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Agricultural soils: A sink or source of methane across the British Isles?

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Abstract

This study summarizes a large diverse dataset of methane (CH_4) fluxes measured from agricultural sites across the British Isles. A total of 53,976 manual static chamber measurements from 27 different sites were investigated to determine the magnitude of CH_4 fluxes from a variety of agricultural fields across the UK and Ireland. Our study shows that contrary to some studies, agricultural soils (both arable and grassland) are small net emitters of CH_4 rather than sinks. Mean fluxes measured from arable and grassland sites (excluding fertiliser and tillage events) were 0.11 ± 0.06 and 0.19 ± 0.09 nmol m^{-2} s^{-1}, respectively, and were not found to be significantly different (Welch t-test, $p = 0.17$). Using the values reported in this study, we estimate that an annual emission of 0.16 and 0.09 Mt of CO_2-eq is expected from arable and grassland agricultural soils in the UK and Ireland (comparable to 0.3 and 0.7% of the current annual CH_4 emission inventories, respectively). Where CH_4 uptake occurs in soils, it is negligible compared to expected emissions of the application of animal manures and tillage events, which were both found to significantly increase CH_4 emissions in the immediate few days to months after events. Our study highlights that there are significant differences in CH_4 uptake and emissions between sites, and that these differences are partially the result of the moisture content of the soil (i.e., the aerobic status of the soil). We expect uptake of CH_4 to be more prevalent in drier soils where volumetric water content does not exceed 35% and emissions to be exponentially greater where agricultural fields become waterlogged.

Highlights

- This study investigated 53,976 CH_4 flux measurements from 27 sites across the UK
- Our study shows both arable and grassland soils are small net emitters of CH_4
- We estimate annual CH_4 emissions of 0.16 Mt of CO_2-eq from agricultural soils in the UK
- We estimate annual CH_4 emissions of 0.09 Mt of CO_2-eq from agricultural soils in Ireland

KEYWORDS

bioenergy, CH_4, flux, GHG, grassland, greenhouse gas, manure, tillage

1 | INTRODUCTION

Due to anthropogenic activities, concentrations of the powerful greenhouse gas (GHG) methane (CH_4) in the atmosphere have risen by over a factor of 2.5, from 772 (pre-industrial era) to 1,866 ppb (Dlugokencky, 2020) presently. Atmospheric CH_4 continues to rise at a rate of approximately 22 Tg every year, as global sources are larger than sinks (also known as uptake, or negative flux) (Ehhalt et al., 2001). The majority of naturally occurring emissions of CH_4 can be attributed to the biogenic processes of methanogens (methane-emitting microorganisms), which are predominant in anaerobic areas rich in organic carbon such as wetlands (Conrad, 2009; Segers, 1998). However, human activities have altered the magnitude of many natural processes over the past two centuries, as well as creating new sources of significant magnitude, which have grown exponentially as the human population has increased. Current estimates show that anthropogenic sources of CH_4 now contribute more to emissions than natural sources, and are responsible for approximately 60% of global emission inventories (Karakurt et al., 2012; Saunois et al., 2016). The largest sources of anthropogenic CH_4 emissions include energy production (i.e., fossil fuel extraction and gas flaring), rice agriculture, biomass burning and ruminant livestock.

Reduction of atmospheric concentrations of CH_4 is required if we are to meet the Paris Agreement target of keeping global warming below 1.5 °C (Nisbet et al., 2020). Due to the relatively short lifetime of CH_4 in the atmosphere (approximately 9 years), reductions in CH_4 could have a significant impact on short-term trends in global warming (Dlugokencky et al., 2011; Collins et al., 2018). The primary removal (sink) of CH_4 from the atmosphere occurs through the abiotic reaction of CH_4 with hydroxyl radicals in the troposphere (Ehhalt, 1974). Globally, this process removes approximately 500 to 600 Tg of CH_4 from the atmosphere annually (Saunois et al., 2016). Another smaller and less well-quantified CH_4 sink is that of terrestrial soils, which contributes to the removal of approximately 30 to 60 Tg of CH_4 annually (Cicerone and Oremland, 1988; Ridgwell et al., 1999). Uptake of CH_4 into soils occurs primarily in aerobic conditions, when CH_4 is oxidized by methanotrophic microorganisms (Angle et al., 2017; Holmes et al., 1999).

Methanogenic bacteria produce CH_4 as a product of anaerobic metabolism during the degradation of organic materials in the soil (Stams, 1994). Conditions best suited to these bacteria are anaerobic soils with a rich supply of organic carbon, such as natural wetlands in the arctic and tropics, and rice paddies (Ehhalt et al., 2001). As methanogenic and methanotrophic (CH_4 consuming) microorganisms are both present in soils, both CH_4 production and oxidation occur simultaneously. Emissions of biogenic CH_4 from soils would be significantly larger if not for the methanotrophic microorganisms that oxidize CH_4 before it escapes the soil profile, reducing CH_4 released by up to 99% of the primary production (Reeburgh, 2007). The heterogeneous nature of soils allows for both anaerobic and aerobic microsites to exist in close proximity, thus the net flux from a particular location is dependent upon a variety of conditions within the soil profile, such as aeration, temperature, pH and available nutrients (Savage et al., 1997; Smith et al., 2000).

Methane currently accounts for 11% of all GHG emissions in the UK (BEIS, 2018) and 23% of all GHG emissions in Ireland (EPA, 2020) (CO_2eq, CO_2 equivalent). Of the estimated 51.5 Mt CO_2eq of CH_4 emitted across the UK and 14.0 Mt CO_2eq of CH_4 emitted across Ireland, 49 and 93% are attributed to agricultural activities, respectively. These emissions are predominantly the result of ruminant livestock production (CH_4 released via enteric fermentation) and manure storage (BEIS, 2018; EPA, 2020). Currently, there is global pressure to reduce GHG emissions in supply chains to reduce the impacts of global warming, and as a

result, policymakers are seeking opportunities to reduce CH_4 emissions from intensive agricultural activities. One topic of contention in GHG accounting at the farm level is whether intensively managed arable and grassland soils should be included as an offset of CH_4 emissions due to the oxidation uptake that is expected to occur (Chianese et al., 2009; Oertel et al., 2016; Jackson et al., 2020). Although wet soils high in organic matter are considered a net source of CH_4 emissions in the UK (Levy et al., 2012), mineral soils commonly used for intensive agriculture are often recognized as a sink (Dutaur and Verchot, 2007; Castaldi et al., 2007). However, the effects of intensive tillage/mechanical agitation of soils, irrigation, compaction (machinery and livestock trampling) and chemical applications (mineral/organic fertiliser, lime, pesticides, etc.) and resultant disturbed soil conditions across managed agricultural land on CH_4 emissions are currently poorly understood. Due to the small magnitude of CH_4 emissions and uptake typically observed from mineral soils, it can be difficult to identify drivers and quantify fluxes from different soil types and management practices (Kim et al., 2010; Segers, 1998). A large number of studies have investigated the net exchange of CH_4 for agricultural soils (e.g., Dengel et al., 2011; Le Mer and Roger, 2001; Levy et al., 2012; Smith et al., 2000) with varying conclusions and some suggesting that uptake of CH_4 by agricultural soils is substantially less than in unmanaged natural soils (Nesbit and Breitenbeck, 1992; Ridgwell et al., 1999).

Agriculture currently accounts for over 70% of land use in the UK, 65% of land use in Ireland and approximately half of the land use in EU countries. This study aims to better understand the reality of CH_4 production and uptake from agricultural soils in temperate northern hemisphere soils such as those found across the British Isles. Here we analyse a large dataset of CH_4 measurements carried out over the past 15 years by the authors. The dataset includes 53,976 manual static chamber measurements from across 27 separate fields, covering a variety of common crop types (arable, plantation and grassland) and agricultural practices such as tillage and fertiliser application. With this dataset, we aim to investigate CH_4 fluxes from intensively managed agricultural soils and establish whether these soils should be classed as a net sink or source of CH_4 for the purposes of GHG accounting across the British Isles.

2 | MATERIALS AND METHODS

2.1 | Flux data

Data used in this study were collected from numerous research groups within the UK and Ireland that specialize in the measurements of GHG from agricultural sources. The majority of the data was originally generated during various research projects with the aim of quantifying emissions of gases from soils, but primarily with a focus on nitrous oxide (N_2O). The method is the same for measuring fluxes of both CH_4 and N_2O and both gases are typically measured in tandem; thus a largely unused and unreported dataset was generated for CH_4 fluxes.

A total of 53,976 manual static chamber measurements from 27 different sites across the British Isles were collated (Table 1). These measurements were all carried out using the principle of flux chamber methodology, whereby an enclosed chamber (of 20 to 40 L volume) was inserted into the soil to form an air-tight seal. During closure of the chamber, gases were allowed to accumulate within the sealed volume, and gas samples were manually extracted from the chamber via a syringe and tap. Samples were analysed on different gas chromatography (GC) instruments based across the UK and Ireland, fitted with flame-ionization detector (FID) and electron capture detector (ECD) devices, which allow concentrations of carbon dioxide (CO_2), CH_4 and N_2O to be quantified at the same time. The rate of exchange of a particular gas between the soil and the atmosphere was then determined using Equation (1):

$$F = \frac{dC}{dt}.\frac{\rho V}{A}, \quad (1)$$

where F is the gas flux from the soil (nmol m^{-2} s^{-1}), dC/dt is the rate of change in the concentration in time in nmol mol^{-1} s^{-1} estimated by linear regression, ρ is the density of air in mol m^{-3}, V is the volume of the chamber in m^3 and A is the surface area enclosed by the chamber in m^2.

Although based on a similar principle, the flux chamber measurement methodology used for the different datasets varied between research groups (see Table 1 publications list for further details of methodology). Chamber size and sampling procedures differed between studies, but fluxes remain consistent as a representation of emissions from a given surface area of the soil over a given period of time (as described in Equation (1)). More efforts have been focused on improving the detection limit of the measurement methodology in recent years by increasing the number of samples taken per chamber from a minimum of one to a standard procedure which requires at least four. A larger number of samples reduces the uncertainty in the regression fit of concentration change over a set period of time (dC/dt), which is recognized as the largest source of error in chamber measurements (Levy et al., 2011). The least precise method is that of taking

TABLE 1 A summary of the origins and measurement methodology of the CH_4 flux data presented in this study (*n* is the number of measurements recorded at each site)

Site	Coordinates Lat/long (°)	Year of study	Crop system	*n*	No. gas samples	Experiment/ management details	Related publication
Arable							
Norfolk	52.6, 0.9	2005	Fallow	56	2	Pigs present on fallow ground	Unpublished
Salisbury	51.1, 1.8	2006	Fallow	65	2	Pigs present on fallow ground	Unpublished
Surrey	51.3, 0.6	2006	Fallow	33	2	Pigs present on fallow ground	Unpublished
Lincolnshire (a)	53.1, −0.5	2008–2010	Rapeseed	110	3	Mineral N fertiliser applied	Drewer et al. (2011)
Lincolnshire (b)	53.1, −0.5	2008–2010	Wheat	109	3	Mineral N fertiliser applied	Drewer et al. (2011)
East grange (a)	56.0, −3.6	2012–2014	Barley	227	4	Mineral N fertiliser applied	Drewer et al. (2017)
Boghall	55.9, −3.2	2013–2014	Wheat	1792	2	Cattle slurry application	Bell et al. (2016); Thorman et al. (2020)
North Wyke (a)	50.8, 3.9	2017	Wheat	496	4	Organic fertiliser (food waste digestate)	Sánchez-Rodríguez et al. (2018)
BE plantations							
Lincolnshire (c)	53.1, −0.4	2008–2010	Miscanthus	114	3	GHG budget of bioenergy crops, no N fertiliser applied	Drewer et al. (2011)
Lincolnshire (d)	53.1, −0.3	2008–2010	Willow	114	3	GHG budget of bioenergy crops, no N fertiliser applied	Drewer et al. (2011)
East grange (b)	56.0, −3.6	2012–2014	Scots pine	184	4	GHG budget of bioenergy crops, no N fertiliser applied	Drewer et al. (2017)
East grange (c)	56.0, −3.6	2012–2014	Willow	210	4	GHG budget of bioenergy crops, no N fertiliser applied	Drewer et al. (2017)
Grasslands							
Cow Park	55.9, 3.2	2003	Silage crop	83	2	Urine/dung application	Jones et al. (2005)
Norfolk	52.6, 0.9	2005	Pig grazed	59	2	Pigs present on grassland	Unpublished
Salisbury	51.1, 1.8	2006	Pig grazed	61	2	Pigs present on grassland	Unpublished
Surrey	51.3, 0.56	2006	Pig grazed	36	2	Pigs present on grassland	Unpublished
Easter Bush (a)	55.9, 3.2	2006–2007	Sheep grazing	704	2	Mineral N fertiliser	Skiba et al. (2013)
Easter Bush (b)	55.9, 3.2	2012–2014	Silage crop	701	2	N fertiliser applied (AN)/ impact of tillage event	Drewer et al. (2017)
Easter Bush (c)	55.9, 3.2	2016	Silage crop	650	4	Mineral N fertiliser applied (AN, urea, urea with inhibitor)	Cowan et al. (2019
Easter Bush (d)	55.9, 3.2	2016	Sheep grazed	811	4	Mineral N fertiliser (urea)	Maire et al. (2018)
House O'Muir	55.9, 3.3	2008	Silage crop	80	2	Mineral N fertiliser applied	Unpublished
Crichton	55.1, 3.6	2012–2015	Silage crop	8,586	2	Urea/slurry applied	Hargreaves et al. (2019)

(Continues)

TABLE 1 (Continued)

Site	Coordinates Lat/long (°)	Year of study	Crop system	*n*	No. gas samples	Experiment/ management details	Related publication
East Grange (d)	56.0, −3.6	2012–2014	Silage crop	184	4	Mineral N fertiliser applied	Drewer et al. (2017)
Boghall Glen	55.5, 3.1	2015–2016	Mixed grazing	559	2	GHG budget of dung and urine application	Unpublished
Kirkton	56.4, 4.7	2015–2016	Mixed grazing	580	2	GHG budget of dung and urine application	Unpublished
North Wyke (b)	50.8, 3.9	2016	Silage crop	704	4	Mineral N fertiliser applied (AN, urea, urea with inhibitor)	Carswell et al. (2019)
Upper Joiner	55.9, 3.2	2017	Silage crop	544	4	Mineral N fertiliser applied (AN, urea, urea with inhibitor)	Cowan et al. (2019)
Johnstown Castle (a)	52.5, −6.5	2017	Silage crop	1,278	4	Mineral N fertiliser (calcium ammonium nitrate [CAN]) and urine application	Maire et al. (2020)
Johnstown Castle (b)	52.5, −6.5	2018	Cattle grazing	2,479	4	Mineral N fertiliser applied (CAN)	Unpublished
Johnstown Castle (c)	52.5, −6.5	2013–2015	Silage crop	15,874	4	Range of mineral N fertiliser and urine/dung applied	Harty et al. (2016); Krol et al. (2016)
Moorepark	52.2, −8.2	2013–2015	Silage crop	16,513	4	Range of mineral N fertiliser and urine/dung applied	Harty et al. (2016); Krol et al. (2016)

several samples of background air as an 'ambient' to act as a t = 0 sample for all chambers, and taking one sample per chamber after a fixed period of time (typically 40–60 min). The uncertainty in a two-point regression fitting cannot be calculated on an individual basis, limiting quality control of data collected using this method. Here, we determine the range of uncertainty for CH_4 measurements to be in the order of ±0.52 nmol m^{-2} s^{-1}, although in reality this value will vary depending upon instrument capability and maintenance, user experience and execution of the chamber measurements themselves (i.e., leaking, disturbance of soil, timing of measurement). Where four (or more) samples are taken, this reduces the uncertainty in dC/dt significantly; thus the detection limit is estimated to be 0.17 nmol m^{-2} s^{-1} for this study, based on unpublished laboratory work carried out by the authors investigating quality control of flux chamber measurements. However, the same caveats exist in that this value is a generalization and in reality, detection limits of a particular method are dependent on many factors that will vary across research groups and experimental methodology. In this study, these approximate detection limits (0.17 to 0.52 nmol m^{-2} s^{-1}) were used to determine the significance of the individually observed fluxes.

2.2 | Site classification

The field sites from which the collated data originate are classified in this study as arable, bioenergy (BE) plantations or grasslands (see Table 1). Fields in which cereal crops were grown and ground that was left unsown for a season (fallow) are classed as arable. These fields had a history of intensive tillage and were typically tilled once or twice per year depending on crop rotation. BE plantations represent fields in which crops (trees) were planted on soils that had until recently been classified as arable. As such, these fields did not have soils similar to those associated with traditional commercial forests. The crops planted in the BE plantations were short rotation coppice willow, short rotation forestry scots pine and *Miscanthus* perennial grass, all commonly used for bioenergy purposes. Where grassland was grazed or grown as a silage

TABLE 2 Site data and soil properties of field sites from which CH_4 flux data were collected in this study. Annual rainfall and soil temperature are based on averages of 10 years of measurement data (where available)

Site	Crop type	Soil type	Soil pH	Bulk density (g cm^{-3})	Annual rain (mm)	Annual soil temperature (°C)
Arable						
Norfolk	Pig grazed	/	/	/	649	/
Salisbury	Pig grazed	/	/	/	743	/
Surrey	Pig grazed	/	/	/	1,405	/
Lincolnshire (a)	Rapeseed	Fine loam over clay	5.6[a]	1.37	936	8.2
Lincolnshire (b)	Wheat	Fine loam over clay	5.4[a]	1.37	936	8.2
East Grange (a)	Barley	Clay loam	6.5	1.57	582	9.9
Boghall	Wheat	Sandy loam	6.0	1.05	849	9.0
North Wyke (a)	Wheat	Silt clay loam	5.8	0.75	1,107	10.0
BE plantations						
Lincolnshire (c)	Miscanthus	Fine loam over clay	6.2[a]	1.53	936	8.2
Lincolnshire (d)	Willow	Fine loam over clay	5.0[a]	1.41	936	8.2
East Grange (b)	Scots pine	Clay loam	7.1	1.47	582	9.9
East Grange (c)	Willow	Clay loam	7.1	1.38	582	9.9
Grasslands						
Cow Park	Silage crop	Sandy clay loam	6.4	1.1	921	12.2
Norfolk	Pig grazed	/	/	/	649	/
Salisbury	Pig grazed	/	/	/	743	/
Surrey	Pig grazed	/	/	/	1,405	/
Easter Bush (a)	Sheep grazing	Clay loam	5.1[a]	1.19	921	9.0
Easter Bush (b)	Silage crop	Clay loam	5.5[a]	1.19	921	9.0
Easter Bush (c)	Silage crop	Clay loam	5.5[a]	1.19	921	9.0
Easter Bush (d)	Sheep grazed	Clay loam	5.4[a]	1.19	921	9.0
House O'Muir	Silage crop	/	6.0	0.27	897	9.0
Crichton	Silage crop	Silt clay loam	6.3	1.20	1,120	10.0
East Grange (d)	Silage crop	Clay loam	5.6	1.49	582	9.9
Boghall Glen	Mixed grazing	Organic/humic ranker	4.6	0.86	980	9.0
Kirkton	Mixed grazing	Organic/podzolic ranker	5.6	0.77	2,528	7.9
North Wyke (b)	Silage crop	Silt clay loam	5.8	0.87	1,107	10.0
Upper joiner	Silage crop	Clay loam	6.2[a]	1.20	921	9.0
Johnstown Castle (a)	Silage crop	Drained sandy loam	6.4	1.01	941	11.4
Johnstown Castle (b)	Cattle grazing	Drained sandy loam	6.4	1.01	941	11.4
Johnstown Castle (c)	Silage crop	Drained sandy loam	5.6	1.01	941	11.4
Moorepark	Silage crop	Sandy loam	5.4	0.98	1,050	10.0

[a]Correction factor applied as outlined in Equation (2).

crop for harvesting (or both), the data are classed as grassland. These sites were less frequently tilled (usually more than 5 years between tillage events), and some grassland sites in upland or boggy locations were only managed extensively (e.g., frequent livestock grazing).

2.3 | Site data

Soil data were collected for each of the different field sites used in this study where available. At the site level, soil pH, bulk density, annual rainfall and temperature data were available for most sites (Table 2). Data were also

collated where measurements of soil properties had been taken in close proximity to flux chambers during or shortly after flux measurements. The three variables most often recorded during gas measurements were volumetric soil moisture, soil pH and soil temperature. The volumetric soil moisture was the most commonly collected parameter across the experiments for the purpose of relating CH_4 fluxes to soil moisture contents (water-filled pore space was not commonly available). *In situ* measurements were taken using hand-held moisture probes, which were inserted to 5–10-cm depth of the soil (typically using an ML3 ThetaProbe Soil Moisture Sensor, Delta-T Devices, Cambridge, UK, or Campbell Hydrosense II, Campbell Scientific, Inc., Logan, UT, USA). A variety of hand-held temperature probes were used at the different sites to measure soil temperature *in situ*. Soil temperature probes would be left for approximately 2 to 10 min in the soil to reach equilibrium temperature before taking a reading. Soil pH was measured by taking a soil sample (0–10 cm deep) from the field for laboratory analysis (see Table 1 publications list for further details of the methodology). Where pH measurements were made in water rather than calcium chloride ($CaCl_2$), a conversion factor was applied to normalize the methods. Due to the unavailability of electrical conductivity (EC) measurements, the linear conversion of Ahern et al. (1995) was used, which is representative of soils ranging from 4.8 to 8.5 (Equation (2)):

$$pH_{Ca} = (0.999 \times pH_{H2O}) - 0.877\,. \quad (2)$$

2.4 | Statistical analysis

Data in this study were analysed using the statistical software 'R' (R Core Team, 2017) and presented using the 'ggplot2' package (Wickham, 2016). Where 95% confidence intervals (95% CIs) are presented, these were estimated by taking the standard error of a sample population and multiplying by 1.96, assuming a ''normal'' statistical distribution of data. The ''leaps'' package for R was used to perform stepwise regression to find the best-fitting model, based on the Akaike information criterion (AIC) (Lumley, 2015). The AIC is a measure of model goodness-of-fit derived from information theory, widely used in model selection (Burnham and Anderson, 2004). For a set of candidate models, the model with the lowest AIC value represents the best choice, given the trade-off between model likelihood and complexity.

3 | RESULTS

3.1 | Flux data

A total of 53,976 manual flux measurements collected using standard static chamber methods are reported in this analysis (Figure 1a), 93% of which are from grassland soils (Figure 1d). The maximum CH_4 emission recorded from an individual measurement was 303.7 nmol m^{-2} s^{-1}, whereas the largest negative flux (uptake) was −19.4 nmol m^{-2} s^{-1}. However, observations of this scale are rare in this dataset and the vast majority of measurements were near zero, with a 95% quantile range of −1.44 to 2.29 nmol m^{-2} s^{-1} (Figure 1). The mean of all fluxes measured from the grassland sites was 0.62 ± 0.21 nmol m^{-2} s^{-1}, whereas the means of all fluxes measured from the arable and BE plantation sites were 0.11 ± 0.06 and -0.05 ± 0.02 nmol m^{-2} s^{-1}, respectively. The large difference between grassland emissions and other management types is partly due to the inclusion of high flux data from one grassland site after a tillage event and contributions from sites after organic fertiliser applications (discussed below), and with their exclusion, the mean flux measured from grassland sites is 0.19 ± 0.09 nmol m^{-2} s^{-1}. With the exclusion of fluxes measured after the tillage event and application of organic fertilisers, differences between fluxes measured from grassland and arable sites were not found to be statistically significant (Welch *t*-test, $p = 0.17$); however, emissions measured from the plantation fields were found to be significantly different to both grassland and arable fields ($p < 0.001$ for both).

The mean value of all CH_4 fluxes measured is 0.58 nmol m^{-2} s^{-1}; however, the majority of observations were negative, with 62% of all fluxes reporting uptake of CH_4 into the soil. The positive mean values are the result of a skewed dataset, with a log-normal distribution tending toward higher fluxes (Figure 1a). If the measurements after manure application (up to 30 days) and the impacts of tillage and subsequent large flux observations (Drewer et al., 2017) are removed from the dataset, then the mean value of the reported CH_4 flux falls to 0.16 ± 0.08 nmol m^{-2} s^{-1} (heavily dominated by grassland sites). Splitting the data by field site reveals that there are significant differences between different fields under the same management type. Specific trends also appear, with uptake most likely to be observed in BE plantation fields and uptake more likely at arable than grassland sites (Table 3). Statistically significant uptake of CH_4 was observed at three of the four plantation sites, three of the eight arable sites and three of the 19 grassland sites. Assuming a global warming potential (GWP) of

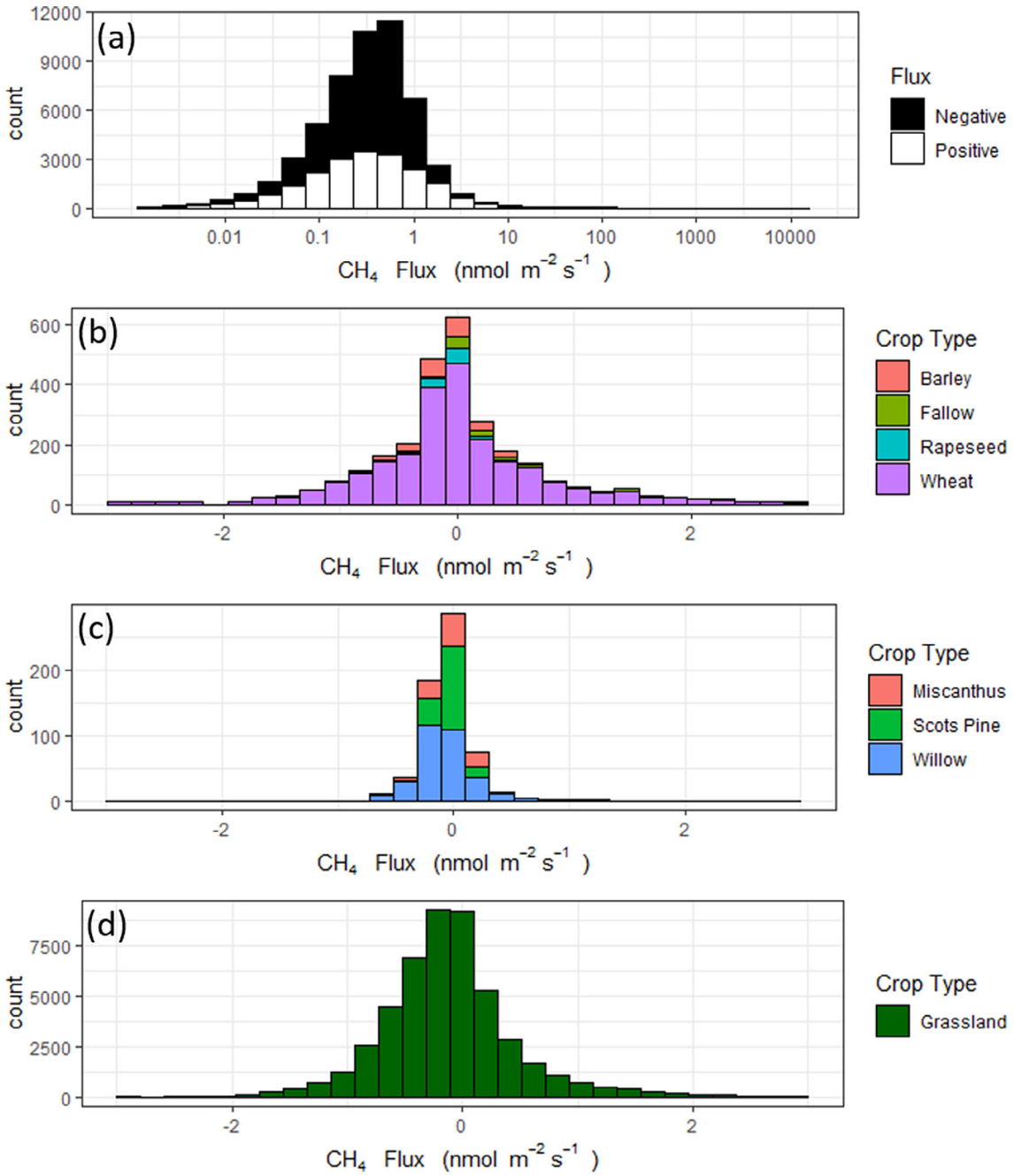

FIGURE 1 (a) Histogram of all CH_4 fluxes carried out using static chambers. (b) Histogram of all CH_4 fluxes measured in arable studies. (c) Histogram of all CH_4 fluxes measured in plantation studies. (d) Histogram of all CH_4 fluxes measured in grassland studies. Histograms 1b, 1c and 1d are limited to a range of −3 to 3 nmol m^{-2} s^{-1}, where the vast majority of observations occurred [Color figure can be viewed at wileyonlinelibrary.com]

28 over 100 years (IPCC 2014), we present the estimated annual flux in units of g-CO_2eq m^{-2}. Although CH_4 emission or uptake was shown to be statistically significantly different from zero in some cases, estimated annual fluxes of CH_4 remained relatively small in terms of GWP, with the largest annual emission of 31.9 g CO_2eq m^{-2} (Kirkton) and largest annual uptake of −3.49 g CO_2eq m^{-2} (House O'Muir) (Table 3).

3.2 | Impact of soil conditions

Direct linear correlations between volumetric soil moisture content, pH and soil temperature measured in tandem with flux measurements (excluding tillage and fertiliser application experiments) were each found to individually correlate with CH_4 flux in a statistically significant manner ($p < 0.005$); however, R^2 values when

TABLE 3 A summary of mean fluxes and 95% confidence intervals (CIs) reported for each of the field management types at the different field sites. Annual global warming potential (GWP) estimates are presented as a multiplication of the mean flux, assuming a GWP of 28 over 100 years. Where uptake is larger than the 95% CI the site is designated as 'uptake significant'

Site	Crop type	Mean CH_4 flux (nmol m^{-2} s^{-1})	95% CIs (±)	Annual GWP (g CO_2eq m^{-2})	Uptake significant
Arable					
Norfolk	Fallow	1.31	0.62	13.84	
Salisbury	Fallow	1.02	0.38	10.77	
Surrey	Fallow	1.39	0.20	14.78	
Lincolnshire (a)	Rapeseed	−0.07	0.03	−0.74	✓
Lincolnshire (b)	Wheat	−0.09	0.02	−0.92	✓
East Grange (a)	Barley	−0.09	0.02	−0.98	✓
Boghall	Wheat	0.05	0.04	0.58	
North Wyke (a)	Wheat	0.16	0.06	1.71	
BE plantations					
Lincolnshire (c)	Miscanthus	−0.04	0.02	−0.45	✓
Lincolnshire (d)	Willow	−0.11	0.03	−1.20	✓
East Grange (b)	Scots pine	−0.03	0.01	−0.36	✓
East Grange (c)	Willow	−0.03	0.03	−0.27	
Grasslands					
Cow Park	Silage crop	0.02	0.02	0.16	
Norfolk	Pig grazed	1.27	0.59	13.48	
Salisbury	Pig grazed	0.08	0.21	0.87	
Surrey	Pig grazed	0.59	0.36	6.25	
Easter Bush (a)	Sheep grazing	0.08	0.02	0.81	
Easter Bush (b)	Silage crop	2.84	0.47	30.09	
Easter Bush (c)	Silage crop	0.02	0.10	0.26	
Easter Bush (d)	Sheep grazed	0.19	0.09	2.01	
House O'Muir	Silage crop	−0.33	0.04	−3.49	✓
Crichton	Silage crop	0.87	0.08	9.27	
East Grange (d)	Silage crop	−0.03	0.01	−0.28	✓
Boghall Glen	Mixed grazing	1.58	0.48	16.72	
Kirkton	Mixed grazing	3.01	0.76	31.89	
North Wyke (b)	Silage crop	2.67	0.71	28.31	
Upper joiner	Silage crop	0.82	0.56	8.65	
Johnstown Castle (a)	Silage crop	−0.15	0.01	−1.57	✓
Johnstown Castle (b)	Cattle grazing	−0.11	0.12	−1.12	
Johnstown Castle (c)	Silage crop	1.07	0.34	11.35	
Moore Park	Silage crop	−0.01	0.04	−0.14	

using individual or multiple linear regression remained small (<0.1). As the majority of measurements fell below the detection limit of the chamber method, comparisons were highly scattered and the meaningful use of more complex statistical comparisons was limited. Trends in the data were clear for both soil moisture content and temperature (Figure 2). Where soils were very dry, uptake was more likely, with emissions typically increasing as soil moisture increases. Fluxes appeared to rise sharply as volumetric soil moisture reached 85% (Figure 2a). There also appeared to be a trend of uptake increasing as soils got warmer, although soil moisture

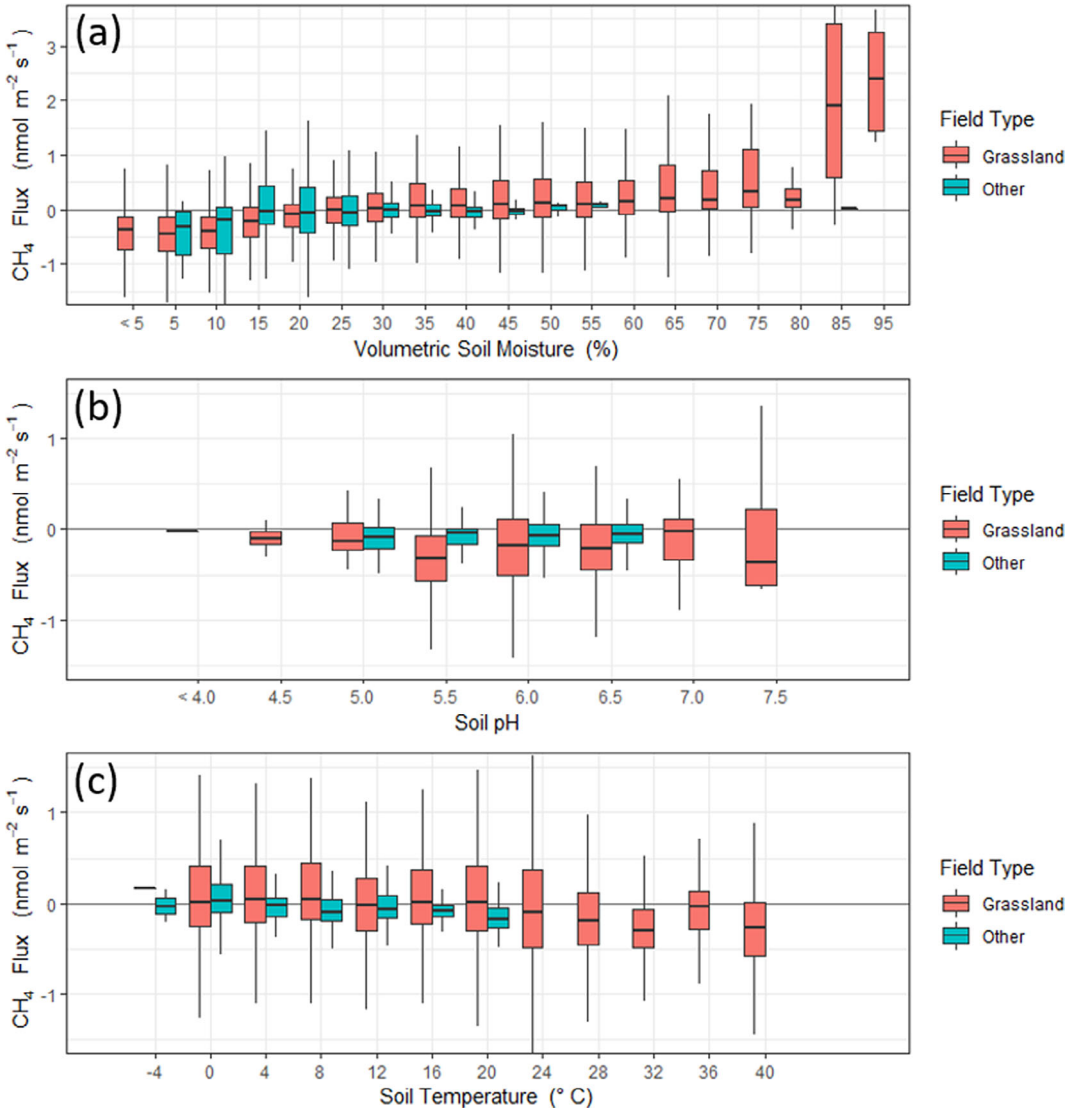

FIGURE 2 Fluxes of methane (CH_4) grouped into statistical bins based on (a) volumetric soil moisture, (b) soil pH and (c) soil temperature, measured in tandem with chamber measurements at each chamber location. Plots show the median values for each data bin, with hinges on the 25% and 75% quantiles [Color figure can be viewed at wileyonlinelibrary.com]

and soil temperature were strongly correlated ($R^2 = 0.3$). When using multiple linear regression, with all soil properties as variables, soil moisture retained statistical significance, whereas soil temperature and pH both became insignificant as predictors ($p > 0.1$). This suggests that the pattern observed between CH_4 flux and soil temperature was almost entirely due to the change in soil moisture content, and not a direct reaction to the temperature change.

At the site level, all measured variables were averaged for comparisons (excluding tillage and fertiliser application experiments). Using multiple linear regression, and rated by AIC value, we investigated the model that best explained the variability in CH_4 fluxes (Figure 3a). Univariate linear regression between CH_4 flux and annual rainfall accounted for 31% of the variance between the field sites; however, this could be slightly improved upon by including the other known soil properties, which then accounted for 38% of the variance between sites (Figure 3b). Site annual rainfall was the only statistically significant predictor of CH_4 flux at the site scale when using multiple linear regression and no clear patterns emerged between other site variables and CH_4 emissions when comparing differences between the sites.

3.3 | Impacts of tillage

Although tillage would have occurred annually at most of the arable sites, and less frequently (every ~5 years)

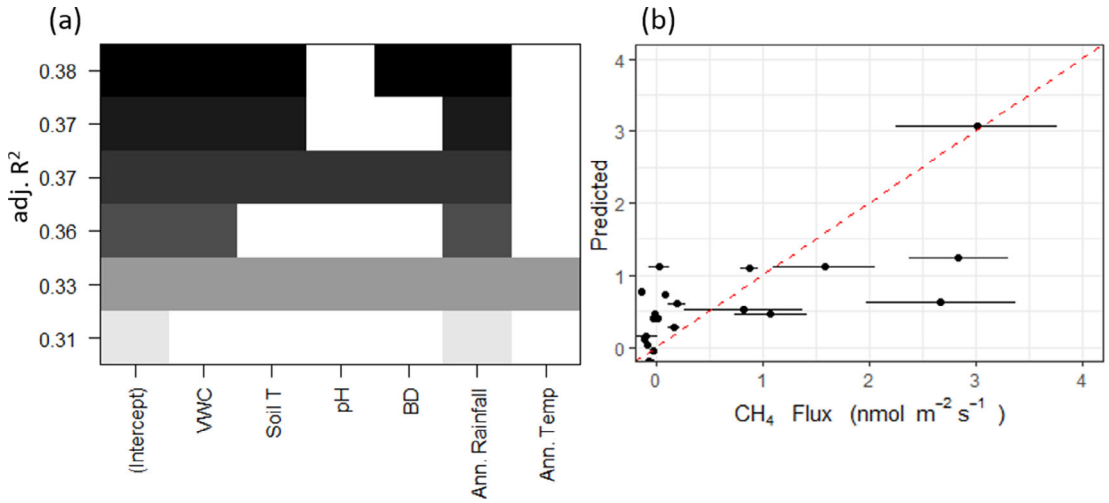

FIGURE 3 Multiple linear regression was applied to correlate CH_4 flux with volumetric soil moisture (VWC), soil temperature (soil T), pH, bulk density (BD), annual rainfall and annual temperature at each of the field sites in this study. (a) Adjusted R^2 values (adjr. R^2) of each of the variables in the multiple regression fit; where the variable is shaded indicates its inclusion within the model to achieve the adjr. R^2on the Y axis, with inclusion based on minimizing the AIC value. (b) The resulting predicted fit (R^2 of 0.38 in 3a) was plotted against measured CH_4 fluxes for each site (95% confidence intervals [CIs] included). The 1:1 ratio was added to aid comparison (red) [Color figure can be viewed at wileyonlinelibrary.com]

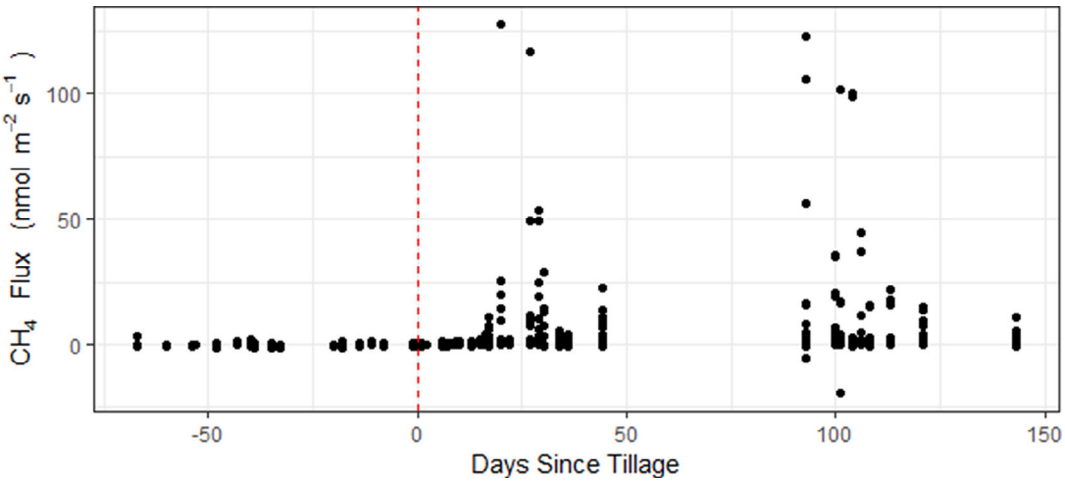

FIGURE 4 Tillage events occurred at two grassland fields at the Easter Bush Field site, one in 2012 and one in 2014 (as reported in Drewer et al., 2017). Data are shown for the measurement period March to October, 2012. After tillage, exponentially large fluxes of CH_4 were observed from some of the chamber sites for several months [Color figure can be viewed at wileyonlinelibrary.com]

on intensively managed grasslands (or not at all on upland hill fields), such data were not available for most of these studies. Chambers are often removed from sites during any tillage operations, and as most of the experiments aimed to study the impact of nitrogen fertiliser application, tillage was deliberately avoided. The exception to this was one study that specifically aimed to investigate the impacts of tillage (Drewer et al., 2017). This study reveals that after tillage events, relatively large fluxes of CH_4 can be released from an otherwise low CH_4 system (Figure 4). The results from this study showed that during these large emission events, the data shifts from a normal distribution of flux data close to zero, to a log-normal distribution, but retains many values close to zero and negative measurements were still observed even when exponentially high fluxes occur in the same field (Figures 4 and 5). Although rainfall may have played some part in the increase in emissions after tillage, CH_4 fluxes are exponentially higher from the tilled field than from the adjacent untilled field in this study (see Drewer et al., 2017).

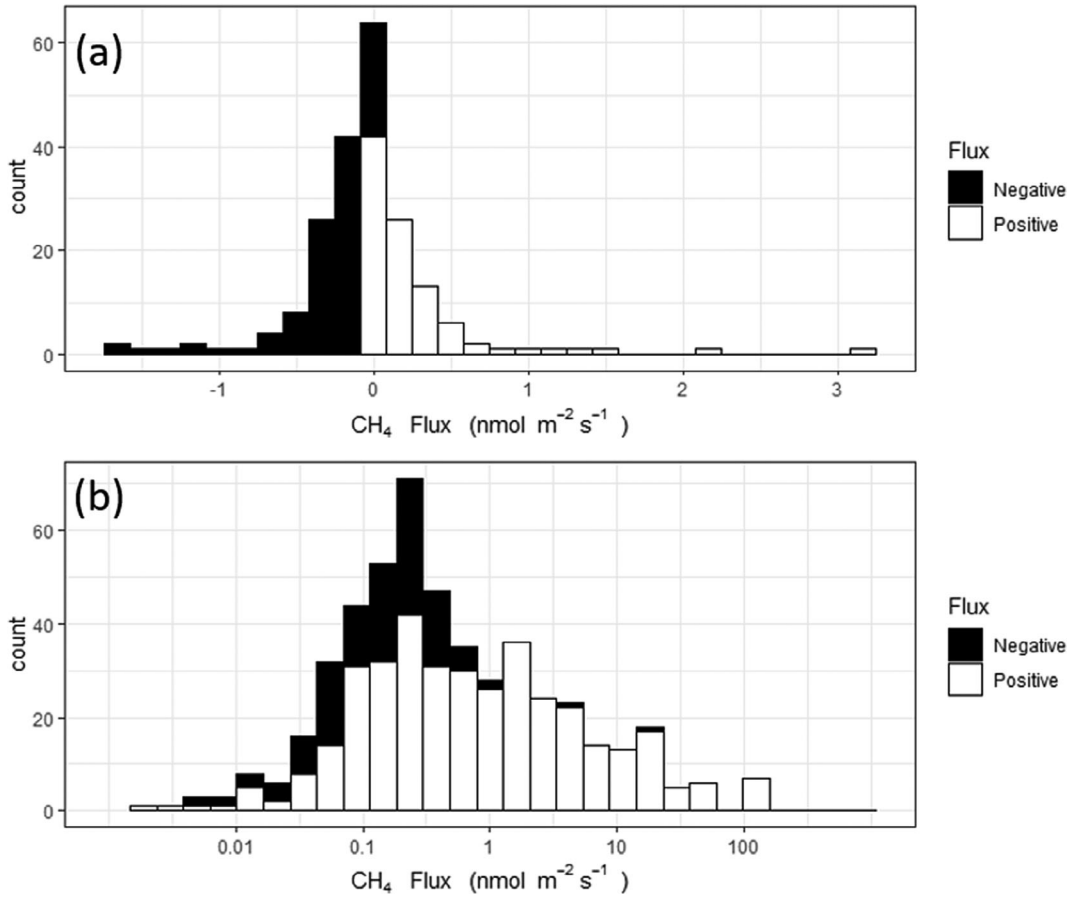

FIGURE 5 Measurements of CH_4 fluxes taken before (a) and after (b) the tillage event reported in Drewer et al. (2017) show that data shift from a normal to log-normal distribution after disturbance

3.4 | Impacts of fertiliser application

Mineral fertiliser application appears to have minimal impacts on CH_4 emissions. Over the 30 days immediately after fertiliser application in all of the studies, only a small number of high flux points were recorded with no obvious temporal pattern (Figure 6). Differences between the fertiliser types were largely experiment/site specific and there was no clear pattern as to whether a particular mineral fertiliser type had any impact on CH_4 (large emissions are generally from grazed fields where animal dung is also present, as is the case for ammonium nitrate (AN) measurements). Application of some organic fertilisers did appear to have a larger impact on CH_4 emissions (Figure 7). In the experiments where dung was applied, a clear disruption of CH_4 emissions was observed for the first week after application. After this, CH_4 emissions dropped back down to background levels rapidly, with a handful of measurements still reporting elevated fluxes. In the experiment carried out by Sánchez-Rodríguez et al. (2018), industrially digested food waste (digestate) was applied in both its acidified and normal state. These peaks were not observed for the acidified digestates for which fluxes were generally negative in the 30 days after application. The application of cow urine appeared to have no effect on emissions, although a small number of high uptake measurements were recorded from these plots (Jones et al., 2005; Maire et al., 2020; Krol et al., 2016).

In the absence of a meaningful model able to gap-fill CH_4 between measurements, any cumulative estimates were open to wide interpretation. Integration using linear interpolation between points is highly sensitive to the number of measurements taken, the presence of outliers and the timing between samples. In this study, therefore, we made the simplistic assumption that the mean flux over 30 days after an application event is proportional to the magnitude of the cumulative flux expected over this period (Table 4). Assuming a GWP of 28 over 100 years (IPCC 2014), we present the estimated fluxes in units of g-CO_2eq m^{-2}. Although the impact of individual (or grouped) events was shown to be significantly different from zero in some cases, fluxes of CH_4 during these periods were low in terms of GWP. The exception to this

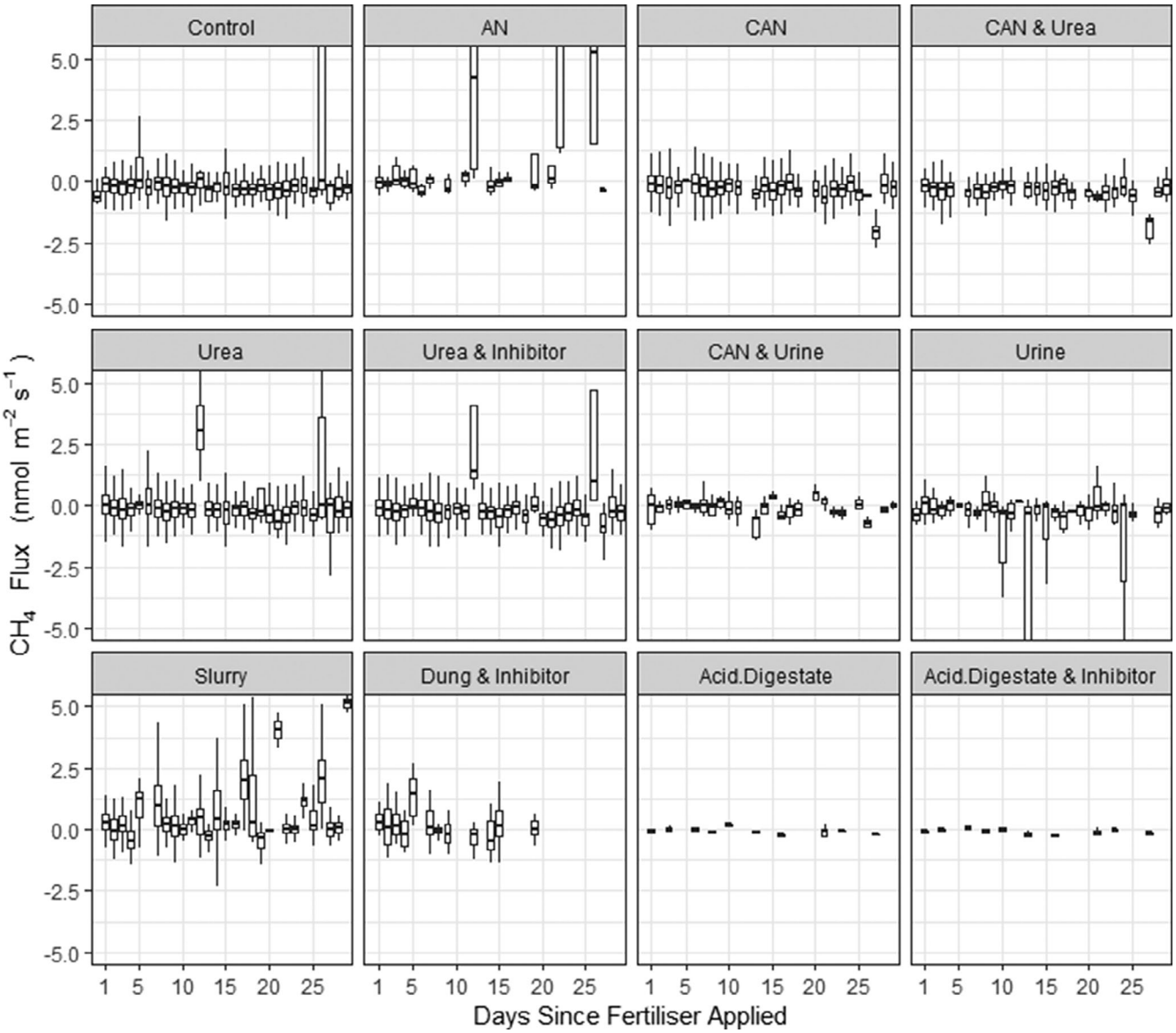

FIGURE 6 Measurements of CH_4 fluxes taken up to 30 days after mineral and organic fertiliser application events (excluding treated digestate and dung) at all sites. Plots show the median values for each data bin, with hinges on the 25% and 75% quantiles. Plots are curtailed visually but not statistically to the −5 to 5 nmol m^{-2} s^{-1} range to exclude outliers. AN, ammonium nitrate; CAN, calcium ammonium nitrate

was dung application, with an estimated emission of up to 21 g CO_2eq m^{-2} in the 30 days after application.

4 | DISCUSSION

This study highlights numerous discussion points regarding the emission and uptake of CH_4 from agricultural soils. The first point to address is that our data show that the vast majority of CH_4 fluxes measured from all agricultural soils are close to zero and fall below the limit of detection of the flux chamber methodologies used to quantify fluxes. For this reason, these CH_4 fluxes are often disregarded by researchers and not reported in studies that use flux chamber methodologies as they are considered negligible or insignificantly different from zero. In many cases where the number of measurements for each particular ''treatment'' is small in short-term experiments, it is unlikely that statistically significant results will be possible. The issue of detection limits and statistical significance of ''noisy'' data does explain in part the contradictory studies from previous decades of CH_4 research where the number of gas samples taken per chamber were low (<4) and GC instruments were less precise than modern equivalents. Care should be taken when assessing small CH_4 flux datasets from soils,

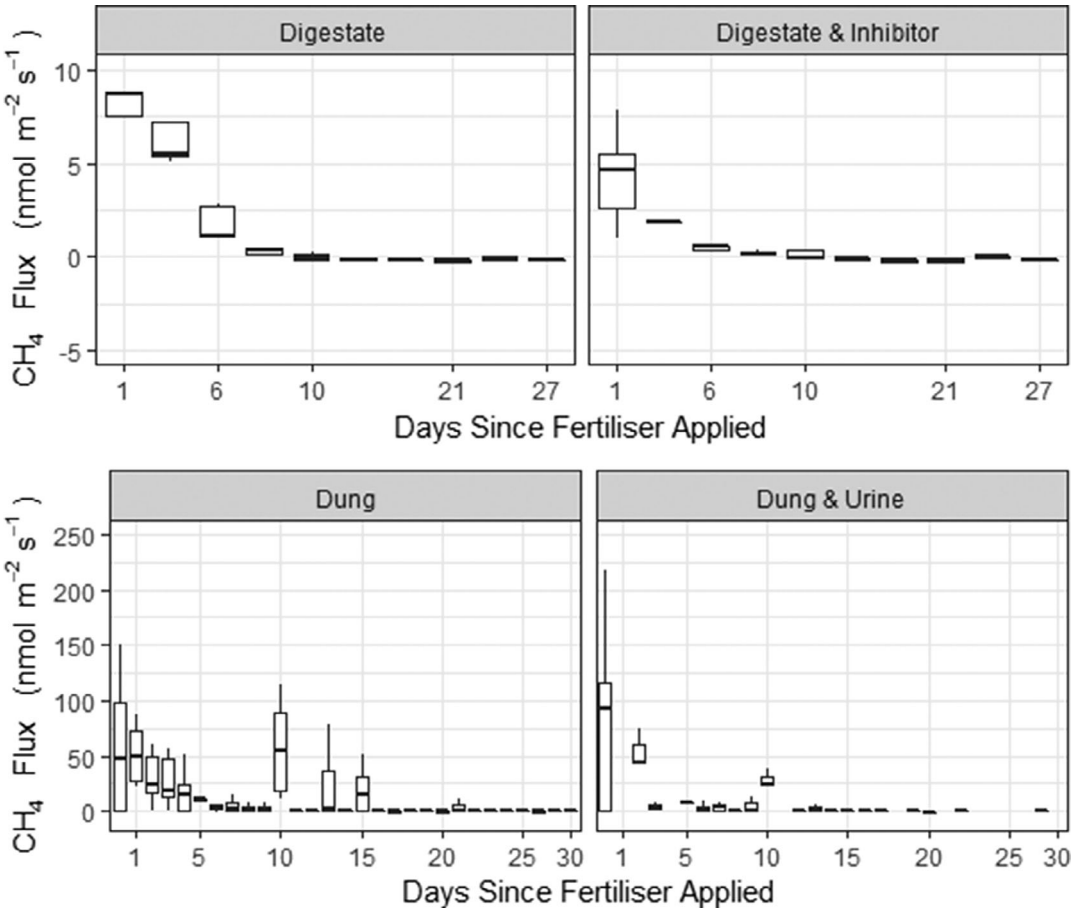

FIGURE 7 Measurements of CH_4 fluxes taken up to 30 days after (top) waste food digestate and (bottom) dung fertiliser application at all sites. Plots show the median values for each data bin, with hinges on the 25% and 75% quantiles. Plots are curtailed visually but not statistically to exclude outliers

especially when detection limits are poor. Ways to improve CH_4 measurements in future studies would be to use modern laser instruments with higher precision than GC instruments, capable of measuring high frequency measurements (1 Hz) from chambers (Hensen et al., 2006).

This study shows that with sufficient data, statistically significant differences can be observed when investigating CH_4 from different agricultural soils during experiments. The data gathered using static chamber methodologies allow for differentiation between fluxes measured from different sites and management practices. Mean fluxes from the 27 different sites reported in this study range from -0.33 ± 0.04 to 3.01 ± 0.76 nmol m^{-2} s^{-1}, showing that although small in magnitude, uptake and emissions of CH_4 are both viable conclusions to reach regarding any particular field site. Our data also suggest that bioenergy crop plantations are significantly more likely to be a small sink of CH_4, with a mean flux of -0.05 ± 0.02 nmol m^{-2} s^{-1} calculated from all plantation site flux measurements. This is perhaps due to no-tillage conditions and limited disturbance by management operations, and agrees well with previous studies that suggested that forest and natural woodland soils are more likely to act as CH_4 sinks than soils under arable and grassland management (Nesbit and Breitenbeck, 1992; Smith et al., 2000). Plantation soils are also likely to be less compacted (i.e., lower bulk densities) than agricultural soils (Nesbit and Breitenbeck, 1992; Smith et al., 2000). Mean fluxes measured from arable and grassland sites (excluding all fertiliser and tillage events) were 0.11 ± 0.06 and 0.19 ± 0.09 nmol m^{-2} s^{-1}, respectively, and were not found to be significantly different (Welch t test, $p = 0.17$).

The strongest predictor of individual flux measurements in the available data was soil volumetric water content (VWC). Although noisy ($R^2 < 0.1$), partly due to the limit of detection of the chamber measurements, the number of measurement points was sufficient to identify a statistically significant relationship between CH_4 flux and VWC. This relationship predicts that in drier soils (<20% VWC), CH_4 uptake is more likely, whereas in very wet (waterlogged, >80% VWC) soils, CH_4 flux will

TABLE 4 The impact of mineral and organic N fertiliser types on CH_4 fluxes. Data are mean and cumulative fluxes of CH_4 estimated for a 30-day period, post fertiliser application, presented as nmol m^{-2} s^{-1} and g CO_2eq m^{-2} (over 30 days) using a global warming potential multiplier of 28. In the absence of a model able to predict CH_4 between measurements, cumulative flux estimates are based on the mean of fluxes recorded from 0 to 30 days after fertiliser application

Fertiliser applied	CH_4 flux 30-day mean (nmol m^{-2} s^{-1})	95% CI	Cum. CH_4 flux 30 days (g CO_2eq m^{-2})	95% CI
Ammonium nitrate	3.64	2.04	3.17	1.77
Calcium ammonium nitrate	−0.22	0.03	−0.19	0.03
Calcium ammonium nitrate/urea	−0.39	0.07	−0.34	0.06
Urea	−0.04	0.06	−0.04	0.05
Urea and inhibitor	−0.22	0.05	−0.19	0.04
Calcium ammonium nitrate and urine	−0.09	0.06	−0.08	0.05
Urine	−0.30	0.31	−0.26	0.27
Slurry	0.88	0.23	0.77	0.20
Dung and inhibitor	0.14	0.12	0.12	0.10
Acidified digestate	−0.06	0.05	−0.05	0.04
Acidified digestate and inhibitor	−0.14	0.11	−0.12	0.10
Digestate	1.63	0.87	1.42	0.76
Digestate and inhibitor	0.75	0.50	0.65	0.44
Dung	24.11	6.21	21.00	5.41
Dung and urine	13.00	5.62	11.32	4.90

increase substantially. These observations agree to some extent with work carried out by van den Pol-van Dasselaar et al., (1998), which identified soil VWC of approximately 20 to 35% as the optimal range of CH_4 uptake in grasslands in the Netherlands. Comparisons between CH_4 flux and VWC reported in this study were somewhat limited by the depth at which measurements were taken, representing only the top 0–10 cm of the soil profiles. Emissions of CH_4 from agricultural soils were also likely to be influenced by CH_4 produced deeper in the soil profiles, dependent upon water table depth, soil aerobicity, soil compaction and available carbon. Where soils are dry and porous on the surface, but very wet and warm with a large carbon content deeper in the profile (anoxic and organic carbon rich), higher emissions of CH_4 might also be expected (Morel et al., 2019).

Fluxes of CH_4 at the site level correlated somewhat significantly with annual rainfall ($R^2 = 0.31$); however, adding further soil properties to the model only slightly improved the prediction. The correlation between site-level CH_4 fluxes and annual rainfall may indicate that CH_4 emissions from a particular area are more dependent on long-term conditions than short-term changes at a site. If this is the case, wetter soils may favour populations of methanogenic bacteria. At the individual measurement and site level, we find poor correlation between fluxes and soil properties that are believed to influence CH_4 emissions (pH, soil temperature, bulk density). Segers (1998) highlighted the difficulty in predicting CH_4 fluxes from soil properties, but other studies have identified relationships between pH and temperature (e.g., MacDonald et al., 1996; Smith et al., 2000). However, the strongest correlations between soil properties and CH_4 fluxes are often found in forest and wetland soils, with emissions from agricultural soils remaining close to zero in most cases (MacDonald et al., 1996; Smith et al., 2000).

The largest impact on CH_4 fluxes of any of the management practices studied here was that of tillage. In Drewer et al. (2017), adjacent intensively managed grazed grasslands were tilled in spring, resulting in a large increase in CH_4 emissions (Figures 4 and 5). This was likely to be a result of the incorporation and subsequent decomposition of grass into the lower soil layers, providing the necessary anaerobic conditions and carbon source for methanogenesis (Badagliacca et al., 2020). Taking simplistic means of the fluxes before tillage and the mean of fluxes after tillage (up to 150 days), fluxes increased from -0.03 ± 0.07 nmol m^{-2} s^{-1} to 4.03 ± 1.29 nmol m^{-2} s^{-1}. Based on these values, it can be estimated that tillage resulted in the net loss of approximately 0.63 g CH_4-C m^{-2} during a 150-day measurement period. Assuming a GWP multiplier of 28, this means that emissions from the field over a 150-day

period resulted in a net loss of 17.7 g CO_2eq m^{-2}. To put that value into context, it would take approximately 55.4 years for this field to absorb the same mass of CH_4 as was released after this tillage event, assuming uptake occurred at pre-tillage rates (−0.03 nmol m^{-2} s^{-1}) for the duration. Large emissions of CH_4 have also been observed from other grassland sites, including Merbold et al. (2014), where CH_4 emissions of 2.65 g CH_4-C m^2 were reported as the annual flux during the year of a tillage event, measured using the eddy covariance method. The change in statistical distribution from normal to log-normal highlights that, potentially, the statistics used to describe this type of activity should take log-normal maths into account. Methods such as a Bayesian approach should be used to determine appropriate means and uncertainties when handling this type of data in future studies (Cowan et al., 2017).

The immediate impact of nitrogen fertilisers (over 30 days) was typically below statistical significance for each of the treatments and dependent on the site at which experiments took place. Individual data points where high fluxes were measured skewed the data. These fluxes were real, but their representation of a particular nitrogen fertiliser application is questionable. In the case of fertiliser application to grazed fields (most of the experiments reported in this study), occasionally animal waste is present in the chambers (although not recorded), which can affect CH_4 flux. Interestingly, all mineral fertiliser applications resulted in uptake, with the exception of AN. As calcium ammonium nitrate (CAN) is expected to slightly increase the pH of soils when applied, an observation of uptake after application is counterintuitive to previous studies, which suggest increasing pH can increase CH_4 emissions from soils (Wang et al., 1993). Our observations mirror those of other studies in which the application of mineral fertilisers is not always well explained by the known soil and microbial chemistry (Hütsch, 1998; Le Mer and Roger, 2001). This is likely to be due to a mixture of the impacts of unmeasured variables and the limit of detection of the chamber methodologies applied. This study cannot determine the long-term impacts of mineral fertilisation on the different field sites, but this factor may affect the extent to which soils can act as a CH_4 sink (Hütsch et al., 1994).

The application of organic fertilisers has a more immediate impact than the application of mineral fertilisers. In the case of dung and digestate additions, an immediate peak and decline in CH_4 emissions were observed, which lasted for approximately 1 to 2 weeks in the data presented. These emissions remained relatively small with a maximum flux of 21 g CO_2eq m^{-2} over a 30-day period after application. This study cannot state a definitive emission factor for CH_4 expected after organic fertiliser application, as this would take many experiments and a functional gap-filling model. The complex properties of livestock dung and urine depend on factors such as animal diet, age and health, which are beyond the scope of this study. What we emphasize in this study is a realistic magnitude of emissions expected after organic manure application. These are documented well in previous studies where similar flux ranges were observed, with emission peaks which typically lasted for 2 to 4 weeks after application (Chadwick et al., 2000; Saggar et al., 2004).

To put the impact of CH_4 emissions and uptake from agricultural soils into context with the other GHGs, we present the data from the Easter Bush grassland (Midlothian, Scotland), where a variety of GHG measurements have occurred (carried out by the authors of this study, see Table 5). CO_2eq emissions are calculated using values provided from the IPCC 2014, Fifth Assessment Report (AR5). In Jones et al. (2017), the CO_2 emissions measured at the site as net ecosystem exchange ranged from −605 to 72 g C m^{-2}, with an annual mean uptake of approximately 217.9 g C m^{-2}. Typically, the Easter Bush field receives approximately 210 kg N per year in the form of AN fertiliser. In Cowan et al. (2020a), emission factors of N_2O measured via the eddy covariance method varied per fertiliser event, ranging from 0.2 to 2.78% of the nitrogen applied as AN, with a mean flux of approximately 0.3 nmol N_2O m^{-2} s^{-1} during periods outside of fertiliser events. Using this estimate as a background flux, an annual estimate of 0.27 g N_2O-N m^{-2} is released from the field when no fertiliser is applied. Using the more refined EF from Cowan et al. (2020), which estimates an EF of 1.34% of the applied N to be emitted as N_2O, the typical annual estimate of N_2O released from the field is approximately 2.75 g N_2O-N m^{-2}. In Cowan et al. (2018), annual soil carbon monoxide (CO) emissions of 0.37 g C m^{-2} at the site were reported. Carbon monoxide is classed as a secondary GHG due to its interactions with OH radicals in the atmosphere, thus extending the lifetime of CH_4 and having a positive GWP as a result. Excluding the study in which the tillage event occurred, the mean CH_4 flux for the Easter Bush field site was 0.1 nmol m^{-2} s^{-1}, which equated to 0.04 g C m^{-2} y^{-1}, and is close to the mean of the CH_4 measured from all sites presented in this study (i.e., 0.19 ± 0.09 nmol m^{-2} s^{-1} for undisturbed grassland).

Using these estimates, the relative contribution of CH_4 to emissions from the soil is negligible when compared to uncertainties that would be expected in CO_2 and N_2O emission estimates. The contributions to the overall CO_2eq emissions from the site account for only 0.4% of the total budget, only fractionally more than those estimated for CO, which contributes an estimated 0.3% of

TABLE 5 The annual greenhouse gas (GHG) budget for the intensively managed grazed grassland for Easter Bush (Midlothian, Scotland) is presented, using global warming potential (GWP) values provided from the IPCC 2014, Fifth Assessment Report (AR5). Annual estimates are made based on the years that measurements took place in Jones et al. (2017) for CO_2, Cowan et al. (2018) for CO and Cowan et al. (2020a) for N_2O

GHG	Annual emission ($g\ C\ m^{-2}$)/($g\ N\ m^{-2}$)	Typical emission ($g\ C\ m^{-2}$)/($g\ N\ m^{-2}$)	AR5 GWP ratio	Annual flux ($g\ CO_2eq\ m^{-2}$)	Annual GWP (%)
CO_2	−605 to 72	−217.9	1	−217.9	88
N_2O	0.64 to 5.42	2.75	265	26.02	11
CH_4	0.01 to 1.07	0.04	28	1.06	0.4
CO	0.35 to 0.38	0.37	∼2	∼0.74	0.3

the total budget. At sites where uptake is expected, a similar comparison could be expected, in which uptake of CH_4 would be statistically insignificant when compared to the uptake and emission of other GHGs. Interestingly, emissions of CO in terms of mass of carbon are higher at the Easter Bush site than CH_4, although CH_4 still contributes more to the total GWP of the site due to the higher GWP of CH_4 molecules. If CH_4 uptake was considered in GHG accounting for agricultural soils, the argument also exists that it would be prudent to include emissions of CO, which is likely to reduce the overall effect of any uptake to even less significance. This budget does not include the effect of tillage, organic fertiliser application or livestock respiration or excreta returns. If these were included, the contribution to the annual GWP of CH_4 is likely to increase (BEIS, 2018; EPA, 2020).

Previous studies have suggested that soils across Europe act as a methane sink, including grassland and upland soils similar to those reported in this study (e.g., Kravchenko 2017; Hörtnagl et al., 2018). It is uncertain from the data presented in this study if there are geographical reasons for soils becoming a CH_4 source across the British Isles (such as high soil moisture or organic matter content), or if comparisons across studies are too sparse or methodologically different to draw a valid comparison. In particular, given the highly skewed nature of the data, the mean is a very uncertain quantity, and the uncertainty bounds on this are rarely quantified correctly, and this affects whether authors conclude that their sites are net sinks or sources. According to estimated global inventories, soils are expected to act as a sink for an estimated 30 to 60 Tg of CH_4 annually (Cicerone and Oremland, 1988; Ridgwell et al., 1999). On average, soil CH_4 uptake at a global level is estimated to be in the order of −0.40 to −0.79 $nmol\ m^{-2}\ s^{-1}$, with an annual uptake of approximately 4.2 to 8.4 $g\ CO_2eq\ m^{-2}$. The UK and Ireland account for 244,654 and 70,273 km^2 of land cover, respectively. If generic global soil uptake values were applied to UK and Irish agricultural soils, an annual CH_4 uptake of 0.94 to 1.88 Mt of CO_2eq would be expected to occur across both countries. In the UK, 56,506 km^2 of land is classed as arable and 96,949 km^2 is classed as grasslands (improved, neutral, calcareous and acid) (Land Cover statistics derived from LCM2015; Rowland et al., 2017). In Ireland, 6,676 km^2 of land is classed as arable and 42,867 km^2 of land is classed as grasslands (CSO, 2018). Using the CH_4 fluxes estimated for these land classes reported in this study, these areas become a source, and an emission of 0.16 and 0.09 Mt of CO_2eq is expected to be emitted from the UK and Ireland, respectively. Compared to the 51.5 and 14.0 Mt CO_2eq of CH_4 emitted across the UK and Ireland from other sources, such as energy production and distribution, landfills, ruminants, manure management and wetlands (BEIS, 2018; EPA, 2020), emissions of CH_4 from UK and Irish agricultural soils are insignificant, representing only 0.3% and 0.7%, respectively, of the total budget.

5 | CONCLUSION

Our study concludes that release and uptake of CH_4 by agricultural soils are both possible and can be statistically significant at any given field site. The data presented also suggest that fluxes of CH_4 from plantation fields are more likely to be negative than those of arable or grassland sites. However, overall, both arable and grassland fields were on average small emitters of CH_4 due to the presence of hotspots (small areas [valid individual measurements] of high fluxes) that skewed data from uptake into a positive emission. Findings suggest that uptake of CH_4 should not be considered for GHG inventories of specific farms, or at a national level. Alternatively, we predict CH_4 emissions from intensively managed agricultural soils to be approximately 0.16 Mt of CO_2eq for UK land and 0.09 Mt of CO_2eq for Irish land. Furthermore, we predict agricultural management practices, such as the application of organic fertilisers and the tillage of fields, are likely to counteract most CH_4 consumption by methanotrophic bacteria in agricultural soils. In comparison to other GHGs, uptake or

release of CH_4 from soils alone can be expected to be in the same order of magnitude in terms of GWP as CO emissions (secondary GHG), further reducing the significance of GHG mitigation by CH_4 uptake in soils if the CO factor is also considered. Our overall recommendation is that where no long-term measurements from a particular (or representative) field site are available, soil CH_4 emissions can be assumed to be negligible in any localized GHG budget within the British Isles. This is due to a lack of evidence to classify soils from a particular area as being either a significant CH_4 source or sink without a dedicated measurement campaign. As such, uptake of CH_4 in agricultural soils should not be used as a sink to offset emissions from livestock in the UK and Ireland.

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- AHDB Dairy
- UK-China Virtual Joint Centre for Improved Nitrogen Agronomy (CINAg, BB/N013468/1)

CONFLICT OF INTEREST

The authors of this study certify that they have no affiliations with or involvement in any organization or entity with any financial or non-financial interest in the subject matter or materials discussed in this manuscript.

AUTHOR CONTRIBUTION

Nicholas Cowan collected the flux data from multiple research groups, carried out data analysis and wrote the manuscript as the primary author. All other authors carried out a significant amount of field measurements and data processing of the original measurement data at the various field sites described in this study. All authors have contributed to the manuscript and have assisted in providing the data and details necessary to complete the study.

DATA AVAILABILITY STATEMENT

The data that support the findings of this study are available on request directly from the relevant authors of the described studies (conditions apply). Restrictions apply to the availability of these data, which were used under license for this study.

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