linear correlations between measured environmental variables

and emissions of CO_2, N_2O and CH_4 in intensively managed horticultural soils.

	Crop	SOM	Soil temp.	Mean air temp.	Air temp.	Daily rainfall	5 d rain	H_2O_{DW}	Water table	NO_3-N	NH_4-N	DIN	Bulk density	Soil pH	Crop p_{AB}
			(°C)	(°C)	(°C)	(mm)	(mm)	(%)	(m)	(mg l^{-1})	(mg l^{-1})	(mg l^{-1})	(g c m^{-3})		(g m^{-2})
CO_2	Crop	20%													
		35%	0.460***	0.472***	0.427***			-0.229***	-0.289***	-0.153**		-0.180***	-0.144**	0.167**	
		70%	0.517***	0.449***	0.385***	-0.189**	0.111*	-0.274***	-0.324***	0.176**			0.125*	-0.214***	0.547***
	Bare	20%					0.223*				0.258*				
		35%	0.390***	0.387***	0.365***			-0.215***			-0.144**				
		70%	0.517***	0.426***	0.434***	-0.212***	0.139*	-0.187**	-0.334***	0.179**			0.125*	-0.244***	
N_2O	Crop	20%	0.197*												-0.661*
		35%	0.129**		0.113*										
		70%	0.216***	0.184***	0.139**	-0.105*			-0.193**	0.137**			0.134**	0.147**	-0.272***
	Bare	20%	0.191*		0.192*										
		35%	0.180***		0.107*			-0.116*		0.094*					
		70%	0.292***	0.232***	0.126**			-0.134*	-0.219***	0.135**				-0.240***	
CH_4	Crop	20%													
		35%													
		70%													
	Bare	20%								0.220*					
		35%			-0.093*					-0.096*		-0.092*			
		70%													

Values are presented as Kendall's tau statistic (τ), with significance levels presented as

* ($p < 0.05$), ** ($p < 0.01$), or *** ($p < 0.001$). Environmental variables presented are: Soil

temp. (soil temperature); Mean air temp. (mean MIDAS daily air temperature); Air temp.

(measured air temperature); Daily rainfall (cumulative MIDAS daily rainfall); 5 d rain

(cumulative MIDAS 5-day rainfall); H_2O_{DW} (Gravimetric soil water content, % of dry weight); Water table (water table depth); NO_3-N (soil extractable nitrate); NH_4-N (soil extractable ammonium); DIN (soil extractable nitrate + ammonium); Bulk density (topsoil bulk density); Soil pH (soil pH); and $Crop_{AB}$ (Crop aerial biomass).

Table 2

Cumulative annual fluxes of CO_2, N_2O and CH_4, and total cumulative greenhouse gas emissions (GWP_{100}) for cropped and bare soils of contrasting organic matter status under horticultural production. The SOC_{LOW}, SOC_{MED}, and SOC_{HIGH} categories correspond to soil organic matter contents of ~20%, ~35% and ~70% respectively. Values represent means ± SEM.

Organic matter status	Cumulative annual fluxes (t CO_2.e ha^{-1} y^{-1})							
	Cropped soil				Bare soil			
	CO_2	N_2O	CH_4	GWP_{100}	CO_2	N_2O	CH_4	GWP_{100}
SOC_{LOW}	19.2 ± 2.7	6.90 ± 2.10	-0.01 ± 0.09	26.1 ± 3.4	13.0 ± 2.4	4.97 ± 0.74	-0.02 ± 0.08	17.9 ± 3.0
SOC_{MED}	30.9 ± 2.5	7.93 ± 0.78	0.04 ± 0.02	38.8 ± 2.4	21.5 ± 1.1	13.9 ± 1.91	0.00 ± 0.01	35.4 ± 2.0
SOC_{HIGH}	28.3 ± 2.3	7.90 ± 2.23	0.01 ± 0.04	36.2 ± 2.1	26.0 ± 2.5	6.66 ± 1.49	0.04 ± 0.05	32.7 ± 2.3

GWP_{100}, Global warming potential over 100 years of combined CO_2, N_2O and CH_4 flux.

Table 3

Two-way analysis of variance of the fixed effects of SOM% and cropping (cropped vs. bare soils) on annual cumulative GWP_{100}, and CO_2 and N_2O emissions (10th Jun 2011 to 9th Jun 2012).

(a) GWP_{100}

Source	df	F-value	P-value
SOM%	2	12.254	<0.001
Cropping	1	4.882	0.031
SOM% × Cropping	2	0.352	0.705

(b) CO_2 [a]

Source	df	F-value	P-value
SOM%	2	8.857	<0.001
Cropping	1	8.395	0.005
SOM% × Cropping	2	1.275	0.286

(c) N_2O [a]

Source	df	F-value	P-value
SOM%	2	5.601	0.006
Cropping	1	0.264	0.609
SOM% × Cropping	2	2.485	0.091

[a] Performed on square-root transformed emissions values.

Table 4

Estimated C and N stocks to 1 m depth at intensively managed horticultural soils of contrasting organic matter status. Values represent means ± SEM.

Soil depth (cm)	Organic matter status														
	SOC$_{LOW}$					SOC$_{MED}$					SOC$_{HIGH}$				
	Bulk density (g cm^{-3})	Total C (%)	C stock (t ha^{-1})	Total N (%)	N stock (t ha^{-1})	Bulk density (g cm^{-3})	Total C (%)	C stock (t ha^{-1})	Total N (%)	N stock (t ha^{-1})	Bulk density (g cm^{-3})	Total C (%)	C stock (t ha^{-1})	Total N (%)	N stock (t ha^{-1})
0 – 10	0.76 ± 0.03	19.1 ± 0.4	145.1 ± 3.1	1.3 ± 0.02	10.1 ± 0.2	0.39 ± 0.01	33.7 ± 0.5	130.7 ± 1.5	2.0 ± 0.03	7.9 ± 0.1	0.35 ± 0.02	38.8 ± 4.2	137.9 ± 19.6	2.4 ± 0.29	8.7 ± 1.2
10 – 20	0.83 ± 0.07	17.9 ± 2.3	145.1 ± 8.3	1.3 ± 0.15	10.5 ± 0.3	0.37 ± 0.002	33.2 ± 0.4	123. 9 ± 1.4	2.0 ± 0.03	7.3 ± 0.1	0.37 ± 0.02	31.0 ± 12.8	110.3 ± 44.5	1.9 ± 0.88	6.5 ± 3.1
20 – 30	0.76 ± 0.01	18.9 ± 1.4	143.0 ± 9.1	1.4 ± 0.10	10.4 ± 0.7	0.39 ± 0.03	44.0 ± 9.0	176.5 ± 46.0	2.6 ± 0.62	10.7 ± 3.2	0.35 ± 0.05	42.8 ± 2.2	150.2 ± 22.1	2.7 ± 0.05	9.6 ± 1.5
30 – 40	0.85 ± 0.05	16.7 ± 2.6	139.9 ± 15.5	1.2 ± 0.16	10.1 ± 0.8	0.38 ± 0.04	45.2 ± 7.5	172.3 ± 40.4	2.7 ± 0.49	10.2 ± 3.2	0.18 ± 0.004	56.1 ± 5.2	101.1 ± 8.5	3.1 ± 0.09	5.6 ± 0.2
40 – 50	0.99 ± 0.02	4.1 ± 0.3	40.8 ± 3.8	0.3 ± 0.01	3.2 ± 0.2	0.27 ± 0.03	39.2 ± 2.7	103.9 ± 6.1	2.4 ± 0.18	6.3 ± 0.4	0.18 ± 0.01	47.8 ± 1.0	87.7 ± 1.1	3.0 ± 0.25	5.5 ± 0.3
50 – 60	0.86 ± 0.04	3.8 ± 0.3	32.7 ± 3.4	0.3 ± 0.02	2.6 ± 0.2	0.22 ± 0.02	46.3 ± 1.7	103.8 ± 11.3	2.5 ± 0.13	5.7 ± 0.6	0.15 ± 0.01	47.2 ± 1.3	72.2 ± 4.7	2.5 ± 0.24	3.8 ± 0.5
60 – 70	0.74 ± 0.09	4.2 ± 1.4	28.6 ± 5.8	0.4 ± 0.12	2.4 ± 0.5	0.66 ± 0.26	18.4 ± 14.0	49.7 ± 15.6	0.8 ± 0.57	2.1 ± 0.5	0.16 ± 0.01	45.5 ± 1.3	72.8 ± 3.4	2.3 ± 0.20	3.8 ± 0.5
70 – 80	0.57 ± 0.11	8.7 ± 3.8	41.7 ± 9.2	0.7 ± 0.33	3.2 ± 0.8	1.06 ± 0.17	3.4 ± 1.6	31.0 ± 14.3	0.2 ± 0.07	1.6 ± 0.5	0.28 ± 0.04	28.8 ± 5.3	74.8 ± 4.2	1.6 ± 0.25	4.3 ± 0.3
80 – 90	0.61 ± 0.10	6.1 ± 1.9	35.3 ± 5.6	0.4 ± 0.09	2.3 ± 0.2	1.36 ± 0.16	0.9 ± 0.4	11.4 ± 4.7	0.1 ± 0.04	1.2 ± 0.5	0.52 ± 0.20	13.6 ± 2.6	63.1 ± 13.9	0.9 ± 0.15	4.0 ± 0.8
90 – 100	nd	nd	nd	nd	nd	1.36 ± 0.21	0.3 ± 0.1	3.7 ± 1.2	0.03 ± 0.02	0.3 ± 0.1	1.36 ± 0.44	9.4 ± 5.2	83.6 ± 9.2	0.6 ± 0.34	5.5 ± 0.5
Total 0 – 100 cm			787.5 [a]		57.0 [a]			907.0		53.1			953.7		57.4

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All bulk density estimates exclude coarse the stone fraction (>2 mm) for the purposes of C and N stock calculations. [a] Total C and N stocks at the SOC_{LOW} site are estimated using values from the 80-90 cm layer for the 90-100 cm layer, since sample collection from the 90-100 cm layer of the SOC_{LOW} site was not possible.