Greenhouse gas fluxes and the carbon balance of temperate agricultural systems: Impact of environmental and management factors

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Declaration and author contributions

The candidate confirms that the work submitted is their own, except where work which has formed part of jointly authored publications has been included. The candidate confirms that appropriate credit has been given within the thesis where reference has been made to the work of the authors, with the contribution of the candidate and other authors to this work explicitly indicated at the end of each chapter in a CRediT author statement.

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Covid-19 impact statement

The PhD commenced in October 2020 and thus was affected by the Covid-19 pandemic until around December 2021. As a result of the pandemic, measurements of net ecosystem exchange using eddy covariance flux towers at the University of Leeds Research Farm commenced later than intended due to social distancing and isolation measures. The datasets of the cropland and permanent pasture reported from this site are therefore shorter than was anticipated when the project commenced.

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Abstract

The agricultural sector contributes significantly to global greenhouse gas (GHG) emissions and soil carbon (C) loss. There is an urgent need to move towards more sustainable agricultural production systems that improve soil C sequestration, reduce GHG emissions, and provide sufficient food and environmental benefits for a growing population. Little is known, however, about how the influence of specific agricultural management practices, such as crop and fertiliser type, on soil C and GHG fluxes varies depending on the local climate conditions and soil type. This thesis aimed to improve the understanding by comparing C fluxes from agricultural soils both globally and in the UK, and by comparing soil GHG fluxes from winter wheat treated with different fertilisers. A global meta-analysis found croplands and managed grasslands to be losing C (a mean loss of 110 g C m^{-2} and 29.9 g C m^{-2} respectively), regardless of the implementation of best management practices. Monitoring of sites in the UK showed that, over one year, a cropland was C neutral (-26 g C m^{-2}) whereas a neighbouring cut and grazed pasture was losing C (311 g C m^{-2}), and that, when grown in rotation, maize lost C over its growing season (136 g C m^{-2}), whereas winter wheat and vining pea behaved as C sinks over their growing seasons (-129 g C m^{-2} and -154 g C m^{-2} respectively). Furthermore, C losses during the maize growing season were higher when maize was grown on peat (290 g C m^{-2}) compared to mineral soil (136 g C m^{-2}). The research highlights the importance of considering C fluxes during fallow periods in addition to those during growing seasons, as total net ecosystem productivity (NEP) over three crop growing seasons was negative (-166 g C m^{-2}), indicating C uptake, whereas total NEP over three fallow periods was positive (375 g C m^{-2}), indicating C loss. Discounting fallow C fluxes can therefore considerably overestimate the C sink activity of a cropland. Additionally, fertiliser type was found to influence GHG fluxes from soil under winter wheat over *c.* 2.5-months post-fertiliser application; nitrous oxide (N_2O) fluxes increased when plasma-treated pig slurry was applied (1.14 g N m^{-2}) compared to untreated pig slurry (0.32 g N m^{-2}) and inorganic fertiliser (0.13 g N m^{-2}), and methane (CH_4) fluxes were significantly greater when untreated pig slurry was applied (3.2 g C m^{-2}) compared to plasma-treated pig slurry (-1.4 g C m^{-2}) and inorganic fertiliser (-1.4 g C m^{-2}). The results of the thesis highlight the importance of C inputs for reducing agricultural C losses, the trade-offs of various management practices, and the need for long-term NEP measurements from UK sites using best practices to reduce GHG emissions and increase soil C storage.

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List of abbreviations

AD	Anaerobic digestion
AES	Agri-environment scheme(s)
BACI	Before-after control-impact
BECCS	Bioenergy with carbon capture and storage
C	Carbon
CAP	Common Agricultural Policy
CEP	Circular Economy Package
CF	Crop field at UoL Research Farm
C_H	Carbon export
CH_4	Methane
C_I	Carbon import
CO_2	Carbon dioxide
CUE	Carbon use efficiency
CUE_h	Harvest carbon use efficiency
DCD	Dicyandiamide
DM	Dry matter
DMPP	3,4-dimethylpyrazole phosphate
EBC	Energy balance closure
EC	Eddy covariance
EF	Emission factor
EIP	Environmental Improvement Plan
ELM	Environmental Land Management
F1	First fallow period in CF at UoL Research Farm
F2	Second fallow period in CF at UoL Research Farm
F3	Third fallow period in CF at UoL Research Farm
H	Sensible heat flux
G	Soil heat flux
GHG	Greenhouse gas
GHGI	Greenhouse gas intensity
GPP	Gross primary productivity

IF	Fertiliser treatment: three applications of inorganic fertiliser
LE	Latent heat flux
LWin	Longwave incoming radiation
LWout	Longwave outgoing radiation
MDF	Minimum detectable flux
MLR	Multiple linear regression
MS	Mineral site
N	Nitrogen
N_2O	Nitrous oxide
NBP	Net biome productivity
NECB	Net ecosystem carbon balance
NEE	Net ecosystem exchange
NEE_{adj}	Adjusted net ecosystem exchange
NEP	Net ecosystem productivity
NEPH	Net ecosystem productivity with harvest
NH_3	Ammonia
NH_4^+	Ammonium
NH_4NO_3	Ammonium nitrate
NO_2^-	Nitrite
NO_3^-	Nitrate
NUE	Nitrogen use efficiency
NUE_{grain}	Nitrogen use efficiency of grain
NUE_{total}	Nitrogen use efficiency of whole-crop
OM	Organic matter
PAR	Photosynthetically active radiation
PP	Permanent pasture at UoL Research Farm
PS	Peat site
RH	Relative humidity
Rnet	Net radiation
SFI	Sustainable Farming Incentive

SFP	Single Farm Payment
SFS	Sustainable Farm Scheme
SD	Standard deviation
SOC	Soil organic carbon
SOM	Soil organic matter
SWin	Shortwave incoming radiation
SWout	Shortwave outgoing radiation
Ta	Air temperature
TER	Total ecosystem respiration
TPS	Fertiliser treatment: two applications of plasma-treated pig slurry and two applications of inorganic fertiliser
Tsoil	Soil temperature
Tsonic	Sonic temperature
u	Horizontal wind velocity in east-west direction
u*	Friction velocity
UoL	University of Leeds
UPS	Fertiliser treatment: two applications of untreated pig slurry and two applications of inorganic fertiliser
v	Horizontal wind velocity in north-south direction
w	Vertical wind velocity
WFPS	Water-filled pore space

Chapter 1 Introduction

1.1 Global agricultural land use

Approximately 38 % of the Earth's terrestrial land area is used for agriculture (FAO, 2020) and has been rapidly expanding since the 1950s as a consequence of the continuously growing population (Potapov et al., 2022). Between 1950 and 2021, the global agricultural land area increased from 3.84 billion ha to 4.79 billion ha (HYDE, 2017; FAO, 2023) due to the conversion of non-agricultural land for use in agricultural production. Around 30 % of the global agricultural land area is cropland and around 70 % is grassland (Moinet et al., 2017; FAO, 2020). An increased demand for food, and more recently biomass for bioenergy production (Hanssen et al., 2020), has placed a strain on agricultural land, with the area of agricultural land per person declining over time from 1.66 ha in 1600 to 0.66 ha in 2016 (HYDE, 2017). As the global population is expected to reach 9.7 billion by 2050 (United Nations, 2022), these pressures on agricultural land will only continue to rise. Widespread deforestation (Angelsen, 2010), peatland drainage (Saurich et al., 2019), and more intensive agricultural management practices (known as industrial agriculture) (Horrigan et al., 2002) as a result of agricultural intensification have detrimental effects on soil health and the wider environment. Agriculture is the primary driver of global soil degradation and biodiversity loss (Lal, 2015a), contributes to air and water pollution (Giannadaki et al., 2018; Tudi et al., 2021), and is responsible for up to 8.5 % of global greenhouse gas (GHG) emissions (IPCC, 2019). The sector is also highly vulnerable to climate impacts. To achieve net zero GHG emissions by 2050, and to meet global food needs and environmental commitments, a systematic shift in agricultural management practices is essential.

Soils represent an important carbon (C) store; globally soils store over 2500 Pg C, of which 1500 Pg is organic C (Zomer et al., 2017). Carbon in soils is important for healthy plant growth, as it supports good soil structure, fertility and water infiltration (Todd and Schulte, 2012). Furthermore, soil C sinks are critical for removing carbon dioxide (CO_2) from the atmosphere, so building the soil C pool will be crucial for combatting climate change (Lal, 2004a). Over the past 200 years, agricultural expansion has resulted in an estimated loss of 133 Pg C from soil (Sanderman et al., 2017) as a result of intensive management practices such as deep tillage (Tanveer et al., 2018). The disruption of soil aggregates exposes soil organic C (SOC) which is then oxidised and emitted as CO_2, contributing to climate change (Lal, 2004a; Jiang et al., 2023). Soil structural degradation, in addition to periods of bare soil (fallow), also enhances soil erosion and subsequent soil C loss (Chowaniak et al., 2020). High rates of biomass removal as harvested or grazed biomass in intensively grazed grasslands reduce the amount of C from organic material that is returned to the soil (Soussana et al., 2007; Tang et al., 2019).

Around 3 % of the global land area is peatland (IUCN, 2024). Peat is highly organic and stores over 600 Gt C, which is a considerable portion of the world's soil C (IUCN, 2021; Zhou et al., 2021). Peat accumulates in waterlogged conditions, and so peatlands must be drained for use in agriculture to create suitable conditions for the growth of a wide variety of crops (Maljanen et al., 2010). Globally, around 50 million ha of peat have been drained thus far for use as cropland, grazing land, forestry or infrastructure (Convention on Wetlands, 2021). Drainage causes peat to dry, subside and rapidly decompose which releases stored C as CO_2 (Lindsay et al., 2014). Furthermore, peatland drainage requires energy-intensive pumps (Evans et al., 2021), meaning that crops grown on drained peat have high GHG production intensities (Carlson et al., 2016).

Global croplands are primarily managed with monocropping – where only one type of crop is grown over multiple growing seasons (Power and Follet, 1987) – or in rotation – where different crops are grown in a sequence (Blanco-Canqui et al., 2013). Monoculture cropping is particularly common in the USA, where the production of soybean, maize and cotton is high (Altieri and Nicholls, 2020). As crops each have specific nutrient requirements, monoculture cropping causes the soil to become quickly depleted of certain nutrients (Salaheen and Biswas, 2019). It is common for landowners to address this nutrient deficit with fertiliser application, particularly in high-intensity systems, although this can cause further problems as a result of elevated GHG emissions (Section 1.4). Growing crops in a rotation aims to replenish the soil with the nutrients used by the previous crop (Ball et al., 2005); the inclusion of a legume crop, for example, will fix nitrogen (N) in the soil (Min et al., 2016) and therefore reduce the requirement for additional inputs of N fertiliser.

Monoculture systems are also more vulnerable to pests and diseases as there is a permanent host crop (He et al., 2019), and so the inclusion of break crops in a cereal rotation, such as oilseed rape, peas or potatoes (Finch et al., 2002), prevents the development of pests, diseases and weeds by disturbing the continuity of the host crop (Ball et al., 2005).

There is considerable C sequestration potential associated with the conversion of croplands to grasslands (Blair, 2018; Guillaume et al., 2022; De Rosa et al., 2023; Wall et al., 2023a). Grasslands usually have higher C inputs (C_I) due to greater root biomass (McGonigle and Turner, 2017) and belowground C translocation (Kuzyakov and Domanski, 2000) as a result of more continuous vegetation cover, and via excreta from grazing livestock (Chang et al., 2015). Grasslands have also been shown to have better soil quality than croplands as there is no disturbance from tillage events (Jones and Donnelly, 2004) and often less synthetic fertiliser addition which can have negative effects on soil microorganisms (Tripathi et al., 2020). Agriculturally managed grasslands are used to produce food for livestock – growing silage to export for feed or by grazing livestock directly on the field – or to produce bioenergy from biomass. Grazed grasslands are commonly managed by either continuous grazing – where livestock are always present in the field – or by rotational grazing – where livestock are frequently moved between fields or paddocks (Liu et al., 2020a). Continuous grazing is often associated with low vegetation productivity as the biomass is constantly removed from the field by livestock at a relatively steady rate (James, 2011) and so has little time in which to re-grow. Continuous grazing also increases the risk of overgrazing, where the vegetation has no time in which to replenish itself, which leads to very poor soil quality and soil C loss, and is common in semi-arid areas (Cipriotti et al., 2019). Rotational grazing allows the vegetation to re-grow before the next grazing event, and has benefits for soil health as vegetation is allowed more time in which to establish, so plant roots can grow bigger and the requirement for fertiliser is reduced (Teutscherova et al., 2021; Albanito et al., 2022). Managing a grassland by alternating between periods of grazing and no grazing to allow grass to grow prior to a harvest event combines these benefits and is common in mixed farming systems.

1.2 UK agricultural land use

Since 2000, the total utilised agricultural area in the UK has remained steady at between 17 and 18 million ha (DEFRA, 2022a). Currently, 71 % of the UK's land area is used for agriculture, with 30 % of this used for crop production and 60 % as managed grassland (DEFRA, 2022a). Most of the UK's agricultural land is on mineral soil, however around 7 % of the country's peat area – equivalent to 44,500 ha – has been drained for use in agricultural production (Evans et al., 2021). Wheat is the most widely grown arable crop in the UK, occupying around 40 % of the country's cropping area (Harkness et al., 2020). Barley and oilseed rape are also commonly grown (DEFRA, 2020a), however other cereals (i.e., maize and oat) and legumes (i.e., peas) are becoming more popular (DEFRA, 2020b; DEFRA, 2022b). The UK is not food self-sufficient, however; 46 % of the food consumed is imported – a large contrast to 22 % in 1984 (AHDB, 2022). Cropland in the UK is not only used for producing food crops; the amount of land dedicated to maize production for bioenergy in the UK has grown particularly fast – increasing from 34,000 ha in 2015 to 75,000 ha in 2020 (DEFRA, 2021a). Anaerobic digestion (AD) or the combustion of biomass to produce energy has received considerable attention as a renewable resource in recent years (Hanssen et al., 2020; Calvin et al., 2021), and, as maize is high-yielding and has a high biogas output (Herrmann, 2013), it has become a popular bioenergy crop. The use of productive agricultural land to grow crops for bioenergy has been met with criticism, however, as it reduces the amount of land available to produce food crops, and so may threaten food security (Kline et al., 2016).

In the UK, agricultural grassland is critical for supporting livestock production, and thus outputs of animal-derived products (Qi et al., 2018). Across the UK, agricultural grassland is commonly managed as either permanent pasture – grassland that has not been re-sown within the last five years and is used for growing vegetation for fodder – as part of an arable rotation – where a field is alternated between crops and grass – or as rough grazing – where livestock, usually cattle or sheep, are present (Kilpatrick et al., 2008; DEFRA, 2022c). UK grasslands can also be classified as temporary – if less than five years old (Kilpatrick et al., 2008) – or improved – grassland that has undergone reseeding and receives regular inputs of N fertiliser (DEFRA, no date a).

1.3 UK agricultural policy

Between 1973 and 2020, UK agricultural policy was integrated with that of the European Union’s, via the Common Agricultural Policy (CAP) (Seidel, 2019). The initial aim of the CAP was to improve agricultural productivity and ensure a consistent supply of affordable food for society, which was achieved through guaranteed prices and assured markets for farmers (European Commission, no date a). This resulted in a large increase in the intensity of agricultural operations, such as land expansion and peatland drainage, as landowners aimed for maximum output to feed a growing population (Emmerson et al., 2016). Furthermore, the CAP was instrumental in the expansion of the biodiesel sector in Europe, as payments were also made to farmers for the growth of non-food crops that could be used to produce bioenergy (Coelho and Goldemberg, 2004). By the 1980s, food commodities in the European Union were being vastly overproduced, resulting in a surplus of some products such as butter and wine (Reinhorn, 2007), which negatively affected the environment. From 1992 onwards, various reforms of the CAP occurred, including the introduction of agri-environment schemes (AES) in 1993 to reduce the negative impacts of agriculture on the environment, and the Single Farm Payment (SFP) scheme, usually referred to as the ‘Basic Farm Payment’, which was implemented in 2005 (European Commission, no date a). The SFP scheme de-coupled subsidies from production, meaning that farmers were allocated one standard payment regardless of the amount produced (Sanders et al., 2011). Farmers were instead encouraged to produce food in response to consumer demand, and so were able to place a greater priority on improving animal health and welfare standards and caring for the environment (Sutherland, 2010). Another significant reform to the CAP occurred in 2013, in response to climate change and the challenges of global markets, with even greater importance placed on reducing the negative impacts of agriculture on the environment (European Union, 2019). This involved farm payments being ‘greened’, meaning that a proportion of direct payments would only be guaranteed if farmers implemented practices that had an environmental benefit, for example using organic production methods, diversifying cultivation by growing multiple crops, and maintaining permanent grassland (Cortignani et al., 2017).

In 2020, following the withdrawal of the UK from the European Union, the Agriculture Act was passed, providing a legal framework for the UK to establish its own agricultural policy (Coe et al., 2020). Agricultural policy is a devolved matter, however, so the four nations of the UK have each developed their own policies, with national legislation introduced where required. In England, these policies are implemented through Environmental Land Management (ELM) schemes, which are slowly replacing the CAP, although are not a finished product as the schemes continue to be developed into 2024. The ELM schemes will pay farmers for producing food using sustainable methods, as well as for the provision of environmental goods and services (DEFRA, 2023a). There are three ELM schemes: the Sustainable Farming Incentive (SFI) which pays for environmentally friendly farming; the Countryside Stewardship scheme which pays for the implementation of actions in specific habitats; and the Landscape Recovery scheme which provides farmers with financial assistance for larger projects that aim to benefit the environment (DEFRA, 2023a). The SFI is focused on supporting farmers to manage their land for sufficient food provision whilst minimising the environmental impacts of doing so, providing subsidies for a range of actions such as the addition of organic matter (OM) to soil and reducing the amount of time that soil is bare for (DEFRA, 2023a). Actions that farmers take through ELM schemes will also contribute to national environmental and climate goals by improving the state of the environment and reducing GHG emissions, which are fundamental to the UK Government's Environmental Improvement Plan (EIP) (UK Government, 2023a). The EIP builds on the 25-Year Environment Plan (UK Government, 2023b) and sets out a framework for how the environment can be improved through the collaboration of landowners, businesses and communities across the environmental, agricultural and marine sectors (UK Government, 2023a).

The goals of multiple policies are aligned with those of the EIP, many of which are specific to agriculture or place a large focus on the actions of the agricultural sector. The Net Zero Government Initiative, introduced in 2023, aims for all sectors of the economy to achieve net zero GHG emissions by 2050 (DESNZ, 2023a). In contribution to the UK's Net Zero Strategy, the NFU has set a target for the agricultural sectors in England and Wales to achieve net zero GHG emissions by 2040 (NFU, 2019). The NFU's approach is centered on working with farmers, scientists, industry and government to achieve this by focusing efforts on three key pillars: boosting agricultural productivity and reducing emissions, increasing C storage in farmland, and using bioenergy with C capture and storage (BECCS) (NFU, 2019). The Biomass Strategy is a key contributor to the Net Zero Government Initiative and identifies actions for how biomass production can become more sustainable and how biomass can be most efficiently utilized for energy generation (DESNZ, 2023b). The Circular Economy Package (CEP), introduced in 2020, forms part of the 25-Year Environment Plan, and now the EIP, and aims to maximize resource use and minimize waste where possible (UK Government, 2020). Finally, the England Peat Action Plan is focused on peat restoration in England, aiming to restore 35,000 ha of peatland by 2025 to benefit wildlife and further contribute to the UK's net zero goal (UK Government, 2021a).

In 2023, the Agriculture and Rural Communities Bill was introduced in Scotland to give Scottish Government the power to develop a new framework to replace the CAP (Scottish Government, 2023a). Due to be implemented in 2025, the framework will help Scotland achieve its 'Vision for Agriculture' which is focused on sustainable food production and regenerative practices (Scottish Government, 2022). The framework will provide payments to farmers across four tiers; Tier 1 will support food producers based on the conditions that climate, environmental and business standards are met; Tier 2 will build on Tier 1 by providing additional support to farms based on the implementation of practices that reduce GHG emissions and enhance nature; and Tiers 3 and 4 are based on more targeted measures such as skills development, knowledge sharing, tree planting and peatland restoration (Shohet, 2022). Around 70 % of payments will be for actions undertaken as part of Tiers 1 and 2, which will be direct and thus available for all farmers who meet the required standards, with the remaining 30 % for Tiers 3 and 4 which are competitive (Shohet, 2022; Corsair, 2024).

Similar to the Scottish Agriculture and Rural Communities Bill, the Agriculture (Wales) Act 2023 is Wales's post-Brexit agricultural support scheme (Welsh Government, 2023a). The Act sets out Sustainable Land Management Objectives which focus on the need to sustainably produce food, mitigate against climate change, and enhance ecosystems and the countryside (Welsh Government, 2023b), and will primarily be implemented through the Sustainable

Farming Scheme (SFS). The SFS will be implemented in 2025, and will pay farmers for practices that reduce GHG emissions, mitigate flood and drought risks, maximise C sequestration, improve water quality and maximise resource efficiency, alongside many other outcomes (Welsh Government, 2023b).

In Northern Ireland, the Future Agricultural Policy Framework Portfolio for Northern Ireland (DAERA, 2021) identifies four priorities for agriculture, including increased productivity and environmental sustainability, to be achieved by the Agriculture Policy Programme (Thomson and Moxey, 2023) which is currently under development. Efforts made towards these specific targets across England, Scotland, Wales and Northern Ireland will also contribute to achieving wider targets; these include the UK Government's Net Zero Initiative, and the Paris Climate Agreement, which aims to limit the global temperature increase to 1.5 ˚C above pre-industrial levels (United Nations, 2015).

1.4 Greenhouse gas emissions from agricultural soils

The agricultural sector is responsible for up to 8.5 % of global GHG emissions (IPCC, 2019) and around 11 % of the UK's GHG emissions (DBEIS, 2023). Around one third of global GHG emissions are produced by the agri-food sector (Crippa et al., 2021), which encompasses emissions originating on-farm, but also from pre- and post-production, food manufacturing and household consumption (FAO, 2022). The main sources of GHGs from the agricultural sector include direct emissions from soil, synthetic fertiliser production, livestock waste management, indirect emissions of nitrous oxide (N_2O) (i.e., leaching and volatilization), vehicle emissions, and enteric fermentation from livestock (DEFRA, 2022c). The contribution of the agricultural sector to global GHG emissions has increased over time, from 4.98 billion tonnes CO_2-equivalent in 1990 to 5.87 billion tonnes CO_2-equivalent in 2020 (Ritchie, 2020) as a result of agricultural expansion and the intensification of management practices. Overall GHG emissions from the UK, however, decreased by 23 % between 1990 (65 million t CO_2-equivalent) and 2020 (50 million t CO_2-equivalent) (Ritchie, 2020). Recent estimates state that in 2021 the agricultural sector was responsible for 1.9 % of the UK's CO_2 emissions, 49 % of its methane (CH_4) emissions and 71 % of its N_2O

emissions (DEFRA, 2024a). Methane and N_2O are particularly powerful GHGs with global warming potentials 28 and 237 times that of CO_2 respectively over a 100-year period (IPCC, 2021), meaning that they remain in the atmosphere for longer and cause more warming than CO_2 (Munoz et al., 2010). The main agricultural sources of these three major GHGs are discussed in turn below.

1.4.1 Carbon dioxide

Carbon dioxide emissions from agricultural soils are dominated by respiration (Wohlfahrt et al., 2008; Eugster and Merbold, 2015) (Figure 1.1), which is affected by both the environmental conditions at a site (i.e., climate and soil) and the management practices used. Soil organic matter (SOM) mineralisation is the process by which soil C and nutrients are transformed to CO_2 and plant-available forms of nutrients, including nitrate (NO_3^-) and ammonium (NH_4^+) (Gan et al., 2020). Soil organic matter mineralisation is regulated by temperature and soil texture; many studies have observed that soils in warmer and wetter climates, and soils that are fine-textured, have favourable conditions for microbial activity and SOM mineralisation, and thus CO_2 emissions (Dilustro et al., 2005; Jager et al., 2011; Shakoor et al., 2021). Intensive agricultural practices like tillage disturb soil structure by breaking up soil aggregates, which exposes the C in SOM to mineralisation (Reicosky, 1997; Eze et al., 2018; Farhate et al., 2018). The application of organic fertiliser, typically livestock manure, slurry and compost (Singh et al., 2020), is a common agricultural practice to provide a supply of OM and nutrients to improve soil structure and fertility, microbial activity and crop growth (Assefa and Tadeese, 2019). A high proportion of the C supplied in organic fertiliser is labile, meaning it is readily decomposed by soil microorganisms (Haynes, 2005; Zhang et al., 2020). This decomposition further stimulates soil microbial activity, and releases CO_2 via microbial respiration or the priming effect – when the decomposition of older soil C is accelerated by the input of new soil C (Liu et al., 2020b; Machiara et al., 2020; Doyeni et al., 2021). Retaining crop residues on the soil surface is promoted to improve soil health, as it contributes to improved soil structure and a higher SOM content (Liang and Wang, 2020). Crop residue retention may cause an increase in CO_2 emissions, however, as the decomposition of residues on the soil surface provides material for soil microbes to use as a substrate, which releases CO_2 both directly to the atmosphere and via microbial respiration (Gebremedhin et al., 2012; Mangalassery et al., 2015; Wegner et al., 2018; Veeck et al., 2022). Alternatively, where crop residues are ploughed into the soil, older SOC is likely to be oxidised and released as CO_2 (Ussiri and Lal, 2009; Ruan and Robertson, 2013; Wegner et al., 2018). Implementing any of the above agricultural practices on drained peatland is likely to result in higher CO_2 emissions compared to when implemented on mineral soil, as peat has a considerably higher OM content, and thus greater potential for C loss (Lohila et al., 2003).

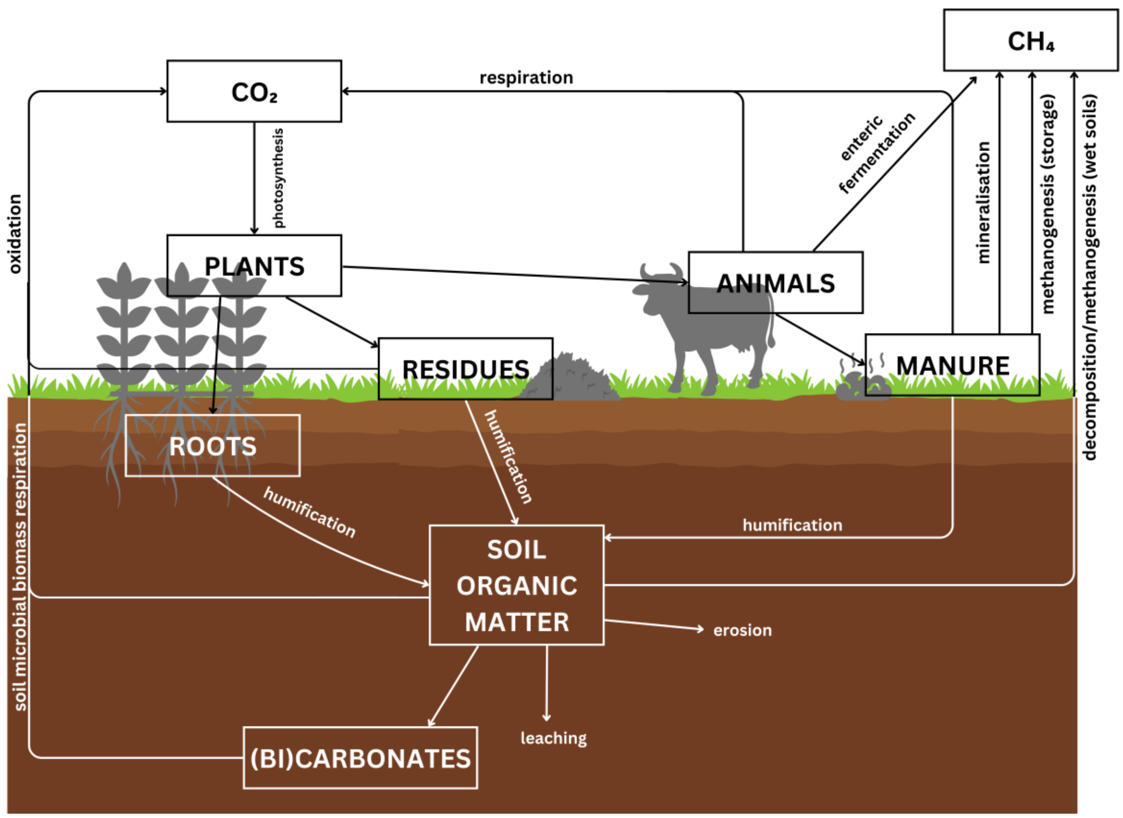

FIGURE 1.1 SOIL CARBON CYCLE IN AN AGROECOSYSTEM (ADAPTED FROM BRADY AND WEIL, 2002).

1.4.2 Methane

Global agricultural CH_4 emissions are primarily attributed to rice cultivation in tropical climates and enteric fermentation from livestock (Chadwick et al., 2000), however a considerable proportion is associated with the storage and use of manure or slurry as an organic fertiliser (Le Mer and Roger, 2001) (Figure 1.1). Applying livestock slurries and manures is promoted to add a supply of OM and C, improve soil quality and contribute to an on-farm circular economy by reducing waste (Case et al., 2017; Chew et al., 2019). Methanogenesis is the process by which methanogens produce CH_4 (Le Mer and Roger, 2001); the key requirements for methanogenesis are anaerobic conditions, a source of C, temperatures between 30 and 40 ˚C and the presence of methanogens (Le Mer and Roger, 2001). These conditions are prevalent in on-farm storage tanks where livestock waste is kept before being applied to soil as fertiliser, with the livestock waste providing the required C source (Mobilian and Craft, 2022). The CH_4 produced in the storage tanks can be either directly emitted to the atmosphere (Baral et al., 2018) or dissolved into the waste and volatilised and emitted to the atmosphere upon application to soil (Rochette and Cote, 2000; Severin et al., 2015). Furthermore, CH_4 emissions can occur after manure or slurry application, as the mineralisation of OM in the organic waste creates anaerobic microsites in the soil where CH_4 is produced and directly emitted to the atmosphere (Pampillon-Gonzalez et al., 2017). In peat soils, or soils where the water table is high such as paddy soils, CH_4 can also be emitted following a range of processes including diffusion, where CH_4 produced in anaerobic layers is released to the atmosphere, and oxidation by methanotrophic bacteria (Busman et al., 2023; Ouyang et al., 2023).

1.4.3 Nitrous oxide

Nitrous oxide emissions from agricultural soils are a product of nitrification and/or denitrification (Khalil et al., 2004; Chantigny et al., 2013; Zhang et al., 2021) (Figure 1.2). Nitrification occurs in aerobic conditions; NH_4^+ is oxidised to nitrite (NO_2^-) and NO_3^-, and N_2O is emitted as a by-product (Ergas and Aponte-Morales, 2014). Denitrification occurs in anaerobic conditions; NO_2^- and NO_3^- are reduced to N_2O which is emitted to the atmosphere (Skiba, 2008). Denitrification often occurs following a rainfall event due to an increase in soil moisture content and reduction in soil oxygen content (Thapa et al., 2015), or as a result of compaction or waterlogging (Bussell et al., 2021), providing enough substrate N is available. The majority of agricultural N_2O emissions are associated with the application of inorganic (i.e., synthetic) and organic N fertilisers (Lu et al., 2021) (Figure 1.2). As N is often limited in agricultural soils, fertilisers containing ammonium nitrate (NH_4NO_3) are added to supply available forms of N to support plant growth (Liu et al., 2014). Organic fertilisers provide a supply of C and OM which have additional benefits for soil health (Lal, 2016). Many studies have measured higher N_2O emissions from soil when organic fertiliser is applied compared to inorganic fertiliser (Yang et al., 2015; Zhou et al., 2017), as soil microorganisms use the labile C in organic fertiliser as a substrate for nitrification or denitrification (Hangs and Schoenau, 2022). The application of fertilisers with a high liquid content can further stimulate N_2O emissions via denitrification, as the soil oxygen content is more limited (Sextone et al., 1985). Nitrogen can also be lost via leaching or runoff, particularly if an excessive amount of fertiliser is applied, or if fertiliser application is followed by a heavy rainfall event (Qin et al., 2012). In the UK, farms in Nitrate Vulnerable Zones, those in areas at risk of agricultural NO_3^- pollution, face restrictions on how much N can be applied to soils to reduce the likelihood of excess leaching into the environment (DEFRA, 2021b). Despite this, however, N_2O emissions from the sector remain high. Emission factors (EFs) are used as a metric to represent the amount of a pollutant produced as a result of a certain activity (Skiba et al., 2012). The IPCC uses a default EF of 1 % for direct N_2O emissions from soil as a result of agricultural activity (IPCC, 1996; Skiba et al., 2012), suggesting that 1 % of the N applied is emitted as N_2O. Many studies have shown that this emission is highly variable, however, as a result of the fertiliser type used and the local climate and soil conditions (Buckingham et al., 2014; van der Weerden et al., 2016; Mazzetto et al., 2020). Bell et al. (2015), for example, found EFs of 0.2 % at sites in England fertilised with NH_4NO_3, and Buckingham et al., (2014) found EFs to range between 0.34 % and 37 %.

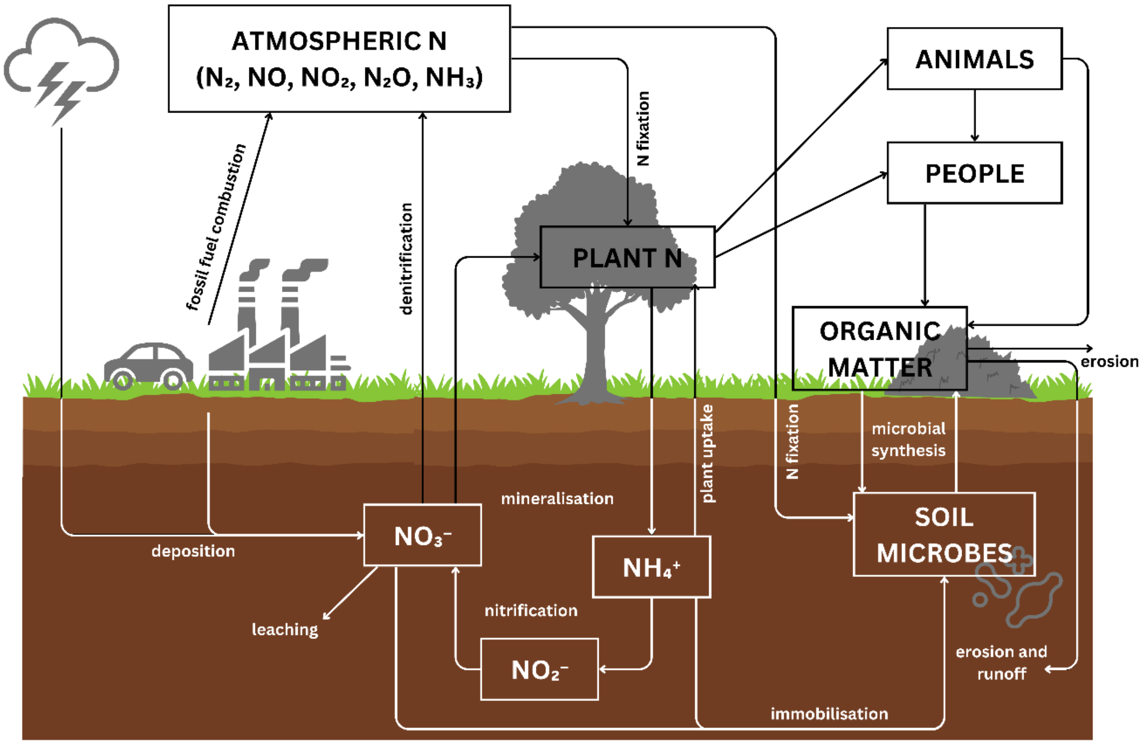

FIGURE 1.2 SOIL NITROGEN CYCLE (ADAPTED FROM BRADY AND WEIL, 2002).

1.4.4 Agriculture and climate change

The climate is changing as a result of anthropogenic activity. Burning fossil fuels, deforestation, and elevated GHG emissions have caused a rise in global temperatures, desertification, and more extreme and frequent weather events such as flooding (European Commission, no date b), all of which have wholly detrimental effects on agricultural production systems. Higher temperatures increase crop respiration and evapotranspiration rates, reduce the length of the growing season, and increase the presence of crop pests, all of which result in reductions of yield and income (Moore et al., 2017; Malhi et al., 2021; Habib-ur-Rahman et al., 2022). With climate change becoming more exacerbated every day, and its effects predicted to become even more extreme, farmers will need to adapt their management practices to mitigate against these negative impacts (Habib-ur-Rahman et al., 2022). This may be achieved by the adoption of best management practices such as diversifying crop rotations, improving water management, introducing measures to prevent soil erosion, and improving grazing management, all of which have been proposed as methods to increase agricultural resilience to climate change (Aryal et al., 2020; Bowles et al., 2020; Srivastav et al., 2021). Whilst the agricultural sector is vulnerable to the effects of climate change, it is also one of the only industries that can contribute to a reduction in these effects by sequestering C in the soil. The management practices proposed to achieve this are wide ranging, with their success highly dependent on the environment in which they are implemented (Section 1.6).

1.5 Measuring greenhouse gas emissions from agricultural soils

The suitability and effectiveness of a method for measuring soil GHG fluxes depends on the gas being measured and scale at which measurements are required. Eddy covariance (EC) is a standardised method for measuring CO_2 fluxes at the field scale (Baldocchi, 2014; Lucas-Moffat et al., 2018), and chamber methodologies are preferred for GHG measurements at the plot scale (Keane et al., 2019; Maier et al., 2022a). The pros and cons of these two methods are discussed in the following sections.

1.5.1 Eddy covariance

Eddy covariance flux towers are a well-established method for measuring fluxes of CO_2 and water vapour (Pastorello et al., 2020; Bastviken et al., 2022) and provide a reliable estimate of CO_2 fluxes at the field scale (Smith et al., 2010; Barba et al., 2017). Nitrous oxide and CH_4 fluxes can also be measured with EC, although with greater expense and lower accuracy as these gases are emitted at lower magnitudes than CO_2 (Laville et al., 1999; Eugster and Merbold, 2015; Krauss et al., 2016; Nemitz et al., 2018). Eddy covariance measures the movement of turubluent air eddies within the atmospheric boundary layer to determine the rate of vertical gas (CO_2) transport between the terrestrial ecosystem (i.e., the soil surface or vegetation canopy) and the atmosphere (Denmead, 2008) (Figure 1.3). The speed and direction of these air eddies – u (horizontal wind velocity in east-west direction), v (horizontal wind velocity in north-south direction), and w (vertical wind velocity) – are continuously measured by a three-dimensional sonic anemometer and the CO_2 concentration is sampled by an infrared gas analyser (Yu et al., 2013; Eugster and Merbold, 2015) (Figure 1.4). Additional micrometeorological measurements – i.e., net radiation, short- and long-wave incoming and outgoing radiation, air temperature and humidity, and soil temperature and moisture – are required for the calculation of turbulent fluxes, which are measured with a net radiometer, air temperature and humidity probes, and soil temperature and moisture probes respectively.

Fluxes measured with EC are processed and computed using open-source software (Yu et al., 2013), commonly EddyPro® (LI-COR Biosciences, 2019) when LI-COR flux towers are used. Net ecosystem exchange (NEE) is calculated as the CO_2 flux plus the CO_2 flux storage term (Nicolini et al., 2018), and is presented in 30-minute average values. Providing the height of the flux tower is below 10 m, the CO_2 storage term, or the change in CO_2 concentrration between the ground and sensor height, is likely to be negligible in comparison to the estimation of NEE, however is likely to influence CO_2 fluxes if the tower is taller than this, and so should be accounted for where appropriate (Nicolini et al., 2018). Carbon dioxide flux measurements incur large potential for error, often as a result of an inadequate sample size per averaging period, or systematic errors (Loescher et al., 2006; Mauder et al., 2013), so during the initial processing stage, the flux data is quality controlled to ensure that only high-quality data is used. If a Gill Windmaster sonic anonemeter is used to measure *w*, EddyPro® will apply a 'w-boost' bug correction (LI-COR Biosciences, 2024) whereby a double coordinate rotation is applied to correct any tilt or misalignment of the anemometer (Wilczak et al., 2001), an issue previously identified and thus rectified by the software. Quality control flags are used to identify high- or low-quality data (Foken et al., 2004) and outliers and clearly implausible values are removed according to Mauder et al. (2013) and Vickers and Mahrt (1997). Any time lags between the sonic anemometer and high-frequency data are corrected using cross-correlation, and fluxes will be corrected for high and low frequency co-spectral attenuation according to Moncrieff et al. (1997; 2004), and for air density fluctuations using the Webb-Pearman-Leuning correction (Webb et al., 1980). In addition, data is removed when: it is classified as a statistical outlier according to Papale et al. (2006); when the signal strength of the LI-COR is higher than the baseline value according to Ruppert et al. (2006); when it is beyond realistic thresholds (i.e., when the sensible heat flux (H) < -200 or > 450 W m^{-2}, when the latent heat flux (LE) < -50 or > 600 Wm^{-2}, or when

NEE < -60 or > 30 g m^{-2}). A footprint model will be produced to determine the area that contribute to the measured fluxes (Kljun et al., 2004). Finally, non-representative data will be removed – i.e., when over 20 % of the data within that 30-minute period was recorded outside of the site boundaries (Kljun et al., 2004).

Gaps in the dataset, either as a result of measurement error or the removal of outliers or low-quality data, are then filled, often using marginal distribution sampling (Reichstein et al., 2005; 2016) which involves simulating NEE values based on the existing high-quality measurements. To determine the amount of C fixed by plants through photosynthesis and released via respiration, NEE can be partitioned into gross primary productivity (GPP) and total ecosystem respiration (TER) (Smith et al., 2010) (Equation 1.1). The micrometeorological sign convention is often used for NEE, where positive values indicate CO_2 loss from an ecosystem and negative values indicate CO_2 assimilation (Baldocchi, 2003).

$$NEE = TER - GPP \qquad \text{(Equation 1.1)}$$

Following processing, the energy balance closure (EBC) method can be used to assess the quality of EC data at a study site (Aubinet et al., 2001; Wilson et al., 2002). This is based on the principle that under 'ideal' conditions the sum of the fluxes measured by EC (LE + H) are equal to the available energy measured by other means (Rnet – G). The closer to 1 the EBC is, the greater amount of energy exchange is being captured by the EC flux tower, and thus the measurements are more accurate. Typical EBC values reported for EC flux towers are between 0.7 and 0.9 (Wilson et al., 2002; Foken, 2008; Wagle et al., 2018), and thus values within this range are considered acceptable and accurate.

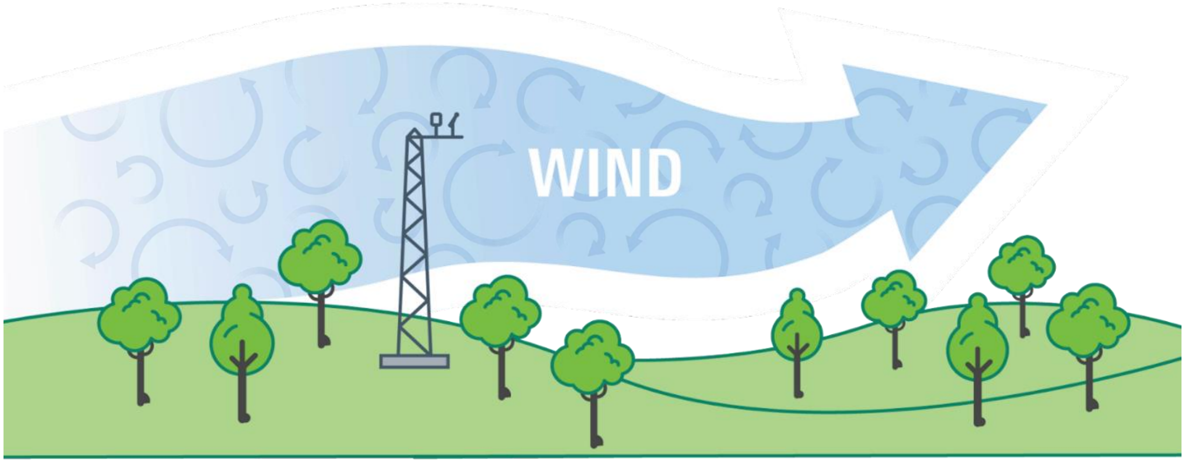

FIGURE 1.3 SCHEMATIC OF EDDY COVARIANCE FLUX TOWER WITH WIND EDDIES AND VEGETATION CANOPY (BURBA, 2022).

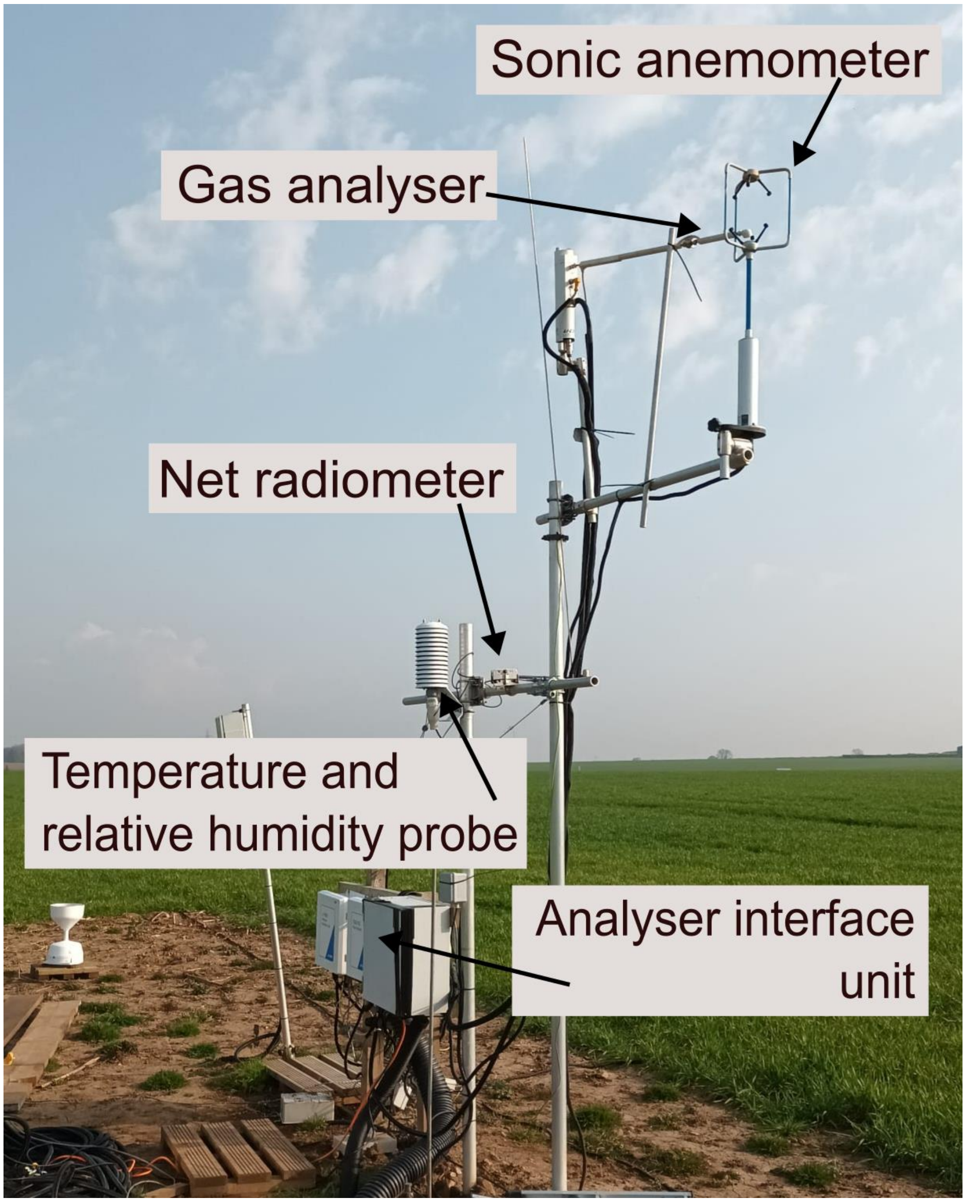

FIGURE 1.4 EXAMPLE EDDY COVARIANCE FLUX TOWER WITH KEY COMPONENTS LABELLED.

It is important to acknowledge that although EC is an established method for measuring field-scale CO_2 fluxes, there are some limitations of the method. Primarily, EC relies on homogeneity of the field being measured (Mauder et al., 2021), and thus if the field is not homogenous then the fluxes will not be representative. Furthermore, when the site being measured is homogenous, it can be difficult to independently quality control the data to verify whether the fluxes measured are in fact representative without additional equipment such as a personal CO_2 monitor. The reliability of EC for measuring NEE is addressed in the wider literature (Baldocchi, 2003; Aubinet et al., 2012; Mauder et al., 2021), however fluxes can be more difficult to verify on an individual site-by-site basis. It is also likely that fluxes measured during the nighttime are underestimated if the movement of air between the terrestrial environment and the atmosphere is not as turbulent as required. Wind speeds are often lower at nighttime, and thus there is the potential for fluxes during this time to be missed (Aubinet, 2008). These limitations should be considered when interpreting EC data, and strengthen the requirement for increased monitoring of NEE using EC flux towers.

1.5.1.1 Net ecosystem productivity

Net ecosystem productivity (NEP) provides an indication of the extent to which an agroecosystem is behaving as a C sink or source. Net ecosystem productivity accounts for lateral fluxes of C – i.e., C exported from the field in harvested or grazed biomass (C_H) and C_I via seed, organic fertiliser or excreta from grazing livestock – in addition to the vertical fluxes which make up NEE (Equation 1.2 – adapted from Evans et al., 2021) (Figure 1.5). Similary to NEE, the micrometeorological sign convention is used for NEP where a positive NEP indicates C loss and a negative value indicates C gain by the agroecosystem (as in Evans et al., 2021).

$$NEP = NEE + C_H - C_I \qquad \text{(Equation 1.2)}$$

Amongst the literature, the number of published studies that measure NEE is considerably greater than those that measure NEP, which can be attributed to the challenges associated with calculating C_H and C_I. To calculate C_H from croplands, the C content of a sample of the harvested biomass can be analysed and upscaled to the reported yield (Ceschia et al., 2010). Calculating the C removed as grazed biomass from managed grasslands can be considerably more difficult, however, and the methodology for doing so is not standardised. This is evidenced by the fact that multiple methods are used throughout the literature. Some studies, for example, measure the difference in height of a specific area of grass before and after grazing and multiply this by the C content of the grass (Skinner, 2008; 2013; de la Motte et al., 2016; Laubach et al., 2019; 2023) whereas others multiply the C content of the grass by a standardised pasture utilisation value (Rutledge et al., 2017; Wall et al., 2019; 2020a; 2023b). Imports of C to a field can be determined by analysing the C content of any added organic fertiliser or seed, and, for grazed grasslands by additionally calculating the proportion of C ingested via grazing that is returned to the soil as livestock excreta. There is no consensus within the literature as to how to derive this proportion, however (Skinner, 2008; 2013; Rutledge et al., 2015; 2017; de la Motte et al., 2016; Laubach et al., 2019; 2023; Wall et al., 2019, 2020a; 2023b); the values reported range between 30 % (Laubach et al., 2019; 2023) and 37 % (Skinner, 2008; 2013) and in some cases are calculated on a site-specific basis based on the number of days livestock are on the pasture and the metabolisable energy of the biomass (Rutledge et al., 2017; Wall et al., 2020a; 2023b).

The net ecosystem C balance (NECB) provides considerably more detail on the C sink or source potential of an agroecosystem compared to NEP, however NECB is reported even less frequently amongst the literature than NEP. The NECB accounts for all possible lateral C fluxes (Ciais et al., 2010a; Smith et al., 2010), considering exports as dissolved C in leachate and C in volatile emissions and CH_4, and imports as dissolved C in precipitation. These C data are difficult to measure, however, and so NEP therefore provides a more accessible estimate of whether an agroecosystem is accumulating or losing C relative to NECB (Chapin III et al., 2006; Ceschia et al., 2010).

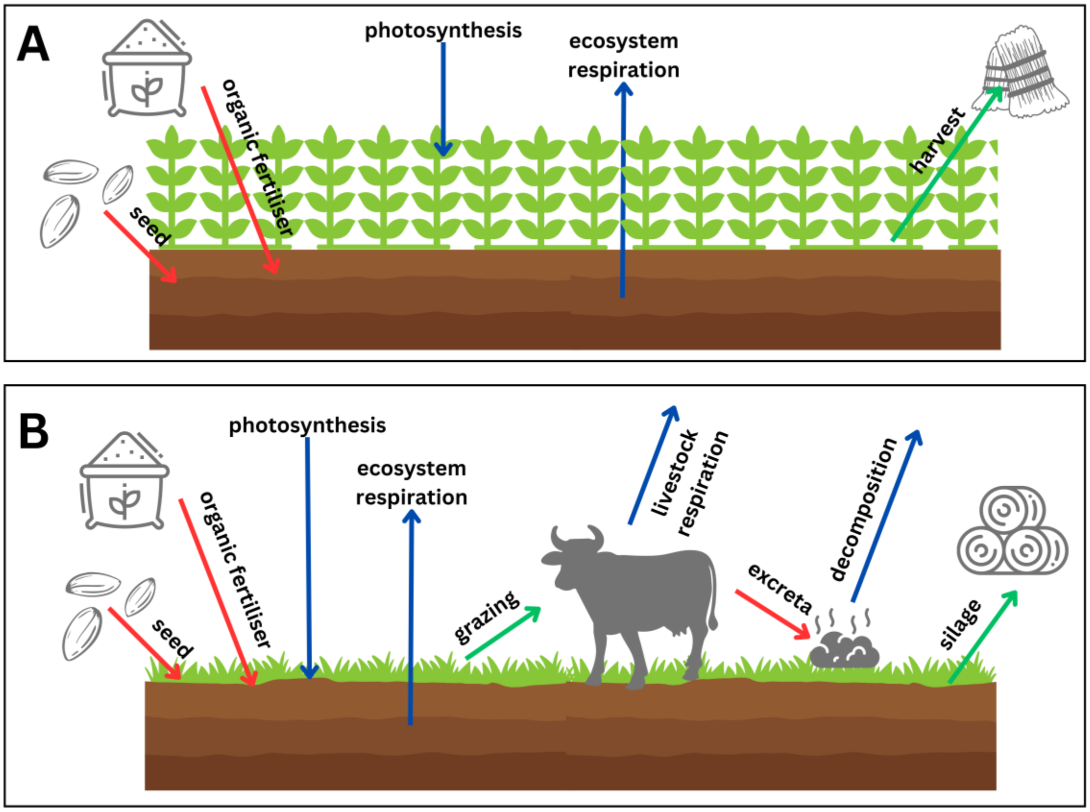

FIGURE 1.5 PATHWAYS OF CARBON EXPORT AND IMPORT TO (A) CROPLANDS AND (B) AGRICULTURALLY-MANAGED GRASSLANDS CONSIDERED IN NET ECOSYSTEM PRODUCTIVITY. BLUE ARROWS REPRESENT FLUXES CONSIDERED IN NET ECOSYSTEM EXCHANGE, RED ARROWS REPRESENT IMPORTS OF CARBON TO THE FIELD AND GREEN LINES REPRESENT EXPORTS OF CARBON FROM THE FIELD.

1.5.2 Chamber methodologies

Chamber-based approaches are utilised to measure gas fluxes at the plot scale, often being used to determine the influence of treatments or management practices on GHG fluxes (Chadwick et al., 2014). Compared to EC flux towers, GHG flux chambers take measurements from a small surface area (Smith et al., 2010; Sainju et al., 2021) and are able to capture fluxes of a lower magnitude, providing accurate measurements of CH_4, N_2O, and ammonia (NH_3) emissions, as well as CO_2 (Yu et al., 2013; Chaichana et al., 2018). There are multiple types of GHG flux chambers, including flow-through, dynamic and static closed chambers, all of which use different methods to measure the rate and concentration of fluxes. Static closed chambers are the most commonly used throughout the literature; a collar is inserted into the soil – usually to a depth of 5-10 cm – on top of which is placed a collar and a lid (Figure 1.6A). If required, extensions can be added between the collar and lid to accommodate tall crops over their growing season (Figure 1.6B) (Maier et al., 2022a). When the chamber lid is closed, gas accumulates in the chamber headspace and is sampled and analysed by a gas analyser (Collier et al., 2014; Sapkota et al., 2014). Fluxes from closed chambers are calculated according to Equation 1.3 (Denmead et al., 2008):

$$F_g = v(\rho_{g,o} - \rho_{g,t}) \div A \qquad \text{(Equation 1.3)}$$

where F_g is the flux density of the gas at the surface (kg m^{-2} s^{-1}), v is the volume flow rate (m^3 s^{-1}), $\rho_{g,o}$ is the gas concentration of the air leaving the chamber (kg m^{-3}), $\rho_{g,i}$ is the gas concentration of the air entering the chamber, and A is the surface area the chamber covers (m^2) (Denmead et al., 2008).

Greenhouse gas flux chambers can be manual or automatic. Manual chambers require frequent human input to place the lid over the chamber and extract the gas sample with a syringe (Clough et al., 2020) for further analysis in the laboratory, usually by gas chromatography (Sapkota et al., 2014). It is recommended that manual gas sampling is done between 10:00 and 12:00, as this is when flux rates are considered most representative of what is emitted over the course of a day (Sapkota et al., 2014; Reeves and Wang, 2015). It is also recommended that samples are taken at least once per week to capture temporal variations (Del Grosso and Parton, 2011). Manual chambers are affordable, however there is considerable potential for error to occur during the extraction, transportation and analysis of gas samples (Loescher et al., 2006) and sampling frequency is logistically limited as humans are involved (Gorres et al., 2016). Automatic chambers minimise the requirement for human input to close the chamber lid and extract samples, as chambers are programmed to close and extract gas samples on a set schedule (Denmead, 2008; Grace et al., 2020). Providing a gas analyser is connected, the samples can be analysed in-field, allowing for continuous sampling and any temporal variability in gas fluxes to be captured (Yao et al., 2009; Charteris et al., 2020). It has been established, for example, that N_2O fluxes are likely to peak following rainfall events (Smith and Dobbie, 2002; Huang et al., 2017; Westphal et al., 2018) which stimulate denitrification (Thapa et al., 2015); as the timing of weather events are difficult to predict, manual sampling strategies can easily miss key events (Asgedom et al., 2014; Grace et al., 2020). Furthermore, diurnal emissions of N_2O have been observed in several studies (Wu et al., 2021) and are more likely to be captured where sampling occurs continuously throughout the day and night. Due to the technology involved, automatic chambers are considerably more expensive than manual chambers, and so thus far have been rarely used throughout the literature, although this is likely to change as they become more widely available.

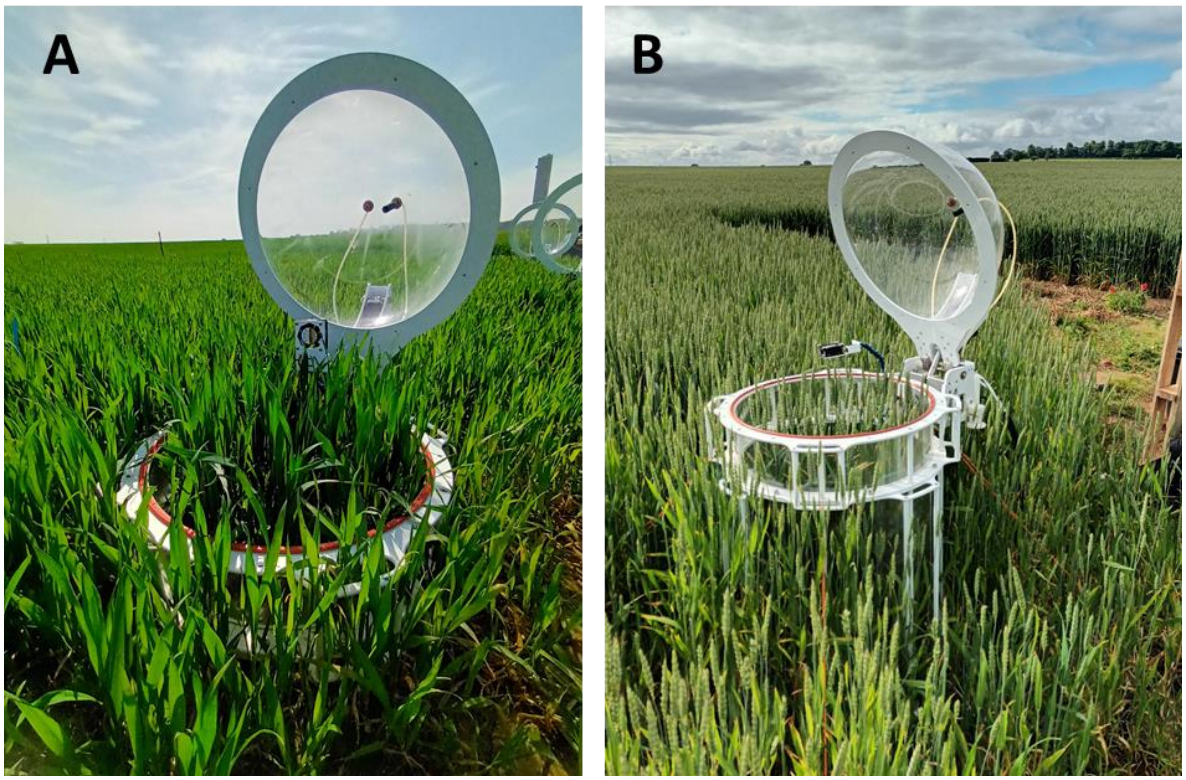

FIGURE 1.6 (A) AUTOMATED CLOSED GREENHOUSE GAS FLUX CHAMBER AND (B) WITH EXTENSION ATTACHED.

1.6 The role of agricultural land use management practices in reducing greenhouse gas emissions and increasing soil carbon storage

Reducing global GHG emissions will be essential to combat climate change and achieve net zero targets. As the agricultural sector is a key contributor to global GHG emissions, it offers considerable opportunities for emissions reductions. It is also critical that the sector provides sufficient food to meet societal needs and adheres to its environmental commitments. Agricultural soils are depleted of C, so SOC sequestration in agricultural soils is a promising route towards climate change mitigation (Minasny et al., 2017). The adoption of more sustainable agricultural production practices has considerable potential to sequester C back into these soils, simultaneously reducing the concentration of CO_2 in the atmosphere (Johnson et al., 2007; Sanz-Cobena et al., 2017) and improving soil health and resilience (Lal, 2006), and reducing GHG emissions, particularly N_2O and CH_4 (Table 1.1).

TABLE 1.1 EXAMPLES OF SUSTAINABLE AGRICULTURAL PRACTICES THAT HAVE THE POTENTIAL TO REDUCE CARBON LOSS AND GREENHOUSE GAS EMISSIONS FROM SOIL.

Aim	Practices
Reduce C loss	Reduce soil disturbance: convert cropland to grassland; conservation tillage (i.e., minimum tillage, reduced tillage, no till/direct drill) Include C4 crops in rotations Grow cover crops during fallow periods
Increase C input	Add organic amendments Reduce occurrence/length of fallow periods in between crops
Reduce N_2O emissions	Optimise fertiliser application: split application, reduce application rate Slurry treatment: nitrification inhibitors, plasma induction
Reduce CH_4 emissions	Slurry treatment: plasma induction

1.6.1 Land use change

The conversion of cropland to grassland is an established method for increasing soil C storage (Puget and Lal, 2005; Mudge et al., 2011; Wall et al., 2023a). Unlike croplands, agricultural grassland soils are not disturbed by tillage, which can reduce CO_2 emissions, and have continuous vegetation cover and longer root systems which facilitate greater C input to the soil and enhance C sequestration (Wall et al., 2023a). Carbon imports may be increased where livestock are grazed on the grassland, as a proportion of the grazed biomass is returned as excreta (Felber et al., 2016). This effect may be counteracted by elevated CH_4 emissions from enteric fermentation, however (Richmond et al., 2015).

1.6.2 Reduced tillage

Reduced tillage aims to limit the disturbance to soil structure, thus reducing the risk of deep soil C being exposed for oxidation (Stavi and Lal, 2013; Farhate et al., 2018; Nunes et al., 2020), whilst still incorporating the benefits associated with conventional tillage such as soil aeration and good water filtration. Reduced tillage encompasses conservation and minimum tillage, where the soil is not inverted and is ploughed no deeper than 25 cm, and no till, where the soil is not ploughed at all and instead direct drilling is used to plant seeds (Mangalassery et al., 2015). Conservation tillage practices also require at least 30 % of crop residues to be left on the soil surface (Triplett and Dick, 2008; Varvel and Wilhelm, 2011). Reduced and minimum tillage methods do involve a degree of soil disturbance, and so no till is often preferred for the purpose of increasing soil C storage (Soussana et al., 2007). In heavy soils, the benefits of no till may have trade-offs with increased soil compaction, however, which can cause further issues such as waterlogging (Nunes et al., 2015). Research on the influence of tillage on soil properties has typically focused on the impacts on soil C, however recent work has explored the influence of different tillage practices on soil N_2O and CH_4 fluxes (Franco-Luesma et al., 2020a; Pareja-Sanchez et al., 2020; Maucieri et al., 2021; Mirzaei et al., 2022; Tang et al., 2022). Compared to conventionally tilled soils, conservation tillage practices increase soil bulk density (Regina and Alakukku, 2010), which reduce the potential for waterlogging, and thus N_2O emission via denitrification, and enhance CH_4 oxidation and its retention in the soil (Lesschen et al., 2011; Jacinthe et al., 2013; Stavi and Lal, 2013; Mangalassery et al., 2014). On the other hand, N_2O and CH_4 emissions may be increased by conservation tillage practices; the creation of anaerobic conditions due to increased soil moisture content and reduced soil oxygen content may can facilitate the production of N_2O and its emission (Mangalassery et al., 2014; Lugato et al., 2018; Tian et al., 2020; Wang et al., 2021). Several studies have also observed higher N_2O production and emission from soils managed with no till due to higher earthworm concentrations and N availability (Lubbers et al., 2015; Guenet et al., 2020; Wang et al., 2021).

1.6.3 Crop management

Crop type affects the C and N dynamics of an agroecosystem. Global coverage of C3 plants – those that use 3-phosphoglyceric acid to fix C – is considerably larger than that of C4 plants – those that use malic or aspartic acid to fix C (Still et al., 2003; Leegood, 2004). The C uptake capacity of C3 plants (i.e., wheat, barley and most grasses) is lower than C4 plants (i.e., maize and sugarcane), however, due to the way C is fixed. C4 plants have the potential to sequester greater amounts of C into the soil as they minimise photorespiration (i.e., the amount of C lost during the photosynthetic process) (Still et al., 2003). Legumes, such as peas and beans, fix N in the soil which replenishes the N depleted by previous crops (Min et al., 2016). The provision of N by legumes reduces the requirement for N fertilisation, and subsequently reduces the upstream GHG emissions associated with the production of synthetic fertiliser and any emissions released following fertiliser application to soil. The findings reported by the literature on the effects of including cover crops in a rotation on soil C are mixed. Cover crops are grown during fallow periods to prevent extended periods of bare soil and improve soil health (Lal, 2015b; Popelau and Don, 2015; Daryanto et al., 2019). Some studies have reported an increase of SOC as a result of growing cover crops (Ruis and Blanco-Canqui, 2017; Jian et al., 2020), however others have observed increased emissions of CO_2 following cover crop harvest as crop residues rapidly decompose on the soil surface, resulting in C loss (Nilahyane et al., 2019; Blanco-Canqui, 2022). The type of cover crop therefore has a considerable impact on whether C is being lost from or added to an agroecosystem.

1.6.4 Improving carbon and nitrogen use efficiency

Carbon use efficiency (CUE) is the efficiency of which assimilated C is converted into biomass relative to the amount being released as CO_2 (Mganga et al., 2022). A higher CUE can increase soil C storage and reduce overall C losses from an agroecosystem (Kallenbach et al., 2019), as more C can be retained in the ecosystem. Research has shown that CUE can be improved with higher soil nutrient availability (Manzoni et al., 2012) and by liming (Moran-Rodas et al., 2023). Nitrogen use efficiency (NUE) is the efficiency of which applied N is assimilated by plants; a higher NUE indicates that crop N uptake is higher and subsequent fertiliser N loss as N_2O is reduced (Sharma and Bali, 2018). Optimising fertiliser application is one of the most effective methods for improving NUE (Rosolem et al., 2017; Cardenas et al., 2019). For example, applying fertiliser throughout a crop growing season, at times when nutrients are most required, rather than only at the start of the growing season (i.e., split application) better matches nutrient application to crop requirement at certain growth stages, and so reduces the risk of excess N being present in the environment (The Fertiliser Institute, 2017; Sharpley, 2018). Furthermore, the N content of livestock waste can be highly variable, so analysing the nutrient content of manures and slurries prior to application is also recommended to avoid an over-application of N, thus reducing the risk of N leaching, runoff or emission as N_2O (Govindasamy et al., 2023).

1.6.5 Livestock waste management

As discussed (Sections 1.4.1 and 1.4.2), livestock wastes are used as organic fertilisers to reduce farm waste and to supply OM and C to soil (Case et al., 2017; Chew et al., 2019). The storage and application of livestock waste for use as organic fertiliser are significant sources of N_2O and CH_4 emissions (Flessa et al., 2002; Amon et al., 2006; Webb et al., 2011). Although not a GHG, emissions of NH_3 associated with organic fertiliser use are also of concern, as NH_3 can be oxidised to N_2O via NO_3^- (The Royal Society, 2020). Several methods have been proposed to regulate the chemical and microbial processes that release N_2O, CH_4 and NH_3 in order to reduce their emissions. Covering slurry or manure storage tanks with a lid, floating cover or plastic film can considerably reduce NH_3 emissions, as NH_3 is concentrated underneath the cover and so further NH_3 production and its release is suppressed (Misselbrook et al., 2016; Kupper et al., 2020). This has the potential to reduce N_2O emission as less NH_3 is available for further oxidation to N_2O. The acidification of livestock waste can also limit NH_3 volatilisation and inhibit methanogenesis and thus CH_4

production (Misselbrook et al., 2016; Sokolov et al., 2019). There are potential trade-offs depending on the acid used to acidify waste, however, as reductions in CH_4 may be offset by increases in N_2O (Dalby et al., 2022). Alternatively, nitrification inhibitors such as dicyandiamide (DCD) and 3,4-dimethylpyrazole phosphate (DMPP) may be added to inorganic and organic N fertilisers to limit and suppress nitrifier activity, which can subsequently reduce N_2O emissions (Misselbrook et al., 2014; Zhou et al., 2020). The treatment of livestock waste with plasma induction is a recently developed method, and has the potential to reduce CH_4 and NH_3 emissions during the storage of livestock waste and after application to soil (Graves et al., 2018). The plasma induction process uses electricity to create nitrogen oxide, which combines with NH_3 to form involatile ammonium nitrate, reducing NH_3 emissions and increasing the amount of inorganic N available for uptake by the crop (Graves et al., 2018). There is the potential for this NO_3^- enrichment to result in an increase of N_2O however, and thus outweigh the benefits of reduced NH_3 emissions (Graves et al., 2018; Hiis et al., 2023). The plasma induction process also inhibits methanogenesis occurring during storage, so CH_4 cannot be produced and dissolved into livestock waste and then emitted on application (Tooth, 2021). Existing research has shown that treating cattle slurry with plasma induction can reduce NH_3 emissions (Tooth, 2021), however, the effects of treating other types of livestock wastes, such as pig slurry, on gases other than NH_3 are relatively unknown as the technology is still being developed.

1.6.6 Summary

Whilst there are clear benefits for GHG emissions reduction and increased soil C storage as a result of adapting agricultural land management practices, there is also considerable potential for these practices to have unintended consequences, primarily in the form of trade-offs with increased emissions of other GHGs. For example, reduced tillage practices are likely to increase N_2O and CH_4 emissions (Mangalassery et al., 2014; Tian et al., 2020), the decomposition of cover crop residues can increase CO_2 emissions (Nilahyane et al., 2019; Blanco-Canqui, 2022), and reduced NH_3 and CH_4 emissions associated with the application of plasma-treated pig slurry may be offset by elevated N_2O (Graves et al., 2018; Hiis et al., 2023). The addition of manures to managed grasslands in the form of excreta from grazing livestock provides a valuable input of C to the soil, however these manures can also decompose on the soil surface, emitting CO_2 to the atmosphere (Figure 1.4). Furthermore, studies often find conflicting evidence as to whether a management practice is successful at increasing SOC content or reducing GHG emissions. A considerable proportion of the existing research focuses on the effects of a management practice on one output only (i.e., SOC, CO_2, N_2O or CH_4), with very few considering the effects on more than one of these parameters. This lack of evidence makes it difficult to establish whether a practice is reducing or increasing GHG and C emissions overall, which causes further problems for policy developers and decision makers on which practices should be promoted and incentivised for an environmental benefit.

1.7 Knowledge gaps

Although continuously developing, the understanding of the influence of agricultural management practices on soil GHG emissions is not comprehensive. The existing literature reporting GHG fluxes from agricultural soils is concentrated in the USA, China, Germany and New Zealand, with considerably fewer measurements from Europe (excluding Germany), South America and the UK. It is therefore difficult to discern the scope of GHG emissions from agricultural soils in the context of the climate and soil type of these countries. It will be necessary to measure the GHG emissions associated with the agricultural practices currently being used in these countries to assess if they can be reduced by implementing best management practices. This will be critical for informing UK policy and advising governments on which practices should be incentivised in AES (Section 1.3). This thesis was therefore designed to fill this research gap and provide information that can be used to formulate future AES and achieve net zero.

The existing research using EC flux towers to measure GHG emissions from UK land has predominantly focused on the influence of the water table level on CO_2 fluxes from peatlands (Helfter et al., 2015; Flechard et al., 2019; Peacock et al., 2019; Evans et al., 2021). The published research reporting GHG emissions measured with EC from UK croplands or managed grasslands, and how these are influenced by agricultural management practices, is very limited (Ceschia et al., 2010; Eugster et al., 2010). Most studies measuring GHG emissions from UK soils have employed manual chamber methodologies due to the affordability and accessibility of this equipment relative to EC. Much of this research has compared the effect of a treatment or management practice on simultaneously measured CO_2, CH_4 and N_2O emissions – i.e., inorganic versus organic fertilisers (Jones et al., 2007; Louro et al., 2013; Ball et al., 2014), agricultural land use type (Dinsmore et al., 2009; Levy et al., 2011; Mills et al., 2011; Cowan et al., 2017), vegetation type (Dlamini et al., 2021; Dlamini, 2022; Button et al., 2023), tillage method (Ball et al., 2014; Alskaf et al., 2021) and the presence of livestock (Marsden et al., 2017; 2019). The extent of the current research using flux chambers to measure GHG emissions from agricultural soils in the UK is low relative to that from other countries. There is a clear knowledge gap surrounding CO_2, CH_4 and N_2O fluxes from agricultural soils in the UK and how these are affected by management practices. It will be critical to measure these GHG fluxes to understand how and where agriculture can reduce its GHG emissions and SOC loss to achieve net zero. In addition, as manual chambers are used more than automated chambers, there is a general knowledge gap surrounding the impact of agricultural management practices on diurnal GHG emissions, which are likely to vary as a result of environmental factors (Wu et al., 2021).

1.8 Research questions and approach

1.8.1 Overall thesis aim

The overall aim of this research project is to address the knowledge gaps highlighted above by assessing how NEP and GHG emissions from agricultural soils in the UK are affected by the management practices used, including the type of crop grown, and how these fluxes are related to the climate and soil conditions.

1.8.2 Research questions

This thesis aims to answer six key research questions:

1. How do climate, soil type and agricultural management influence the NEP of global agricultural soils?

2. How does soil type affect the NEE and NEP of maize grown for bioenergy in the UK?

3. How does crop type affect the NEE and NEP of agricultural land in the UK?

4. How does agricultural land use affect the NEE and NEP of agricultural land in the UK?

5. How does fertiliser type influence GHG fluxes from a winter wheat crop grown on a mineral soil in the UK?

6. What are the implications of environmental and management factors on C fluxes and GHG emissions from temperate agricultural systems for future research and policy development?

The five main research questions are addressed in Chapters 2-6 and discussed further in a synthesis of the findings (Chapter 7). Each chapter also has its own individual objectives or hypotheses. Question 6 is considered in each of the main research chapters and the final synthesis chapter.

1.8.3 Research approach

First, a meta-analysis of the existing literature was conducted to review the effects of climate, soil type and agricultural management practices on the NEP of croplands and managed grasslands around the world (Question 1, Chapter 2). This provided context for the results of subsequent research in this thesis which was conducted in the UK. Three observational studies were carried out at the University of Leeds (UoL) Research Farm using EC to measure CO_2 fluxes: the first measuring the NEE and NEP of a bioenergy maize crop over its growing season and comparing these results to those of a bioenergy maize crop grown in East Anglia on a drained peatland soil (Question 2, Chapter 3); the second measuring the NEE and NEP of a cropland over 2.5-years which included the following crops: maize, winter wheat and vining pea (Question 3, Chapter 4); and the third measuring the annual NEE and NEP of a cropland and an adjacent cut and grazed permanent pasture (Question 4, Chapter 5). A short-term experiment was also conducted at the UoL Research Farm during a single winter wheat growing season, where automated flux chambers were used to determine the influence of fertiliser type on N_2O, CH_4 and CO_2 emissions, in particular whether treating the pig slurry with plasma induction resulted in a reduction of GHG emissions following its application to land (Question 5, Chapter 6).

1.8.4 Study sites

To answer research questions 2-5 (Chapters 3-6) GHG flux measurements were taken at the UoL Research Farm. The UoL Research Farm is a 320 ha commercial mixed arable and pasture farm near Tadcaster, Yorkshire, Northeast England, UK (Figure 1.7). The soil is mainly a loamy calcareous brown earth, typically 50-90 cm deep, and is underlain by dolomitic limestone (Holden et al., 2019). The farm has a temperate oceanic climate, with mild winters and warm summers (Beck et al., 2018). Data was collected in a crop field (CF) and permanent pasture (PP). The crop field (53°51’56.26” N, 1°19’28.22” W; 49 m elevation, 10.4 ha) has been managed continuously under crop rotation with conventional tillage since 1994 (when set-aside land was no longer a requirement). An EC flux tower was installed in CF in 2020; an LI-7200 RS enclosed infrared CO_2/H_2O gas analyser (LI-COR Biosciences, USA) was used to measure CO_2 fluxes (sampled at 10 Hz) between 2021 and 2023 (Chapters 3, 4 and 5). Also recorded were: atmospheric turbulence and sonic temperature, measured with a Gill Windmaster three-dimensional sonic anemometer (Gill Instruments Ltd., UK); energy fluxes, including long- and short-wave incoming and outgoing radiation, measured with an SN-500 net radiometer (Apogee Instruments, USA); and air temperature and humidity, measured with an HMP155 temperature and humidity probe (Vaisala, Finland). Sensors were mounted on an extendable mast, the height of which was altered over the measurement period to ensure a minimum of 2 m between the sensors and crop canopy. Soil temperature and moisture content were measured with TEROS 11 temperature and moisture probes (METER Group Inc., USA). All data were combined by a CR1000X data logger (Campbell Scientific, USA) via a Smartflux 2 processing computer (LI-COR Biosciences, USA). The permanent pasture (53°51'58.64" N, 1°19'11.08" W; 46 m elevation, 3.05 ha) has been managed with alternating periods of sheep grazing and growth/harvest for silage since 1998. An EC flux tower was installed in PP in 2021 and used to measure CO_2 fluxes between 2021 and 2022 (Chapter 5). The EC setup in PP was identical to that in CF.

Automated flux chambers were used to measure fluxes of CO_2, N_2O and CH_4 from winter wheat grown in CF during summer 2022 (Chapter 6). Nine circular collars (0.5 m diameter) were inserted into the soil to a depth of 0.1 m and Eosense eosAC-LT chambers with an internal volume of 0.072 m^3 (Eosense, Canada) were attached. One month into the measurement period, vertical extensions (0.7 m height) were attached between the collar and lid to accommodate for the increased height of the winter wheat, which increased the internal chamber volume to 0.209 m^3. Over the measurement period the nine chambers were sampled in turn in a continuous loop sequence, controlled by an Eosense eosMX-P multiplexer and eosLink-AC software (Eosense, Canada). The multiplexer was connected to a Picarro G2508 GHG analyser (Picarro, USA) and so CO_2, N_2O and CH_4 samples were analysed immediately on-site. Soil moisture and temperature were measured next to each GHG chamber using TEROS 11 moisture and temperature sensors (METER Group Inc., USA).

In addition to UoL Research Farm, measurements were also taken from a farm in East Anglia during the 2021 maize growing season (Chapter 3) by flux scientists at UKCEH (Ross Morrison, Brenda D'Acunha, Alex Cumming and Chris Evans). The site name has been anonymised for the purposes of this research, and so the farm is subsequently referred to as the peat site (PS). The PS is a commercial arable and horticultural farm located in the East Anglian Fens, Eastern England, UK (Figure 1.7), and is situated on lowland peat which was drained in the 1940s for agriculture (Evans et al., 2016). Similar to UoL Research Farm, PS has a temperate oceanic climate. Fluxes of CO_2 (sampled at 20 Hz) were measured with an LI7500A open path CO_2/H_2O gas analyser (LI-COR Biosciences, USA) in one of the crop fields at PS (52°26'40.89" N, 0°25'26.39" E, -2 m elevation, 41.2 ha). In addition, a Gill Windmaster three-dimensional sonic anemometer (Gill Instruments Ltd., UK) was used to measure atmospheric turbulence and sonic temperature, an SN-500 net radiometer (Apogee Instruments, USA) was used to measure energy fluxes (as at UoL Research Farm), and air temperature and humidity were measured with an HMP155 temperature and humidity probe (Vaisala, Finland). Similar to at UoL Research Farm, the height of the sensors at PS were altered over the measurement period to ensure a minimum distance of 2 m between the sensors and crop canopy. Soil heat flux was measured using HFP01-L heat flux plates (Hukesflux, Netherlands in Campbell Scientific, USA), soil temperature and soil moisture were measured using TDT soil temperature and moisture sensors (Acclima, USA), and water level was measured with a CS451 pressure transducer (Campbell Scientific, USA). All data were combined by a CR3000 data logger (Campbell Scientific, USA). Data collected at PS during the maize growing season of 2021 was used in this research (Chapter 3). Processing of the data collected at PS was also conducted by the aforementioned flux scientists from UKCEH.

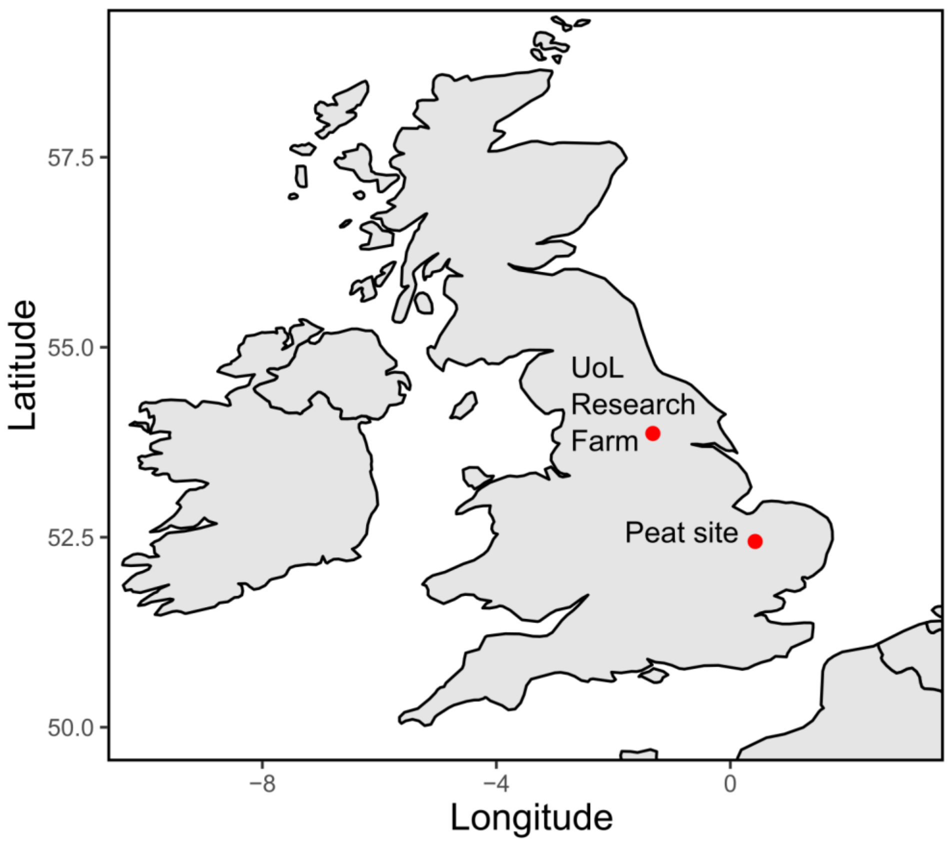

FIGURE 1.7 MAP OF THE UK SHOWING LOCATION OF UNIVERSITY OF LEEDS (UOL) RESEARCH FARM AND THE PEAT SITE.

1.8.5 Research methodology

Question 1 was addressed by conducting a meta-analysis (Chapter 2). Meta-analysis is an established approach for summarising and statistically comparing the results of multiple publications (Weerasinghe, 2014). The focus of this chapter was to explore the results of global research, and so a meta-analysis was the most feasible option to achieve this within the context of the PhD timeframe. Conducting a meta-analysis also allows the results of the PhD research (conducted in the UK) to be placed within a global context, highlighting its importance and urgency.

Question 2 was addressed by measuring NEE with EC flux towers (Chapter 3). Fluxes were measured at two farms where maize was grown for bioenergy over the 2021 growing season (May to October) – one farm with mineral soil (MS, UoL Research Farm) and one on peat (PS). In addition to CO_2 fluxes, samples of maize were taken from both sites prior to harvest and were analysed in the laboratory for moisture and total C content. The C content of each crop was scaled to the reported yield of each field to calculate C_H, and this was used to calculate the NEP of both fields. To contextualise the sites, soil samples were taken from both locations and analysed in the laboratory for OM content, pH, bulk density, total C, total organic C, total N, Olsen's phosphorus and plant-available N.

Question 3 was also addressed by measuring NEE with EC flux towers, this time at the UoL Research Farm only (Chapter 4). To evaluate the impact of crop type on NEE and NEP, CO_2 fluxes were measured from 2021 to 2023 in a crop field (CF). This provided measurements of NEE over the maize, winter wheat and vining pea growing seasons, and during the fallow periods between these crops. At the end of each crop growing season, biomass samples were collected and analysed for moisture and C content in the laboratory. The C content was scaled to the reported yield of each crop to calculate C_H, and any C_I as organic fertiliser or seed were used to calculate NEP for each crop growing season. Net ecosystem productivity was also calculated for the fallow periods in between the crop growing seasons. Soil samples were taken from both fields and analysed in the laboratory for OM content, pH, bulk density, total C, total organic C, total N, Olsen's phosphorus and plant-available N to provide contextual site information.

Question 4 was addressed by measuring NEE with EC flux towers at the UoL Research Farm (Chapter 5). To evaluate the impact of agricultural land cover on NEE and NEP, CO_2 fluxes were measured in CF and a neighbouring cut and grazed permanent pasture (PP). Measurements recorded between 11/10/2021 and 10/10/2022 were used to compare the annual field-scale NEP of CF and PP, which encompassed the winter wheat growing season in CF and grazing and cutting events in PP. To calculate C_H via sheep grazing, exclusion cages were used to prevent livestock from grazing certain areas of the field; a quadrat was used to sample grass from inside and outside of the exclusion cages, and the difference in the weight of these was determined as the amount ingested via grazing. The moisture and C content of the grass samples were analysed in the laboratory and the C content scaled to the estimated amount ingested by sheep and the yield when the field was harvested for silage in July 2022. In addition to the existing soil measurements taken in CF (Question 3, Chapter 4), soil samples were taken from PP and analysed in the laboratory for OM content, pH, bulk density, total C, total organic C, total N, Olsen's phosphorus and plant-available N.

Question 5 was addressed by measuring fluxes of CO_2, CH_4 and N_2O with automated static closed flux chambers from winter wheat treated with organic and inorganic fertilisers (Chapter 6). The experiment was conducted in CF at the UoL Research Farm during the winter wheat growing season (2022). There were three replicates of three treatments (i.e., inorganic fertiliser only, untreated pig slurry and inorganic fertiliser, and plasma-treated pig slurry and inorganic fertiliser); the treatments were applied inside the chamber collars and to a neighbouring plot to compare yield and to account for any potential greenhouse effect of the chambers. The chambers sampled GHGs on a loop sequence, providing one measurement per chamber every 2-hours. At the end of the experiment, biomass was sampled from the neighbouring plots, yield determined, and was analysed for moisture and total C and N content in the laboratory. Grain samples were also analysed for protein content in the laboratory.

1.9 Thesis outline

This thesis consists of the research findings of five manuscripts and ends with a synthesis of the main findings, implications for future policy and suggestions for further research, as outlined below.

Chapter 1: Introduction

A background to the literature and current research gaps were explained. Research questions and research methodology were also outlined.

Chapter 2: Factors affecting the net ecosystem productivity of agroecosystems on mineral soils: A meta-analysis

The impacts of climate, soil type and agricultural management on the field-scale NEP of global croplands and managed grasslands are investigated, synthesising the results of a meta-analysis.

This chapter has been submitted for review to Agroecology and Sustainable Food Systems as “Factors affecting the net ecosystem productivity of agroecosystems on mineral soils: A meta-analysis” (Isobel L. Lloyd, Ross Morrison, Richard P. Grayson, Marcelo V. Galdos, Pippa J. Chapman).

Chapter 3: Maize grown for bioenergy on peat emits twice as much carbon as when grown on mineral soil

The growing season NEE and NEP of maize grown for bioenergy on two contrasting soil types – mineral soil and peat – were measured and compared.

This chapter has been published in Global Change Biology Bioenergy as “Maize grown for bioenergy on peat emits twice as much carbon as when grown on mineral soil” (Isobel L. Lloyd, Ross Morrison, Richard P. Grayson, Alex M. J. Cumming, Brenda D’Acunha, Marcelo V. Galdos, Chris D. Evans, Pippa J. Chapman; 2024; https://doi.org/10.1111/gcbb.13169).

Chapter 4: Net ecosystem productivity of a UK cropland over 2.5 years

The annual NEE and NEP of a cropland growing maize, winter wheat and vining pea was measured and the NEP of the three crops compared.

Chapter 5: Comparing net ecosystem productivity of neighbouring arable and pasture systems over one year

The annual NEE and NEP of a neighbouring cropland and cut and grazed permanent pasture were measured and compared.

Chapter 6: Nitrous oxide and methane fluxes from plasma-treated pig slurry applied to winter wheat

Fluxes of N_2O, CH_4 and CO_2, and crop yield, were measured from winter wheat treated with different fertilisers.

This chapter has been published in Nutrient Cycling in Agroecosystems as "Nitrous oxide and methane fluxes from plasma-treated pig slurry applied to winter wheat" (Isobel L. Lloyd, Richard P. Grayson, Marcelo V. Galdos, Ross Morrison, Pippa J. Chapman; 2024; https://doi.org/10.1007/s10705-024-10363-8).

Chapter 7: Synthesis

The results of Chapters 2-6 are presented and placed in the wider context of C loss and GHG emission from agricultural soil in the UK and globally. The implications and limitations of the research are discussed and areas for future work are suggested.

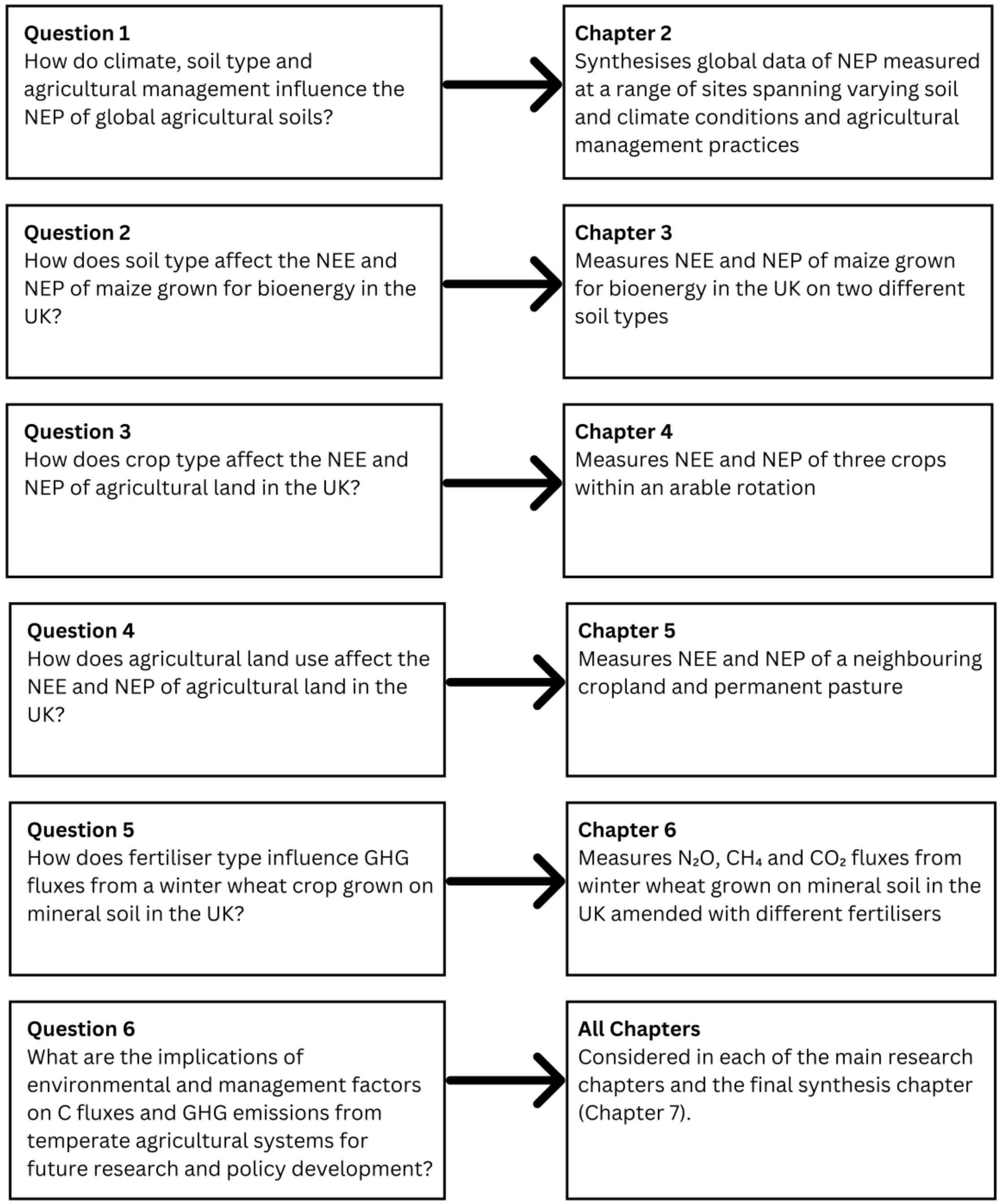

FIGURE 1.8 RESEARCH QUESTIONS AND CORRESPONDING CHAPTERS IN WHICH QUESTIONS ARE ADDRESSED.

Chapter 2 Factors affecting the net ecosystem productivity of agroecosystems on mineral soils: A meta-analysis

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Abstract

To optimise agricultural land management for soil carbon (C) sequestration, it is necessary to identify whether agroecosystems are accumulating or gaining C. This can be done by determining an agroecosystem's net ecosystem productivity (NEP). This study collated data from 40 papers, containing 242 annual measurements of NEP, to assess the impact of climate, soil type and management on the annual NEP of croplands and managed grasslands. Croplands lost significantly more carbon (110 g C m^{-2}) than managed grasslands (29.9 g C m^{-2}) and there was little influence of climate, soil or management practice on annual NEP. For agroecosystems to sequester C, there should be a shift in focus towards implementing management practices that increase C retention within agroecosystems.

2.1 Introduction

Soil is a major component of the global carbon (C) cycle; the top three metres store around 2500 Gt of soil organic C (SOC), which exceeds that stored by the atmosphere and vegetation combined (Jobbagy and Jackson, 2001; Scharlemann et al., 2014). Soil organic C is important for soil structure, nutrient provision and ecosystem functioning (Billings et al., 2021), and can help mitigate against drought by increasing soil water holding capacity (Iizumi and

Wagai, 2019). Global land use change, particularly the conversion of non-agricultural land to agricultural land, has led to an estimated loss of 50 Gt C, equivalent to 186 Gt carbon dioxide (CO_2), between 1860 and 2020 (Smith et al., 2016). The decline in SOC is due to an increase in the decomposition rate of soil organic matter (SOM) and a decrease in the amount of C being returned to the soil. In agriculture, this can be attributed to tillage, which disturbs and increases the oxygenation of the soil profile, and biomass removal via harvesting or grazing, which reduces the amount of litter returned to the soil (Stavi and Lal, 2013). Plants assimilate C during photosynthesis, however increased rates of biomass removal associated with increased yields from agricultural intensification mean that less organic matter (OM), and therefore organic C, is being returned to the soil, thus reducing net C storage within agroecosystems (Haberl et al., 2007; Ray and Foley, 2013). Furthermore, higher stocking densities, nutrient fertilisation and mowing frequency associated with the intensification of livestock farming has increased grass utilisation (Soussana and Lemaire, 2014; Manning et al., 2015). Severe depletion of the SOC pool is of global concern, as it degrades soil quality, leading to a decline in soil fertility and crop yield, and an increased reliance on fertiliser application. Such declines in soil health also compromise soil hydraulic functioning – i.e., infiltration, water storage and runoff – which increase the risk of soil erosion and flooding (Ogle et al., 2019). This can subsequently increase greenhouse gas (GHG) emissions from the agricultural sector, further contributing to anthropogenic climate change.

To meet climate targets, including the UK's aim to achieve net zero GHG emissions by 2050 or earlier (Climate Change Committee, 2020), a global reduction in GHGs (including CO_2) together with an increase in SOC storage, called 'negative emissions' or 'CO_2 removal', is required. Whilst the agricultural sector currently contributes to climate change, it also has considerable potential to mitigate against it. Policies such as the 4 per 1000 Initiative place a strong focus on the use of agricultural soils for GHG removal via SOC sequestration (Minasny et al., 2017). There are several 'climate-smart' farming practices which have been shown to enhance SOC sequestration under certain conditions (Chapman et al., 2018). Such practices include minimal tillage or no till (Nunes et al., 2020), the use of cover crops during fallow periods (Lugato et al., 2018), greater crop residue retention (Qiu et al., 2020), increasing plant species diversity to include those with deeper roots and greater root mass (Smith, 2004) and rotational grazing or mixed agriculture (Albanito et al., 2022). Soil organic

C sequestration has additional benefits of improving soil health and food security (Lal, 2016), however the rate of sequestration depends on soil texture, soil drainage characteristics, climate, and the length of time that the management practices have been implemented for. To understand where and how agricultural emissions can be reduced and soil C sinks increased, the C sequestration potential of climate-smart management practices across contrasting soils and climate conditions must be evaluated.

To establish whether an ecosystem is acting as a source or sink of CO_2, net ecosystem exchange (NEE) is determined as the difference between the CO_2 flux assimilated by photosynthesis (gross primary productivity – GPP) and respired from plant and soil processes (total ecosystem respiration – TER) (Eugster and Merbold, 2015). The magnitude of GPP and TER are controlled by a combination of crop type, climate, soil type and management (Davidson and Janssens, 2006). Climate conditions and soil texture regulate SOM mineralisation; warmer and wetter climates and fine-textured soils create favourable conditions for soil microbial activity and subsequently increase TER (Dilustro et al., 2005; Jager et al., 2011; Shakoor et al., 2021). Temperature influences crop growth rate and GPP (Baly, 1935). Intensively managed grasslands typically have higher SOC stocks than croplands as they have longer periods of vegetation cover and less frequent or intense soil disturbance (Guo and Gifford, 2002; Ciais et al., 2010a). Vegetation type influences GPP due to variations in photosynthetic rate, phenology, and length of the growing season (Wohlfahrt et al., 2008; Prade et al., 2017), and TER can be enhanced by greater soil disturbance via intensive tillage (Abdalla et al., 2013; Mohammed et al., 2021). Furthermore, grazed grasslands are likely to have a faster turnover of C than cut grasslands as non-digestible C is returned to the soil via excreta (Chang et al., 2015).

At the field scale, eddy covariance (EC) flux towers are widely used to determine NEE (Moncrieff et al., 1997). In agroecosystems, however, NEE does not account for lateral C fluxes, which are important for understanding whether a system is accumulating or losing C, and thus its potential to mitigate climate change. Net ecosystem productivity (NEP) provides an estimate of the C sink or source strength of an ecosystem and considers lateral fluxes of C – C imported via organic amendments and livestock excreta (C_I), and C exported in harvested or grazed aboveground biomass (C_H) – as well as NEE (Evans et al., 2021). Alternatively, the net ecosystem C balance (NECB) can be calculated, which accounts for all possible lateral C fluxes (Ciais et al., 2010a; Smith et al., 2010). In addition to the lateral fluxes in NEP, NECB considers: C_H as dissolved organic and inorganic C in leachate and C in volatile organic emissions, and C_I as dissolved organic and inorganic C in precipitation and C in seeds. Net ecosystem productivity therefore provides a more accessible estimate of whether an agroecosystem is accumulating or losing C, as the lateral C fluxes it considers are considerably larger and easier to measure than those included in NECB (Chapin III et al., 2006; Ceschia et al., 2010). Amongst the literature, NEP is reported less frequently than NEE, however must be measured to gain a comprehensive overview of the C sink or source strength of agroecosystems.

How NEP varies as a result of climate, soil type, land use and/or the agricultural management practices used is poorly understood, yet without this knowledge it is difficult to identify the practices that promote C sequestration, and this information is urgently needed for effective policy decision making. To truly understand how agriculture can contribute to increased C sequestration, we first need an appreciation of the net C sink or source strength of agroecosystems from a combination of climates, soil types and management practices. This study collated published data to (i) assess the impact of climate, soil and agricultural management (including land use, crop cover, tillage intensity, fertilisation, and grassland management) on the annual NEP of global croplands and managed grasslands, and (ii) identify directions for future research.

2.2 Methods

2.2.1 Data collection

Publications were collated from Web of Science (Clarivate, 2022) using three separate search terms (Table 2.1) to conduct a rapid meta-analysis. All publications considered were peer-reviewed journal articles published before 01/09/2023. The search terms were designed to focus the output of the literature search to identify the most relevant publications for this meta-analysis. The authors acknowledge, however, that due to the specific search terms used (Table 2.1), some publications containing relevant information may not have been identified by the literature search and subsequently not included in this review. The initial search produced 719 publications. Given the overwhelming evidence that the C source or sink strength of peat is primarily controlled by drainage and the water table level (i.e., lowering the water table of peat soils can effectively reduce CO_2 emissions (Evans et al., 2021)), publications that measured C fluxes of agroecosystems on peat were discarded and only those on mineral soil were considered. Additionally, publications measuring C fluxes of agricultural land used to grow perennial grasses for bioenergy production were excluded, as the focus of this analysis is on food and fodder production systems. In instances where some measurements included in a publication fulfilled the criteria and some did not (i.e., multiple sites were measured with some on mineral soil and some on peat, or multiple crops were measured with some grown for bioenergy and some for food), only the measurements from site years that fulfilled the criteria were included.

Each publication was then screened against the following criteria: (1) the publication contained primary data and was not a review or meta-analysis; (2) the publication reported data measured in the field (i.e., results were not taken from an online database); (3) the publication reported NEE, or GPP and TER which could be used to calculate NEE (Equation 2.1); (4) the publication reported the components necessary to calculate NEP at the field scale (Equation 2.2) on an annual basis (i.e., measurements were taken over a 365-day period) so that comparisons could be made across sites. If a publication measured data over multiple years, each measurement year was recorded separately; (5) the publication reported a value > 0 for C_H. This is necessary as, by definition, there will always be C_H from a cropland or managed grassland as harvested produce or grazed biomass. Studies that reported crop yield and not C_H were excluded, as crop yield alone does not provide an indication of C_H, as it may not consider all components of the aboveground biomass removed from the field. A cropland site and cut grassland site could be included if it reported C_H (> 0) and no C_I (i.e., no organic amendments are added), however grazed grassland sites had to report both C_H and C_I (> 0) to be included as there would be an import of C via livestock excreta; (6) the publication used EC to measure annual NEE or GPP and TER (i.e., not chambers or the flux gradient method);

(7) the study site is either a cropland growing a food or fodder crop or a managed grassland (i.e., cut for fodder, grazed or both cut and grazed); (8) the publication includes information on soil texture or reports the sand, silt and clay content so that soil type could be calculated (Table 2.2); (9) the publication specifies the crop or vegetation type grown during the measurement period; (10) the publication presents annual NEE (or GPP and TER), C_H and C_I (if applicable) in a numeric format; (11) the publication is written in or has been translated into English.

Occasionally, identical measurements were reported across multiple publications and so only one measurement per study site per year was recorded to avoid duplication.

TABLE 2.1 OVERVIEW OF SEARCH TERMS USED TO COLLATE PUBLICATIONS.

Search term	Number of results
TS=(Eddy covariance) AND TS= (net ecosystem exchange) AND TS= (agricultur* OR crop* OR grass* OR pasture)	573
TS=(Eddy covariance) AND TS= (net ecosystem carbon balance) AND TS= (agricultur* OR crop* OR grass* OR pasture)	52
TS=(Eddy covariance) AND TS= (net ecosystem productivity) AND TS= (agricultur* OR crop* OR grass* OR pasture)	94

2.2.2 Data extraction

The screening activity identified 40 publications from which relevant data were extracted to compile a database of 242 annual NEP measurements and associated meta-data (Tables A1.1 and A1.2). Data were digitised manually from tables or from within the text.

Where Köppen climate classification was not reported, this information was extracted from mindat.org (Hudson Institute of Meteorology, 2022) based on the latitude and longitude of the study location. Where soil texture was not reported it was estimated using the sand, silt and clay percentages provided within the publication using a soil texture calculator (United States Department of Agriculture, 2022). Each observation was then given a corresponding soil classification based on its textural class according to Hill et al. (2018) (Table 2.2). Irrigation management was not included as a management practice within the meta-analysis, as irrigation management was only acknowledged by 10 of the 40 papers and the irrigation amount reported by 8 of these 10. A requirement for irrigation management data would therefore have significantly limited the size of the dataset. Soil organic C content was not included as a potential driver of annual NEP as it was reported by only 12 of the 40 papers and thus would have significantly limited the size of the dataset had it been a requirement. Furthermore, very few papers reported grazing intensity or the number of cuts for the managed grasslands (N=9 and N=7 respectively). These variables were therefore not included as potential drivers of annual NEP, as the small sample sizes for each group would be insufficient for robust analysis.

TABLE 2.2 SOIL CLASSIFICATION AS DESCRIBED BY HILL ET AL. (2018).

Soil texture or type	Soil classification
Loam, loamy sand, sandy, sandy loam, silt, silt loam	Light
Clay loam, sandy clay loam, silty clay, silty clay loam	Medium
Clay, sandy clay	Heavy

Where annual NEE was not explicitly reported in the publication, but annual GPP and TER were, it was calculated as follows:

$$NEE = TER - GPP \qquad \text{(Equation 2.1)}$$

The micrometeorological sign convention is used for annual NEE; a positive NEE indicates that CO_2 is lost from the agroecosystem to the atmosphere, and a negative NEE indicates a net uptake of CO_2 from the atmosphere by the agroecosystem (Baldocchi, 2003).

Annual NEP was calculated as follows (adapted from Evans et al., 2021):

$$NEP = NEE + C_H - C_I \qquad \text{(Equation 2.2)}$$

As in Evans et al. (2021), we use the micrometeorological sign convention for annual NEP, where a positive NEP indicates the agroecosystem is losing C and a negative NEP indicates the agroecosystem is accumulating C.

For each annual NEP measurement, information on the climate, soil type and agricultural management practices used during the measurement period were recorded into categories and groups to understand their effects on annual NEP (Table 2.3). For analysis purposes, the amount of nitrogen (N) fertiliser added was converted from a continuous to a categorical variable with categories increasing in 100 kg N ha^{-1} yr^{-1} increments. Where applicable, data were converted into standardised units to enable comparison between studies (i.e., components of annual NEP converted to g C m^{-2} and N fertiliser rate to kg N ha^{-1} yr^{-1}). For data classified as cropland, the crop type (i.e., annual or perennial) was assigned based on the crop grown during the measurement period – if the crop lived for only one growing season it was classified as annual, however if the crop was able to regrow it was classified as perennial (Figure A1.1).

TABLE 2.3 CATEGORIES AND GROUPS USED TO CLASSIFY DATA.

Data	**Category**	**Groups**
Croplands and managed grasslands	Agricultural land use	Cropland
		Managed grassland

	Köppen climate classification	Aw: Wet tropical savannah
		BSk: Cold semi-arid (steppe)
		BWk: Cold desert
		Cfa: Humid subtropical
		Cfb: Temperate oceanic
		Csb: Warm-summer Mediterranean
		Cwa: Monsoon-influenced humid subtropical
		Dfa: Hot-summer humid continental
		Dfb: Warm-summer humid continental
		Dfc: Subarctic
		Dwa: Monsoon-influenced hot-summer humid continental
	Amount of N fertiliser added (kg N ha^{-1} yr^{-1})	0
		1-100
		101-200
		201-300
		301-400
		>401
	Soil type	Light
		Medium
		Heavy
Croplands only	Inclusion of cover crops	Yes
		No
	Crop type	Annual
		Perennial
	Residues retained	Yes
		No

	Tillage	Conventional tillage
		Reduced tillage
		No till
Managed grasslands only	Management	Cut
		Grazed
		Cut + grazed

2.2.3 Data analysis

Data were analysed using The R Language and Environment for Statistical Computing V4.1.3 (R Core Team, 2021). To determine the effect of environmental and management factors (Table 2.3) on annual NEP, we conducted tests for statistically significant differences between the annual NEP of climate and soil type, and management groups. First, normality tests were conducted using the Shapiro-Wilk method. Tests for statistically significant differences between groups within categories were conducted using independent t-tests, Wilcoxon tests, Kruskal Wallis tests, Dunn's tests, one-way ANOVA or Tukey tests as appropriate, depending on the normality of the data and the number of groups being compared.

Mixed effects models were used to assess the variable importance of climate, soil and management practices on the annual NEP of croplands and managed grasslands. As the model requires complete cases of data, data where one or more of the variables of interest were not reported by the publication were removed. The size of the croplands dataset for analysis was N=75 and for managed grasslands was N=98. As the datasets contained some data that was collected from the same site over multiple years, the site and measurement year were included as random effects in the model. Environment and management variables were included as fixed effects in the model; for croplands the fixed effects were: Köppen climate classification, soil type, amount of N fertiliser added, inclusion of cover crops, residue retention and tillage method, and for managed grasslands the fixed effects: were Köppen climate classification, soil type, management method and amount of N fertiliser added. Crop type (i.e., annual or perennial) was not included as a fixed effect in the croplands model as data were from sites growing annual crops only once incomplete cases had been removed.

2.3 Results

2.3.1 Overview of the dataset

A total of 242 individual annual NEP measurements and corresponding meta-data were obtained from the 40 publications (Tables A1.1 and A1.2): N=141 for croplands and N=101 for managed grasslands. The measurements were from a total of 11 countries with the majority from the USA and Germany (Tables A1.1, A1.2 and A1.3); compared to temperate regions, tropical regions were underrepresented. Of the 40 publications: 5 measured the annual NEP of one field for one year; 12 measured the annual NEP of one field over multiple years; 6 measured the annual NEP of multiple fields over one year; and 17 measured the annual NEP of multiple fields over multiple years. Very few of the studies within the dataset were designed to specifically test the influence of environmental conditions or management practices on annual NEP. Annual NEP values ranged from 764.8 g C m^{-2} (highest C loss) for an annual cropland growing a cover crop, silage maize and winter wheat in Germany with a temperate oceanic climate (Cfb) and light soil (silt loam) receiving no organic amendments (Poyda et al., 2019) to -499 g C m^{-2} (highest C gain) for a cut grassland in Japan with a warm-summer humid continental climate (Dfb) and light soil (silt loam) receiving 770 g C m^{-2} of organic amendments (Hirata et al., 2013). The mean (± standard deviation) annual NEP across the dataset was 76.6 ± 211 g C m^{-2}. Graphical summaries of the annual NEP of croplands and managed grasslands grouped by Köppen climate classification, soil type and agricultural management are presented in Figures 2.1 and 2.2.

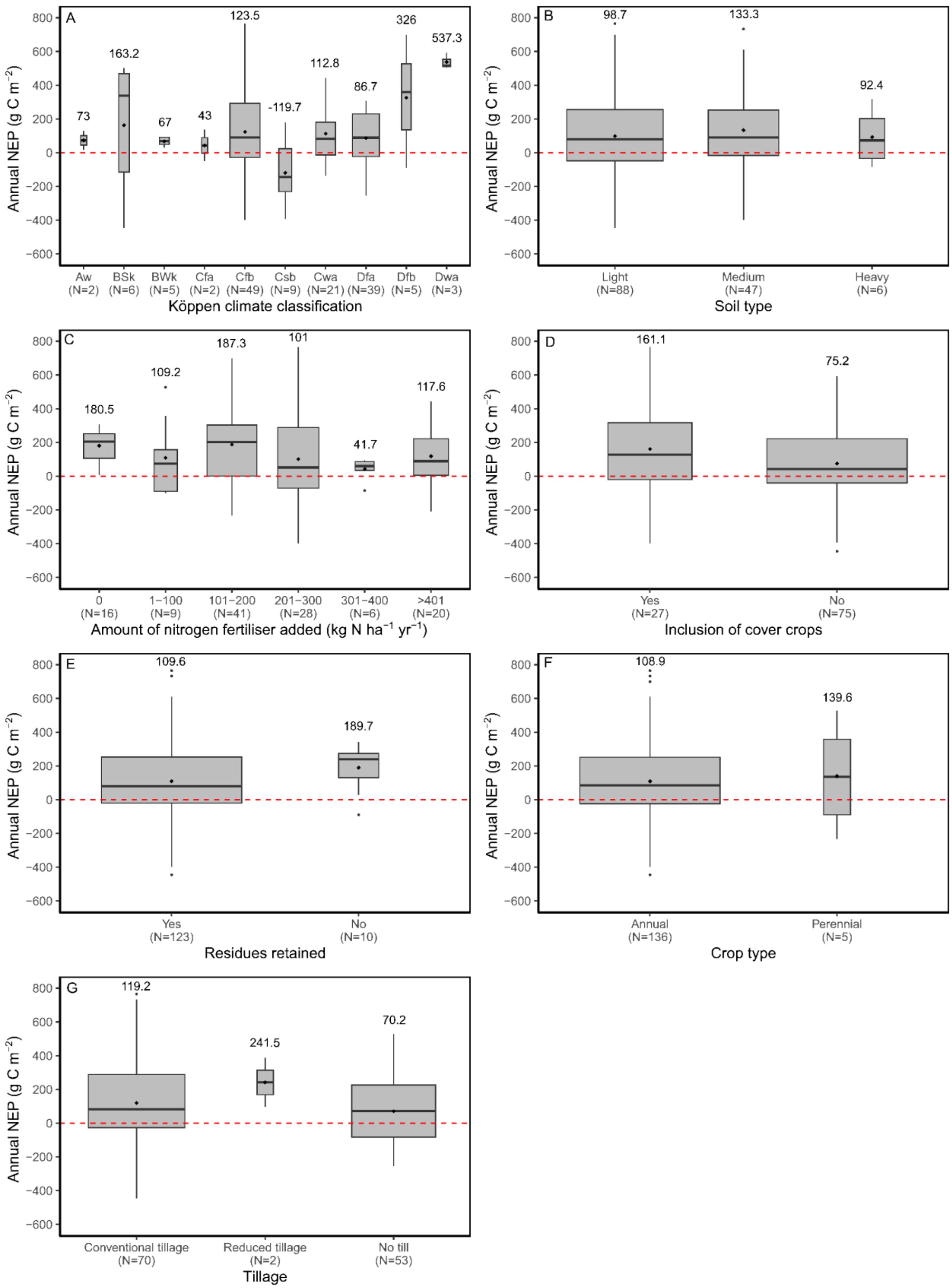
A
B
C
D
E
F
G
Annual NEP (g C m^{-2})
Köppen climate classification
Aw (N=2)
BSk (N=6)
BWk (N=5)
Cfa (N=2)
Cfb (N=49)
Csb (N=9)
Cwa (N=21)
Dfa (N=39)
Dfb (N=5)
Dwa (N=3)
73
163.2
67
43
123.5
-119.7
112.8
86.7
326
537.3
Soil type
Light (N=88)
Medium (N=47)
Heavy (N=6)
98.7
133.3
92.4
Amount of nitrogen fertiliser added (kg N ha^{-1} yr^{-1})
0 (N=16)
1−100 (N=9)
101−200 (N=41)
201−300 (N=28)
301−400 (N=6)
>401 (N=20)
180.5
109.2
187.3
101
41.7
117.6
Inclusion of cover crops
Yes (N=27)
No (N=75)
161.1
75.2
Residues retained
Yes (N=123)
No (N=10)
109.6
189.7
Crop type
Annual (N=136)
Perennial (N=5)
108.9
139.6
Tillage
Conventional tillage (N=70)
Reduced tillage (N=2)
No till (N=53)
119.2
241.5
70.2

Figure 2.1 Boxplots summarising the annual NEP database for croplands, displaying the range of annual NEP measurements grouped by: (A) Köppen climate classification, (B) soil type, (C) amount of N fertiliser added, (D) use of cover crops or not, (E) crop residue retention or removal, (F) crop type (i.e. annual or perennial) and (G) type of tillage. N= indicates the number of observations within each group, and C, D, E and G only display data from observations that reported information on that category. The width of each boxplot is proportional to the number of samples in each group; the diamond within each box and the value associated with each box represent the mean of the group. Positive values indicate C loss from and negative values indicate C accumulation within the agroecosystem. See Tables 2.4, 2.5, 2.6 and A1.1 for further information.

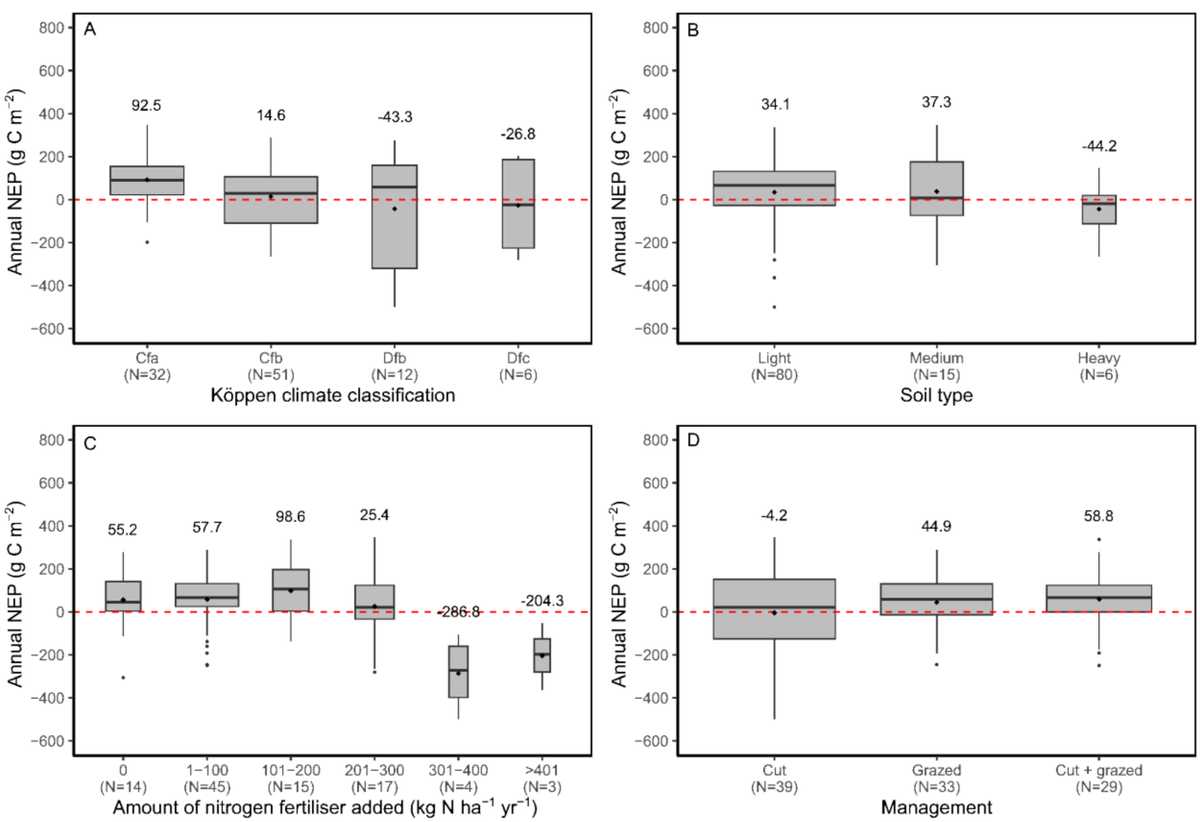

Figure 2.2 Boxplots displaying the range of annual NEP measurements grouped by: (A) Köppen climate classification, (B) soil type, (C) amount of N fertiliser added and (D) grassland management. N= indicates the number of observations within each group, and C only displays data from observations that reported information on that category. See Tables 2.4, 2.5, 2.7 and A1.2 for further information.

2.3.2 Annual NEP of arable croplands and managed grasslands

A t-test showed a significant difference between the mean annual NEP (± standard deviation) of croplands (110 ± 234 g C m^{-2}) and managed grasslands (29.9 ± 164 g C m^{-2}) (P = 0.02). The annual NEP of croplands had a greater range than of managed grasslands (Figure 2.3). For both land uses, there were more sites with a positive annual NEP than negative (i.e., most sites were losing C); there were a greater proportion of croplands with a positive annual NEP (69 %) than managed grasslands (65 %).

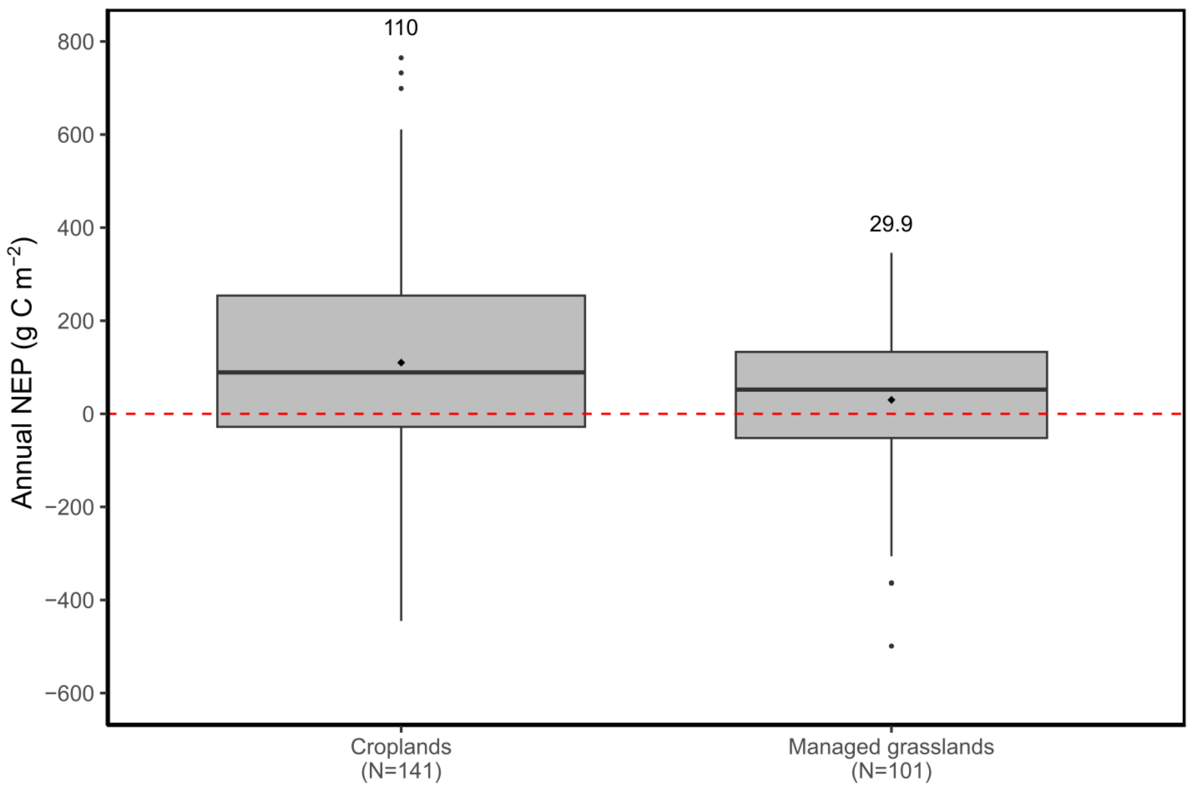

FIGURE 2.3 RANGE OF ANNUAL NEP MEASUREMENTS FOR CROPLANDS AND MANAGED GRASSLANDS. THE WIDTH OF EACH BOXPLOT IS PROPORTIONAL TO THE NUMBER OF SAMPLES IN EACH GROUP; THE DIAMOND WITHIN EACH BOX AND THE VALUE ASSOCIATED WITH EACH BOX REPRESENT THE MEAN OF THE GROUP.

A t-test showed the mean annual *in-situ* NEE (± standard deviation) of croplands (-252.9 ± 218 g C m^{-2}) was significantly more negative than that of managed grasslands (-184.6 ± 159 g C m^{-2}) ($P = 0.005$); more atmospheric CO_2 was being taken up by croplands than managed grasslands during periods of active growth. A Wilcoxon test showed that the mean C_I (± standard deviation) was significantly lower, by around 10 times, for croplands (15.2 ± 54 g C m^{-2}) than for managed grasslands (161.1 ± 185 g C m^{-2}) ($P = <0.001$). The mean C_H (± standard deviation) from croplands (378.1 ± 203 g C m^{-2}) was similar to that from managed grasslands (375.6 ± 175 g C m^{-2}) ($P = 0.73$). The mean C_H from croplands was considerably greater than the mean annual CO_2 being assimilated as NEE and C_I via organic amendments, so mean annual NEP was positive and there was overall C loss. The mean C_H from managed grasslands, however, was similar to the mean CO_2 that was assimilated as NEE and the mean C_I via organic amendments and excreta from grazing livestock, so NEP was close to neutral.

2.3.3 Environmental drivers of annual NEP

2.3.3.1 Climate

The majority of annual NEP measurements in our dataset (95 %) were from temperate and continental climate zones. Standard deviation of mean annual NEP was high for most climatic zones, ranging from 28 to 407 g C m^{-2} (Table 2.4), as the sample size of each Köppen climate zone was highly variable. For croplands, the mean annual NEP (± standard deviation) of sites with a warm-summer Mediterranean (Csb) climate (-119.7 ± 177 g C m^{-2}) was significantly lower than that of sites with a warm-summer humid continental (Dfb) climate (326 ± 312 g C m^{-2}) ($P = 0.01$) and a Monsoon-influenced hot-summer humid continental (Dwa) climate (537.3 ± 48 g C m^{-2}) ($P = 0.0007$); and the mean annual NEP (± standard deviation) of sites with a hot-summer humid continental (Dfa) climate (86.7 ± 158 g C m^{-2}) was significantly lower than that of sites with a Monsoon-influenced hot-summer humid continental (Dwa) climate (537.3 ± 48 g C m^{-2}) ($P = 0.03$). Köppen climate classification was identified by the mixed effects model as the only variable significantly influencing the annual NEP of croplands (Figure A1.2). The Csb climate zone was the only group with a negative mean annual NEP, as 67 % of these sites were accumulating C; all other climate zones had a greater proportion of sites with a positive mean annual NEP than negative, indicating that most of these sites lost C. The managed grasslands sites covered fewer Köppen climate zones than the croplands (Table 2.4). There were no statistically significant differences between the mean annual NEP of any of the Köppen climate zones (P = 0.15), and the mixed effects model showed that Köppen climate classification had no significant effect on the NEP of managed grasslands (Figure A1.2). Mean annual NEP was positive for sites in temperate climates (Cfa and Cfb) and negative for sites in subtropical climates (Dfb and Dfc); there were a greater proportion of sites that lost C in temperate climates than subtropical climates.

TABLE 2.4 MEAN ANNUAL NEP ± STANDARD DEVIATION (SD), THE PROPORTION OF SITES WITH POSITIVE AND NEGATIVE ANNUAL NEP MEASUREMENTS, AND AN INDICATION OF SIGNIFICANT DIFFERENCES BETWEEN THE MEAN ANNUAL NEP OF KÖPPEN CLIMATE CLASSIFICATION GROUPS FOR THE CROPLANDS AND MANAGED GRASSLANDS DATA. N= INDICATES THE NUMBER OF OBSERVATIONS WITHIN EACH GROUP.

	Köppen climate classification	**N=**	**Mean annual NEP ± SD (g C m^{-2})**	**% positive observations**	**% negative observations**	**Significant difference**
Croplands	Aw: Wet tropical savanna	2	73 ± 78	100	0	Between groups (P = 0.002): Csb and Dfb (P = 0.01), Csb and Dwa (P = 0.0007), Dfa and Dwa (P = 0.03)
	BSk: Cold semi-arid	6	163.2 ± 407	67	33	
	BWk: Cold desert	5	67 ± 28	100	0	
	Cfa: Humid subtropical	2	43 ± 132	50	50	
	Cfb: Temperate oceanic	49	123.5 ± 265	65	35	
	Csb: Warm-summer Mediterranean	9	-119.7 ± 177	33	67	
	Cwa: Monsoon-influenced humid subtropical	21	112.8 ± 161	71	29	
	Dfa: Hot-summer humid continental	39	86.7 ± 158	72	28	
	Dfb: Warm-summer humid continental	5	326 ± 312	80	20	

	Dwa: Monsoon-influenced hot-summer humid continental	3	537.3 ± 48	100	0	
Managed grasslands	Cfa: Humid subtropical	32	92.5 ± 124	81	19	None (*P* = 0.15)
	Cfb: Temperate oceanic	51	14.6 ± 138	59	41	
	Dfb: Warm-summer humid continental	12	-43.3 ± 265	58	42	
	Dfc: Subarctic	6	-26.8 ± 230	50	50	

2.3.3.2 Soil

Most of the data (69 %) were from sites with light soil (i.e., well-drained, high sand content); sites with heavy soil (i.e., poorly drained, high clay content) were underrepresented in our dataset (Table 2.5). For most soil types, standard deviation of mean annual NEP was high, ranging from 143 to 238 g C m^{-2}. No significant differences were observed between the mean annual NEP of croplands ($P = 0.71$) or managed grasslands ($P = 0.32$) when grouped by soil type (Figure A1.2). For croplands, mean annual NEP was positive for all soil types and there were a greater proportion of sites with a positive annual NEP than negative. Mean annual NEP was negative for managed grassland sites with heavy soil and positive for managed grassland sites with light and medium soils; most managed grasslands with heavy soil accumulated a small amount of C, whereas those with light or medium soil lost a small amount of C. It should be noted that the considerable disparity in sample sizes of the soil types in our dataset is likely to be influencing the lack of significant difference observed.

TABLE 2.5 MEAN ANNUAL NEP ± STANDARD DEVIATION (SD), THE PROPORTION OF SITES WITH POSITIVE AND NEGATIVE ANNUAL NEP MEASUREMENTS, AND AN INDICATION OF SIGNIFICANT DIFFERENCES BETWEEN THE MEAN ANNUAL NEP OF SOIL TYPE GROUPS FOR THE CROPLANDS AND MANAGED GRASSLANDS DATA. SEE TABLE 2.2 FOR SOIL TYPE CLASSIFICATION. N= INDICATES THE NUMBER OF OBSERVATIONS WITHIN EACH GROUP.

	Soil type	N=	Mean annual NEP ± SD (g C m^{-2})	% positive observations	% negative observations	Significant difference
Croplands	Light	88	98.7 ± 237	68	32	None (*P* = 0.71)
	Medium	47	133.3 ± 238	70	30	
	Heavy	6	92.4 ± 159	67	33	
Managed grasslands	Light	80	34.1 ± 162	85	15	None (*P* = 0.32)
	Medium	15	37.3 ± 182	53	47	
	Heavy	6	-44.2 ± 143	33	67	

2.3.4 The influence of agricultural management practices on annual NEP

2.3.4.1 Croplands

Mean annual NEP (± standard deviation) was not significantly different between croplands as a result of the amount of N fertiliser added, the inclusion of cover crops, residue retention, crop type (i.e., annual or perennial) or tillage method (*P* = >0.05) (Table 2.6). None of these variables had a significant influence on annual NEP (Figure A1.2). All management practices had a greater proportion of sites with a positive mean annual NEP than negative; standard deviation of mean annual NEP was high, ranging from 67 to 312 g C m^{-2} (Table 2.6).

TABLE 2.6 MEAN ANNUAL NEP ± STANDARD DEVIATION (SD), THE PROPORTION OF SITES WITH POSITIVE AND NEGATIVE ANNUAL NEP MEASUREMENTS, AND AN INDICATION OF SIGNIFICANT DIFFERENCES BETWEEN THE MEAN ANNUAL NEP OF MANAGEMENT PRACTICES FOR THE CROPLANDS DATA. N= INDICATES THE NUMBER OF OBSERVATIONS WITHIN EACH GROUP.

		N=	Mean annual NEP ± SD (g C m^{-2})	% positive observations	% negative observations	Significant difference
	0	16	180.5 ± 95	100	0	

Amount of N fertiliser added (kg ha^{-1} yr^{-1})	1-100	9	109.2 ± 217	67	33	None (P = 0.45)
	101-200	41	187.3 ± 238	76	24	
	201-300	28	101 ± 292	54	46	
	301-400	6	41.7 ± 67	83	17	
	>401	20	117.6 ± 178	75	25	
	Unknown	21				
Inclusion of cover crops	Yes	27	161.1 ± 295	67	33	None (P = 0.17)
	No	75	75.2 ± 213	68	32	
	Unknown	39				
Residues retained	Yes	123	109.6 ± 233	69	31	None (P = 0.27)
	No	10	189.7 ± 135	90	10	
	Unknown	8				
Crop type	Annual	136	108.9 ± 232	69	31	None (P = 0.77)
	Perennial	5	139.6 ± 312	60	40	
Tillage	Conventional tillage	70	119.2 ± 267	66	34	None (P = 0.37)
	Reduced tillage	2	241.5 ± 204	100	0	
	No till	53	70.2 ± 188	66	34	
	Unknown	16				

2.3.4.2 Managed grasslands

Significant differences in mean annual NEP were observed between managed grasslands as a result of the amount of N fertiliser added (P = <0.05) but not as a result of the grassland management practice used (P = 0.5) (Table 2.7). Mean annual NEP was significantly higher from sites fertilised with 1-100 kg N ha^{-1} yr^{-1} (57.7 ± 119 g C m^{-2}) and 101-200 kg N ha^{-1} yr^{-1} (98.6 ± 148 g C m^{-2}) than with 301-400 kg N ha^{-1} yr^{-1} (-286.8 ± 179 g C m^{-2}) (P = 0.04 and P = 0.02 respectively). The amount of N fertiliser applied had the greatest (and only significant) influence on the annual NEP of managed grasslands (Figure A1.2). Mean annual NEP was positive for most of the management practices – excluding those fertilised with 301-400 and >401 kg N ha^{-1} yr^{-1}, and those that were cut. Standard deviation of mean annual NEP was high across all groups, ranging from 119 to 204 g C m^{-2}.

Table 2.7 Mean annual NEP ± standard deviation (SD), the proportion of sites with positive and negative annual NEP measurements, and an indication of significant differences between the mean annual NEP of management practices for the managed grasslands data. N= indicates the number of observations within each group.

		N=	**Mean annual NEP ± SD (g C m^{-2})**	**% positive observations**	**% negative observations**	**Significant difference**
Amount of N fertiliser added (kg N ha^{-1} yr^{-1})	0	14	55.2 ± 149	71	29	Between groups (*P* = <0.006): 1-100 and 301-400 (*P* = 0.04), 101-200 and 301-400 (*P* = 0.02)
	1-100	45	57.7 ± 119	80	20	
	101-200	15	98.6 ± 148	73	27	
	201-300	17	25.4 ± 181	53	47	
	301-400	4	-286.8 ± 179	0	100	
	> 401	3	-204.3 ± 156	0	100	
	Unknown	3				
Management	Cut	39	-4.2 ± 204	54	46	None (*P* = 0.5)
	Grazed	33	44.9 ± 123	73	27	
	Cut + grazed	29	58.8 ± 139	72	28	

2.4 Discussion

This study compiled data from 40 publications that measured land-atmosphere and lateral C fluxes to evaluate how environmental conditions and management practices control the annual NEP of agroecosystems; the dataset comprised a total of 242 individual annual NEP measurements and associated meta-data. The mean annual NEP (± standard deviation) of the dataset was slightly positive (76.6 ± 211 g C m^{-2}), although the standard deviation of the mean was high which reflects the large range of values. 67 % of the sites in the dataset had a positive annual NEP (69 % of cropland sites and 65 % of managed grassland sites), confirming that on average these agroecosystems lost C, as also found by Smith et al. (2007). The mean annual NEP (± standard deviation) of croplands (110 ± 234 g C m^{-2}) was significantly higher than that of managed grasslands (29.9 ± 164 g C m^{-2}); croplands lost over

3.5 times more C than managed grasslands. Our results are similar to those reported by Ceschia et al. (2010), who found that European crop sites lost, on average, 138 ± 239 g C m^{-2} $year^{-1}$ and that 70 % of sites within their dataset lost C. Based on this C loss, they predict that 2 % of SOC content is being lost from European croplands annually (Ceschia et al., 2010). Our results show that the implementation of best management practices made no statistical difference to the NEP of croplands and that the NEP of the managed grasslands was only significantly influenced by N fertiliser rate.

Mean annual NEE was negative for both agroecosystems, though the *in-situ* uptake of CO_2 was greater for croplands than managed grasslands. This was compensated for by the significantly greater mean annual C_I to managed grasslands, which was around ten times greater than that to croplands. The mean annual C_H was similar from and accounted for the largest proportion of mean annual NEP in both agroecosystems. For the croplands, the mean annual C_H was considerably greater than the C added to the system (via plant photosynthesis and organic amendments), meaning that, on average, croplands lost C. For managed grasslands, mean annual C_H was only slightly higher than C_I to the system (via plant photosynthesis, organic amendments and excreta), however, meaning that overall managed grasslands were near C-neutral and lost only a small amount of C.

Multiple studies have proposed that soil C loss is higher from croplands compared to managed grasslands, which tend to accumulate C or be C-neutral (Prescher et al., 2010; Altimir et al., 2016). Croplands typically experience greater soil disturbance via tillage and the inclusion of bare soil or fallow periods within annual crop rotations, both of which have been shown to increase CO_2 emissions (Ciais et al., 2010a; Oertel et al., 2016; Jansson et al., 2021) and NEP. We found that, on average, croplands did lose more C than managed grasslands, although this was not solely attributed to the influence of management practices on NEE, as suggested above, and instead was largely influenced by the amount of C_I. Furthermore, there is large potential for uncertainty when calculating C_H and C_I, which is larger than the uncertainty associated with NEE measurement by EC (Ceschia et al., 2010); this was likely to be a factor contributing to the large variation in our results.

2.4.1 Environmental drivers of NEP

2.4.1.1 Climate

Köppen climate classification was the only variable, of those considered, to have a significant influence on the mean annual NEP of croplands. Croplands with a warm-summer Mediterranean (Csb) climate accumulated three times as much C, on average, than those with a warm-summer humid continental (Dfb) climate, and five times as much as those with a monsoon-influenced hot-summer humid continental (Dwa) climate, both of which lost C. Contradictorily, managed grasslands with temperate climates (Cfa and Cfb), on average, lost C, while managed grasslands with subtropical climates (Dfb and Dfc) accumulated C, although the differences in mean annual NEP were not significant. Subtropical climates are usually warmer than temperate climates, and agroecosystems in warmer regions have been observed to have higher rates of microbial activity, SOM decomposition and TER, and subsequently higher NEE and NEP (Lopez-Garrido et al., 2014; Maia et al., 2019; Bandaru, 2022). Other studies have observed higher C loss from croplands and managed grasslands in warmer climates compared to those in colder climates (Waldo et al., 2016).

2.4.1.2 Soil

Soil type had no statistical influence on the mean annual NEP of the croplands or managed grasslands within our dataset. It is notable, however, that the proportion of managed grassland sites that accumulated C increased with increasing soil clay content; on average managed grasslands with light and medium soils lost C, whereas those with heavy soils accumulated C. Clay particles protect SOC from decomposition, and it has been observed that soils with a higher clay content have lower CO_2 emission compared to lighter soils (Beziat et al., 2009; Li et al., 2010; Mangalassery et al., 2015; Maia et al., 2019; Prout et al., 2022) which can increase NEP (i.e., reduce overall C loss). The majority of the sites in our dataset were on light soil, and so the lack of significant difference in mean annual NEP between the soil types can probably be explained by the small number of sites with heavy and medium soils. Because of this, it should be noted that robust conclusions cannot be made on the influence of soil type on annual NEP and should be addressed in future research.

2.4.2 The influence of management practices on annual NEP

The cropland sites in our dataset spanned a variety of crop types (see Crop Species in Table A1.1) and management practices, although due to the spatial disparity within the dataset were dominated by crops grown in Europe and North America. The managed grassland sites were dominated by multi-species mix, which predominantly consisted of ryegrass, and were either managed for cutting, grazing or both cutting and grazing.

None of the management practices considered – crop type (i.e., annual or perennial), residue management (i.e., retention or removal), the inclusion of cover crops, the amount of N fertiliser added or the tillage method – had a statistical influence on the annual NEP of croplands. For the managed grasslands, the amount of N fertiliser added had a statistically significant influence on mean annual NEP, however the grassland management method (i.e., cut, grazed or cut and grazed) did not.

Croplands. The mean annual NEP of the croplands was not significantly influenced by the type of tillage, crop type (i.e., annual or perennial), retention of crop residues, the inclusion of cover crops or the amount of N fertiliser added, suggesting that the adoption of other best management practices, such as increasing C_I, may have greater success in reducing C losses. Relative to conventional tillage, no till aims to reduce SOM decomposition and soil CO_2 losses by disturbing the soil structure less (Smith, 2004; Olson et al., 2005; Stavi and Lal, 2013). Numerically, our results evidence this, as sites managed with conventional tillage lost more C than those managed with no till, although the difference was not significant. Tillage practices and crop residue management are often interlinked, with no till and crop residue retention often promoted in conservation agriculture to improve soil health (Farhate et al., 2018). Crop residues that are left on the field can be incorporated into the soil with tillage or left on the soil surface if no till is adopted (Fernandez et al., 2015) and can improve soil quality by reducing erosion and providing an input of organic C (Oertel et al., 2016; Nunes et al., 2020). There is a large consensus across the literature, however, that retaining crop residues, regardless of the tillage method used, can increase CO_2 emissions (Brye et al., 2006; Sainju et al., 2010): combining crop residue retention with conventional tillage can oxidise older SOC and release it as CO_2 (Ussiri and Lal, 2009; Ruan and Robertson, 2013; Wegner et al., 2018), whereas retaining residues and using no till leaves biomass to decompose on the soil surface, where it becomes more available to microorganisms for use as a substrate for priming and is then released as CO_2 (Mangalassery et al., 2015; Wegner et al., 2018). Our results corroborate this; the croplands sites in our dataset tended to lose C, and the amount of C lost was not significantly different between sites with residues retained and residues removed. The crop type (i.e., annual or perennial) also had no statistical influence on the variability of annual NEP. Sites growing annual crops often have higher C loss than those growing perennial crops (Amiro et al., 2017; Sarauer and Coleman, 2018), as perennial crops have longer growing seasons and extensive root systems which add slowly-decaying C into the soil and increase SOC (Smith, 2004; Ostle et al., 2009; Pausch and Kuzyakov, 2017). Furthermore, annual cropping systems are associated with more frequent tillage, as the soil is often ploughed after harvest which reduces the C sequestration potential (Flynn et al., 2012; Ledo et al., 2020). Our results do not corroborate this, however, although this may be due to the large disparity in sample sizes between the annual and perennial sites in our dataset. To improve the understanding of the influence of crop type on annual NEP, further investigation should consider crop type more specifically (i.e., by species (see Crop Species in Table A1.1) or rotation). The literature evaluating the impact of cover crops on C fluxes is conflicting. Cover crops can decrease annual NEP by providing an addition of C to offset some of the C lost at harvest, and can reduce soil erosion and thus CO_2 emission (Abdalla et al., 2013; Cates and Jackson, 2019). Alternatively, some studies observe higher CO_2 emissions from soils with cover crops compared to bare soils (Sanz-Cobena et al., 2014). Cover crop biomass is often left on the soil surface after termination, which is likely to have a similar effect on annual NEP as crop residue retention, increasing C losses as a result of priming (Wegner et al., 2018). The average annual NEP of sites with cover crops shows that these sites lost over twice as much C as those without cover crops, which supports the findings of Abdalla et al. (2013) and Cates and Jackson (2019), although the difference was not significant.

Managed grasslands. Managed grasslands that received over 301 kg N ha^{-1} yr^{-1} gained C on average, whereas those that received less than this lost C. Our findings contradict those of de la Motte et al. (2016) who found lower C losses from a managed grassland in years when less N fertiliser was added, but corroborate those of Hirata et al. (2013) who found C uptake increased with N fertilisation rate. In addition, managed grasslands fertilised with 0 kg N ha^{-1} yr^{-1} had just over twice the C loss of those fertilised with 201-300 kg N ha^{-1} yr^{-1}, showing greater C loss with lower N fertilisation and corroborating the findings of Hirata et al. (2013). A supply of N is required for C sequestration in agroecosystems (Flechard et al., 2005; Soussana et al., 2007; Moinet et al., 2017; Dmuchowski et al., 2022), and N fertilization can enhance C sequestration by increasing the retention of new C stocks (Das et al., 2024). High rates of N fertilisation could increase vegetation growth and photosynthesis, increasing annual CO_2 uptake and lowering annual NEP, as found by Liu et al. (2019), but could also result in increased C_H via biomass removal. There are negative impacts associated with applying N at high rates, however, including leaching and ammonia volatilization (Qin et al., 2012) and so N addition must be carefully matched to crop requirements to avoid this.

When comparing the impact of how grasslands were managed (i.e., cut, grazed, or cut and grazed), cut and grazed grasslands had the highest C losses, followed by grazed grasslands, and cut grasslands had a small uptake of C; all were close to C neutral and not significantly different from one another, however. Rutledge et al. (2015) and Carswell et al. (2019) propose that C_H is usually higher from managed grasslands that involve cutting compared to grazing, although C_I may be higher when livestock are present as excreta will be returned to the soil in addition to any organic fertiliser. Concomitantly, the presence of livestock within the EC footprint is likely to increase NEE as the CO_2 respired by grazing animals will be measured by the flux tower (Senapati et al., 2014). These factors may partially explain the numerically higher mean annual NEP from cut and grazed grasslands. Carbon fluxes from managed grasslands are also highly likely to vary as a result of management intensity (Zeeman et al., 2010) – i.e., stocking density and harvest frequency – however these management practices were only reported by a small number of the managed grasslands studies in our dataset and thus not considered as variables affecting NEP in our statistical model. To further understand the controls on the NEP of managed grasslands, our dataset would therefore benefit from sufficient information on the grazing intensity, grazing species, number of cuts, yield and the amount of C removed with each cut.

It is important to consider the challenges and potential error introduced when calculating C_H as grazed biomass and C_I as livestock excreta for agriculturally managed grasslands that include grazing livestock. Multiple methods were used to calculate these values across the grassland publications used in this analysis. The C_H via grazing was calculated by multiplying the C content of the grass by either the difference in height of a measured area of grass before and after grazing (Skinner, 2008; de la Motte et al., 2016; Skinner, 2013; Laubach et al., 2019; 2023) or by a standardised pasture utilisation value of 0.85 (Rutledge et al., 2017; Wall et al., 2019; 2020a; 2023b). All publications containing grazed grasslands considered the C_I as excreta as a proportion of the C ingested via grazing, however the proportion itself is variable: Skinner (2008; 2013) assumes 37 % of ingested C to be returned as dung; Rutledge et al. (2015) assume this to be 34 %; and other studies use a more comprehensive calculation which includes the non-digestible fraction of the grazed biomass and the amount of time livestock spend on the paddock (de la Motte et al., 2016; Rutledge et al., 2017; Laubach et al., 2019; 2023; Wall et al., 2019; 2020a; 2023b).

2.5 Recommendations for future research and policy

Our results show that, on average, global agroecosystems are behaving as C sources despite the implementation of best management practices which are encouraged as methods to increase soil C sequestration. On average, the croplands in our dataset lost C, whereas the managed grasslands were close to C neutral. However, over 65 % of all sites in both categories had positive NEP values.

Our dataset is limited both spatially and temporally, as NEE, and C_H and C_I for the calculation of NEP, are not reported consistently across the literature. To provide a more comprehensive and robust understanding of the controls on the annual NEP of agroecosystems we propose the following recommendations for future research: (i) more measurements from sites in different climates, with different soil types and management practices; (ii) standardised reporting of NEE, C_H and C_I for the calculation of NEP, taking measurements on an annual timescale, and reporting sufficient meta-data to make more direct comparisons between sites. These meta-data should include but not be limited to: mean air temperature and total precipitation during the study period, soil texture, SOC content and stock, grassland management, crop or vegetation type (including for managed grasslands), vegetation yield, N fertiliser rate, amount and type of C_I, amount of C_H (i.e., in grain yield and harvested residue), tillage management, grazing species, grazing duration, grazing intensity, the weight of harvested residues, whether cover crops were grown, number of harvests, and any management (i.e., tillage or fertilisation) occurring during the non-growing season; (iii) use before-after control-impact (BACI) type paired studies, such as in Zenone et al. (2013) and Skinner (2013), to provide more direct evidence of how altering management practices could influence NEP (i.e., conventional versus no till, cover crops versus no cover crops, residue retention versus residue removal); (iv) measure SOC at sites where EC is used to measure NEE to directly compare the impacts of management and land use practices and the relationship between NEE and SOC. This would require longer measurement periods – i.e., 5 to 10 years – to identify changes to SOC; (v) measure NEP over an entire crop rotation, as also suggested by Ceschia et al. (2010), as C_I may not occur in every year; (vi) to reduce uncertainties in the global GHG balance of croplands, systematically measure other GHG fluxes (i.e., N_2O) at the plot scale to update emission factors for a range of field operations (Ceschia et al., 2010; Osborne et al., 2010; Smith et al., 2010); (vii) introduce one standardised method to determine the amount of grass ingested by grazing livestock and the C returned to the soil via excreta. Furthermore, there are potential relationships between climate and land use type – due to the changing climate arable and grassland sites are now found in multiple climate types, and the climate conditions are likely to have an impact on the way that this agricultural land is managed. Due to the lack of data observed, this was not possible in this study, however future analysis would benefit from exploring these relationships once sufficient data is available to do so.

The agricultural sector would benefit from more targeted policy recommendations as to which agricultural practices will reduce soil C loss; our results show that using no till and growing cover crops do not always necessarily result in soil C gain, and so their effectiveness may be dependent on the environment in which they are grown. Guidance on the combinations of climate, soil type and management practices that are more likely to increase soil C sequestration would help farmers take more targeted action, although much of the ability to do this is dependent on evidence from research that uses the recommendations proposed above. Furthermore, greater communication on the importance of adding organic amendments to agricultural soils to provide an input of C would be beneficial (Bruni et al., 2022), as is currently being done in the UK Sustainable Farming Incentive and the international 4 per 1000 Initiative.

CRediT author statement

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Chapter 3 Maize grown for bioenergy on peat emits twice as much carbon as when grown on mineral soil

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Abstract

The area of land dedicated to growing maize for bioenergy in the UK is rapidly expanding. To understand how maize production influences soil carbon (C) dynamics, and whether this is influenced by soil type, we measured net ecosystem exchange (NEE) using the eddy covariance technique over the 2021 growing season. We combined the NEE data with C imports and exports to calculate the net ecosystem productivity (NEP) of two maize crops grown for bioenergy in the UK, one site on mineral soil and the other on lowland agricultural peat. Maize was similarly productive at both sites – gross primary productivity (GPP) was 1107 g C m^{-2} at the site with mineral soil and 1407 g C m^{-2} at the peat site. However, total ecosystem respiration (TER) was considerably higher from the peat site (1198 g C m^{-2}) compared to the mineral soil site (678 g C m^{-2}). After accounting for the removal of C in harvested biomass, both sites were net C sources, but C losses were over two times greater from the peat site (NEP = 290 g C m^{-2}) than the mineral site (NEP = 136 g C m^{-2}). While annual crops may be needed to produce bioenergy in the short term, growing maize for bioenergy in the UK does not appear to be a viable option for C sequestration over the long term, as it leads to high C losses from agroecosystems, especially those on organic soils. Instead, growing perennial bioenergy crops on mineral soils with a low organic C content is a more appropriate option.

3.1 Introduction

Bioenergy has received attention as a renewable resource and potential climate change mitigation measure, both as an alternative to fossil fuels and a method of carbon (C) sequestration when combined with C capture and storage (Hanssen et al., 2020; Calvin et al., 2021; de Freitas et al., 2021). In the UK, bioenergy is a significant source of renewable energy, generating around 11 % of the country's total electricity supply in 2022 (DESNZ, 2024). Given the role of bioenergy in decarbonising the energy sector, and the UK's legally-binding commitment to reach net zero greenhouse gas (GHG) emissions by 2050 or earlier, the demand for biomass is expected to increase significantly (DESNZ, 2023b). There are a range of crops, both annual and perennial, that can be grown for bioenergy production (Pugesgaard et al., 2014). As of 2020, 121,000 ha of land, equivalent to 1.4 % of the agricultural land area, were used to grow biomass for energy in the UK (Booth and Wentworth, 2023). Biogas is produced by anaerobic digestion (AD), where organic material is decomposed by microorganisms in an oxygen-limited environment, producing methane (CH_4) for use as energy (Gould, 2015; Vasco-Correa et al., 2018), and via biomass combustion, where organic material is combusted to produce heat (Skoufogianni et al., 2019).. Although the C emitted via combustion during AD is balanced by the C fixed by plant photosynthesis, bioenergy cannot be described as completely C neutral because the carbon dioxide (CO_2) savings are likely to be offset by emissions of CO_2, CH_4 and nitrous oxide (N_2O) during crop growth, field management, biomass processing and transport (Crutzen et al., 2008; Don et al., 2011).

Much of the existing research has proposed that growing perennial crops for bioenergy, such as willow and *Miscanthus*, rather than annual crops like maize (*Zea mays L.*) and wheat, has fewer negative impacts on the environment as perennials have more permanent root systems and require less fertiliser input (Karp and Richter, 2011; Pugesgaard et al., 2014; Kantola et al., 2022). Globally, maize is one of the most grown bioenergy crops, as it is high-yielding and has a high biogas output when anaerobically digested (Herrmann, 2013; Bright Maize, 2022). Maize is also grown extensively for bioethanol production, particularly in Brazil and the USA (Skoufogianni et al., 2019). To increase the scale and reliability of biogas production, the amount of arable land dedicated to the production of bioenergy crops, including maize, is growing (Souza et al., 2015; Hill, 2016). In 2021, 75,000 ha of land was used to grow maize for bioenergy production in the UK (DEFRA, 2021c). In the UK, maize is usually harvested in October, meaning that the field is left bare over winter and is vulnerable to soil erosion, as there is insufficient time for a winter crop or cover crop to be sown and established (Naylor et al., 2022). In addition, whole-crop harvesting of maize for AD results in large-scale removal of crop residues that can deplete soil organic C (SOC) (Ceschia et al., 2010; Raffa et al., 2015; Poyda et al., 2019; Wall et al., 2020b). While most of the agricultural land in the UK is on mineral soil, around 1.1 % (194,000 ha) is on drained lowland peat, representing approximately 7 % of the UK's total peat area (Evans et al., 2017). Natural peatlands are a considerable C store; and so peat drainage, initiated at scale in the UK in the 1600s to facilitate agricultural expansion, increases soil aeration and thus decomposition, leading to soil C loss as CO_2 (Evans et al., 2016). Agricultural mineral soils are also sources of C following intensive management (Ussiri and Lal, 2009; Franzluebbers, 2021; Bhattacharyya et al., 2022), however to a lesser extent than drained lowland peatlands (Freeman et al., 2022).

The use of agricultural land to grow maize for bioenergy is ongoing in the UK and is expected to increase despite the debate within the field on how sustainable or environmentally friendly this is, particularly when these crops are grown on peat soils (Evans et al., 2024). The phase out of biomethane crops grown on peat in Europe has received little attention, unlike palm oil grown on tropical peats, where Jeswani et al. (2020) reported that palm oil may emit 3-40 times more GHG emissions than fossil diesel. Despite the likely continued increase in maize production for bioenergy in the UK, the existing research on GHG emissions from agricultural soils during the maize growing season, particularly on agricultural peat, is not comprehensive (Pohl et al., 2015). While there is an urgent need to move away from fossil fuels in the energy sector, it is important to improve our understanding of the C fluxes and potential environmental impacts associated with different components of the biomass supply chain and calculate GHG emissions related to biogas production from feedstock crops. Given the predominance of growing maize for bioenergy, it is important to determine the impacts of growing maize for bioenergy on agricultural emissions and how this varies because of the environment in which it is grown (Lohila et al., 2003). The aim of this study was to determine the impact of soil type on the CO_2 sink or source strength of growing maize for bioenergy. This was achieved by carrying out the following objectives: (i) quantifying the CO_2 fluxes associated with growing maize for bioenergy at two commercial farms using an eddy covariance (EC) tower at each, one on mineral soil and the other on peat; and (ii) estimating the C sink or source strength of these systems by calculating net ecosystem productivity (NEP). It has been shown that GHG emissions (i.e., CO_2, nitrous oxide (N_2O) and methane (CH_4)) are higher from crops grown on peat than on mineral soil (Oertel et al., 2016; Evans et al., 2021); thus, we hypothesise that the CO_2 balance will be more positive from the maize grown on peat than the maize grown on mineral soil.

3.2 Methods

3.2.1 Study sites

The two sites used in this study are both commercial farms in eastern England. One is located in Yorkshire on a loamy calcareous brown earth from the Aberford series of Calcaric Endoleptic Cambisols (Cranfield University, 2018), (subsequently referred to as the mineral soil site (MS)) and the other is located 250 km south in East Anglia on drained lowland peat (subsequently referred to as the peat soil site (PS)). Both sites have a temperate oceanic climate characterised by mild winters and warm summers (Beck et al., 2018). Between 1992 and 2021 average annual temperature was higher at PS (10.7 ± 0.5 ˚C, ranging from 9.5 ˚C to 11.7 ˚C) than at MS (9.5 ± 1 ˚C, ranging from 6 ˚C to 10.8 ˚C) (Met Office, 2019; Met Office, 2023), whereas average annual precipitation was higher at MS (639 ± 142 mm, ranging from 289 mm to 916 mm) than at PS (561 ± 95 mm, ranging from 309 mm to 699 mm) (Met Office, 2006; Met Office, 2023). During the measurement period (2021 maize growing season), average daily temperature and total precipitation were 15.5 ˚C and 230 mm at MS, and 15.6 ˚C and 249 mm at PS respectively (Figure 3.1); the similar air temperature and precipitation at the two study sites can be attributed to the north of England experiencing warmer and drier than average conditions through summer 2021, whereas the southeast was closer to average.

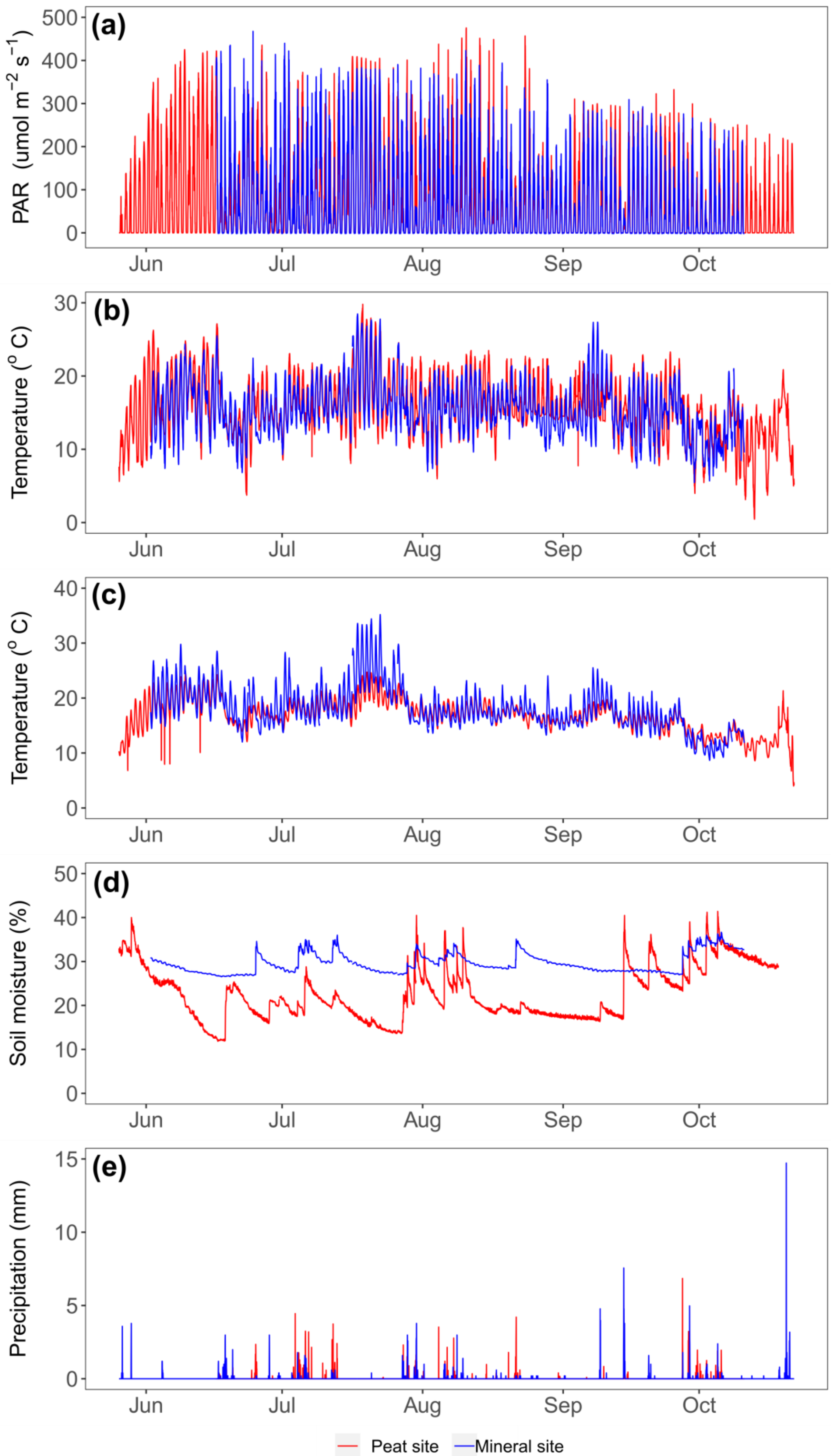
(a)
PAR (umol m^-2 s^-1)
500
400
300
200
100
0
Jun
Jul
Aug
Sep
Oct
(b)
Temperature (°C)
30
20
10
0
(c)
Temperature (°C)
40
30
20
10
0
(d)
Soil moisture (%)
50
40
30
20
10
0
(e)
Precipitation (mm)
15
10
5
0
Peat site
Mineral site

FIGURE 3.1 (A) PHOTOSYNTHETICALLY ACTIVE RADIATION (PAR), (B) AIR TEMPERATURE, (C) SOIL TEMPERATURE (5 CM), (D) SOIL MOISTURE (5 CM) AND (E) PRECIPITATION MEASURED OVER THE MAIZE GROWING SEASONS AT THE STUDY SITES.

The field at MS (10.4 ha) has been under continuous arable rotation with conventional tillage since 1994 with a rotation of winter wheat, spring or winter barley, and oilseed rape, and occasionally vining peas or potatoes. Prior to this, set aside and grass leys were included in the crop rotation. In September 2020, linseed was sown in the field, however the crop failed due to frost conditions and so was terminated and planted with maize in June 2021. The PS is highly fertile and nutrient rich. From the 1600s onwards, lowland peatlands across the UK were widely drained for use in agricultural crop production (Rowell, 1986) but since the advent of electric pumps in the 20th century the process has become more efficient, leading to deeper drainage. The field at PS (41.7 ha) was drained during the 1940s and since then has been cultivated for agriculture with the water table controlled by electric pumps. During the measurement period the average daily water table depth was -139 cm, ranging from -160 cm to -110 cm. Soil properties of the maize fields are summarised in Table 3.1; notably, organic matter content, total C, total organic C and total N are higher at PS than at MS.

TABLE 3.1 SOIL INFORMATION FOR EACH SITE (MEAN ± STANDARD DEVIATION, N=9, FOR TOPSOIL 0-30 CM).

	Mineral site (MS)	**Peat site (PS)**
Soil type [a]	Calcaric Endoleptic Cambisol	Histosol
Soil texture [b]	Clayey loam	Loamy peat over sand
Water table depth (m)	-	< 1
Organic matter (%)	6.7 ± 0.6	59.2 ± 2.2
pH ($CaCl_2$)	6.9 ± 0.2	7.3 ± 0.1
Bulk density (g cm^{-3})	1.3 ± 0.1	0.5 ± 0.1
Total carbon (g kg^{-1})	39.5 ± 9	278.6 ± 37.6
Total organic carbon (g kg^{-1})	22.9 ± 4.9	229.7 ± 9.1

Total nitrogen (g kg^{-1})	2.3 ± 0.6	16.4 ± 2.2
C:N ratio	10:1	14:1
Plant available nitrogen (g kg^{-1})	0.013 ± 0	0.085 ± 0.4

[a] Data obtained from World Reference Base for Soil Resources (IUSS, 2022); [b] Data obtained from UK Soil Observatory (UK Research and Innovation, 2021)

Detailed information on management practices at both sites during the study period are presented in Table 3.2. The planting density of maize was slightly higher at MS (110,000 seeds ha^{-1}) than at PS (95,000 seeds ha^{-1}), and nitrogen (N) fertilisation was simliar at the two sites (76 kg N ha^{-1} at PS and 72.5 kg N ha^{-1} at MS). At MS maize was planted on 02/06/2021 and harvested on 10/10/2021 (131 days) and at PS maize was planted on 27/04/2021 and harvested on 21/10/2021 (178 days). The farmer at MS opted for a high sowing density to maximize the potential for crop growth to compensate for the later planting date resulting from the failure of a previously sown autumn crop. Crop yield data for both sites were provided by the farmer; as quadrats were not used to measure yield, standard deviation of yield is therefore not reported.

TABLE 3.2 MANAGEMENT INFORMATION FOR EACH SITE OVER THE MAIZE GROWING SEASON (DM = DRY MATTER).

Mineral site (MS)		**Peat site (PS)**	
Date	**Management**	**Date**	**Management**
Spring 2021	Fertiliser (N26+5SO3): 50 kg N ha^{-1}, 9.6 kg S ha^{-1}	27/04/2021	Planted maize (Pioneer variety) using precision drill: 95,000 seeds ha^{-1}
16/04/2021	Herbicide (Amega Duo): 2.1 L ha^{-1} (with 0.5 L ha^{-1} Phase II and 0.5 L ha^{-1} Spryte Aqua)	30/04/2021	Fertiliser (CHAFER N30.3+10.8SO3): 76 kg N ha^{-1}, 10.8 kg S ha^{-1}
06/06/2021	Herbicide (Pendimethalin): 3.3 L ha^{-1}	02/06/2021	Pesticide (Maya): 1 L ha^{-1}

	Herbicide (Glyphosate): 2 L ha^{-1}		
18/05/2021 19/05/2021	Non-inversion tillage: 20-25 cm	10/06/2021 14/06/2021 29/06/2021	Fertilisers (Headland Copper 435, Headland Boron 150, Headland Zinc 150): 64 g copper ha^{-1}, 22.5 g boron ha^{-1}, 75 g zinc ha^{-1}
02/06/2021	Planted maize (Fieldstar variety) using precision drill: 110,000 seeds ha^{-1} Fertiliser (Di-ammonium phosphate): 22.5 kg N ha^{-1} and 57.5 kg P ha^{-1}	21/10/2021	Harvest: 11.3 t DM ha^{-1}
10/10/2021	Harvest: 12.3 t DM ha^{-1}		

3.2.2 Measurement of CO_2 fluxes

Turbulent fluxes of CO_2 (µmol m^{-2} s^{-1}) and sensible and latent heat fluxes (H, LE; W m^{-2}) were measured with EC flux towers (Moncrieff et al., 1997; Baldocchi, 2003). At MS, CO_2 fluxes were measured using an LI-7200 RS enclosed infrared CO_2/H_2O gas analyser (LI-COR Biosciences, USA); data were sampled at 10 Hz and combined with ancillary measurements by a CR1000X data logger (Campbell Scientific, USA) via a Smartflux 2 processing computer (LI-COR Biosciences, USA) and stored on a USB drive. At PS, CO_2 fluxes were measured with an LI7500A open path CO_2/H_2O gas analyser (LI-COR Biosciences, USA); data were logged at 20 Hz using a CR3000 data logger (Campbell Scientific, USA). At both sites a Gill Windmaster three-dimensional sonic anemometer (Gill Instruments Ltd., UK) was used to measure atmospheric turbulence (u, v, w; m s^{-1}) and sonic temperature (Tsonic; ˚C). Sensors were mounted on extendable masts, the height of which were increased over the maize growing season to ensure a minimum distance of 2 m between the EC sensors and crop canopy. At MS, the mean peak footprint distance was 40 m and had an average 90 % contribution of 110 m (Figure A2.1; Kljun et al., 2015). At PS, the mean peak footprint distance was 35 m and an average 90 % contribution of 97 m (Figure A2.2; Kljun et al., 2015). All measurements were taken during the 2021 maize growing season. The monitoring period at MS was 131 days (02/06/2021-10/10/2021) and at PS was 149 days (26/05/2021-21/10/2021); at PS, EC measurements are available from around one month after maize was planted due to instrument failure, and so this should be considered when interpreting results.

3.2.3 Calculation of CO_2 fluxes

EddyPro® 7 V7.0.6 (LI-COR Biosciences, 2019) was used to compute 30-minute fluxes of H, LE and net ecosystem exchange (NEE) from raw EC data. Net ecosystem exchange was calculated as the CO_2 flux plus the CO_2 storage term; as both towers had a height of below 10 m, the CO_2 storage term is likely to be negligible in comparison to the estimation of NEE (Nicolini et al., 2018). As Gill Windmaster sonic anemometers were used at both sites, the software applied the 'w-boost' bug correction (LI-COR Biosciences, 2024) and applied a double coordinate rotation to correct for any tilt or misalignment of the anemometer (Wilczak et al., 2001). Cross-correlation was used to compensate for any time lags between the sonic anemometer and atmospheric scalars (Moncrieff et al., 1997; 2004) and fluxes were corrected for air density fluctuations using the Webb-Pearman-Leuning correction (Webb et al., 1980).The software removed statistical outliers and implausible values in the raw timeseries according to Mauder et al. (2013). Fluxes were also corrected for high and low frequency co-spectral attenuation according to Moncrieff et al. (1997; 2004). Random uncertainty estimation due to sampling error was estimated according to Finkelstein and Sims (2001).

Quality control was applied using The R Language and Environment for Statistical Computing V4.1.3 (R Core Team, 2022) to ensure only high-quality flux data were used, following the workflow by Morrison et al. (2019). Examples of when data were removed include: statistical outliers (Papale et al., 2006); data obtained when the signal strength of the LI-COR was higher than the baseline value (Ruppert et al., 2006); data identified as non-representative by the footprint model (i.e., when > 20 % of the data was recorded outside of the site boundaries) (Kljun et al., 2004); data that was beyond realistic thresholds (i.e., when H < -200 or > 450 W m^{-2}, when LE < -50 or > 600 W m^{-2}, or when NEE < -60 or > 30 g m^{-2}), and when friction velocity (u*; m s^{-1}) < 0.06 at MS and < 0.08 at PS. The REddyProc package (Reichstein et al., 2016) was used to gap-fill and partition fluxes of NEE according to Reichstein et al. (2005). Periods of missing data (excluding the first month of the growing season at PS) were gap-filled using marginal distribution sampling and uncertainty was estimated as the standard deviation of the observations used to fill gaps (Reichstein et al., 2005; 2016). Gap-filled NEE accounted for 10 % and 36 % of the overall dataset at MS and PS respectively.

The micrometeorological sign convention is used for NEE, where a positive value indicates the ecosystem is losing C and a negative value indicates the ecosystem is accumulating C (Baldocchi, 2003). Net ecosystem exchange of CO_2 is the difference between gross primary productivity (GPP) and total ecosystem respiration (TER) as shown in Equation 3.1 (Smith et al., 2010). Following gap-filling, NEE was partitioned into GPP and TER (Reichstein et al., 2016).

$$NEE = TER - GPP \qquad \text{(Equation 3.1)}$$

3.2.4 Ancillary measurements

Additional micrometeorological measurements were recorded at both sites. Energy fluxes, including net radiation (Rnet), short-wave incoming radiation (SWin), short-wave outgoing radiation (SWout), long-wave incoming radiation (LWin) and long-wave outgoing radiation (LWout); W m^{-2}) were measured with an SN-500 net radiometer (Apogee Instruments, USA). Air temperature (Ta; ˚C) and relative humidity (RH; %) were measured with an HMP155 temperature and humidity probe (Vaisala BV, Finland). At MS, soil temperature (Tsoil; ˚C) and soil moisture (%) were measured using TEROS 11 temperature and moisture probes (METER Group Inc., USA) at a depth of 5 cm, soil heat flux (G; W m^{-2}) was measured using HFP01-SC heat flux plates (Hukesflux, Netherlands) at a depth of 5 cm, and precipitation (mm) was measured at a nearby COSMOS-UK weather station with an OTT Pluvio^2 rain gauge (OTT HydroMet, USA) (Cooper et al., 2021). At PS, G was measured using HFP01-L heat flux plates (Hukesflux, Netherlands in Campbell Scientific, USA), Ta and Tsoil were measured using TDT soil water content sensors (Acclima, USA) at a depth of 5, 10, 15 and 25 cm, while water level (cm) was measured with a CS451 pressure transducer (Campbell Scientific, USA), and precipitation was measured with an SBS500 tipping bucket rain gauge (Environmental Measurements Ltd.).

3.2.5 Energy balance

Energy balance closure (EBC) is a method used to assess the quality of EC data at a study site (Aubinet et al., 2001; Wilson et al., 2002). Energy balance closure assumes that the sum of fluxes measured by EC (LE + H) are equal to the available energy measured independently using other instruments (Rnet – G). The measured turbulent fluxes accounted for 76 % and 72 % of the available energy at MS and PS respectively (Figure 3.2). The R^2 values (i.e., amount of variance) are within the typical range of reported EC measurements (0.7-0.9) (Wilson et al., 2002; Foken, 2008; Wagle et al., 2018).

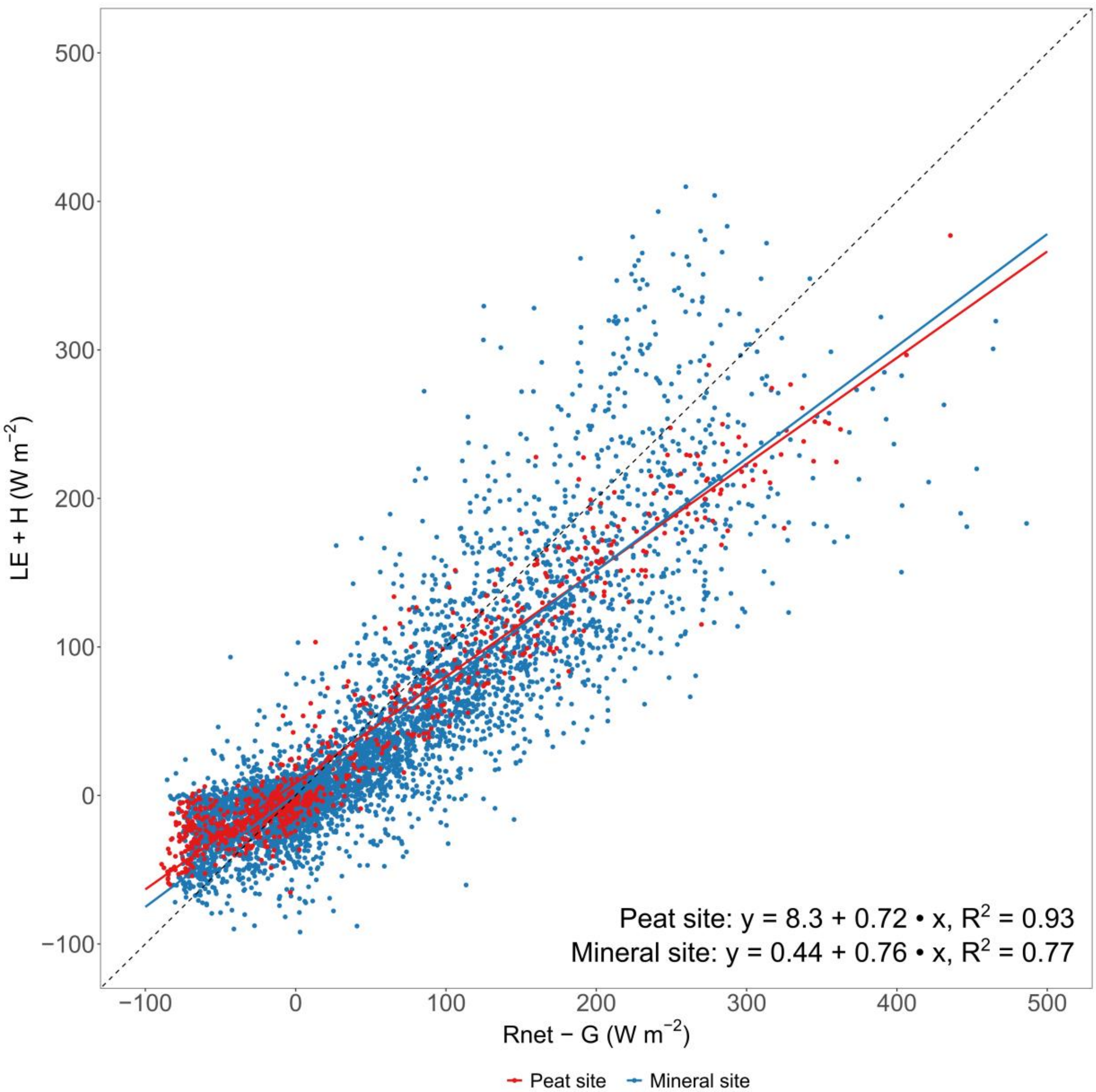

FIGURE 3.2 ENERGY BALANCE AT THE STUDY SITES OVER THE MAIZE GROWING SEASON WHERE H IS SENSIBLE HEAT FLUX, LE IS LATENT HEAT FLUX, RNET IS NET RADIATION AND G IS SOIL HEAT FLUX. NOTE THAT THE EBC DATA FOR PS IS FROM 04/08/2021-21/10/2021 DUE TO MISSING DATA PRIOR TO THIS DATE.

3.2.6 Net ecosystem productivity and crop carbon use efficiency

Net ecosystem productivity (NEP) is a measure of the C sink or source strength of an agroecosystem, and accounts for lateral fluxes of C, that is, C exported from the field via harvested biomass and C imported via seed or organic fertiliser (Equation 3.2 – adapted from Evans et al., 2021), as well as NEE. The C content of harvested biomass (C_H) was calculated by analysing the C content of maize samples taken from the field on the day of harvest, and scaling this to the reported yield for the field. As this study assesses NEP at the field scale, it is assumed that all C within the exported biomass was converted back to atmospheric CO_2 during AD (Eichelmann et al., 2016; Morrison et al., 2019). We note that this assumption requires further analysis; however, as the AD process involves storage and transformations of C across gaseous, liquid and solid phases, but a full life-cycle analysis is beyond the scope of the present study. Carbon import (C_I) was in the form of seed only, as neither site was fertilised with organic amendments prior to maize planting or during the growing season. As in Evans et al. (2021), we use the micrometeorological sign convention for NEP where a positive value indicates the ecosystem is losing C and a negative value indicates the ecosystem is accumulating C.

$$NEP = NEE + C_H - C_I \quad \text{(Equation 3.2)}$$

The C use efficiency of harvested material (CUE_h) is a measure of how efficiently atmospheric C is converted into new plant material (Chen et al., 2018); CUE_h is calculated as C_H over GPP (Kim et al., 2022) as in Equation 3.3.

$$CUE_h = \frac{C_H}{GPP} \quad \text{(Equation 3.3)}$$

3.3 Results

3.3.1 Carbon fluxes

Over the maize growing season, both sites exhibited *in situ* net CO_2 uptake as NEE, however the net CO_2 uptake at PS (-208 ± 49 g CO_2-C m^{-2}) was less than half of that at MS (-429 ± 57 g CO_2-C m^{-2}) (Figure 3.3; Figure 3.4; Table 3.3). Maximum CO_2 uptake was greatest at

MS during August and at PS during September (Figure 3.4). Both sites were similarly productive, with GPP 1107 ± 113 g C m^{-2} at MS and 1407 ± 129 g C m^{-2} at PS, however TER was nearly twice as high at PS (1198 ± 100 g C m^{-2}) than at MS (678 ± 62 g C m^{-2}) (Table 3.3). Total ecosystem respiration was notably higher during the night at PS compared to MS (Figure 3.4).

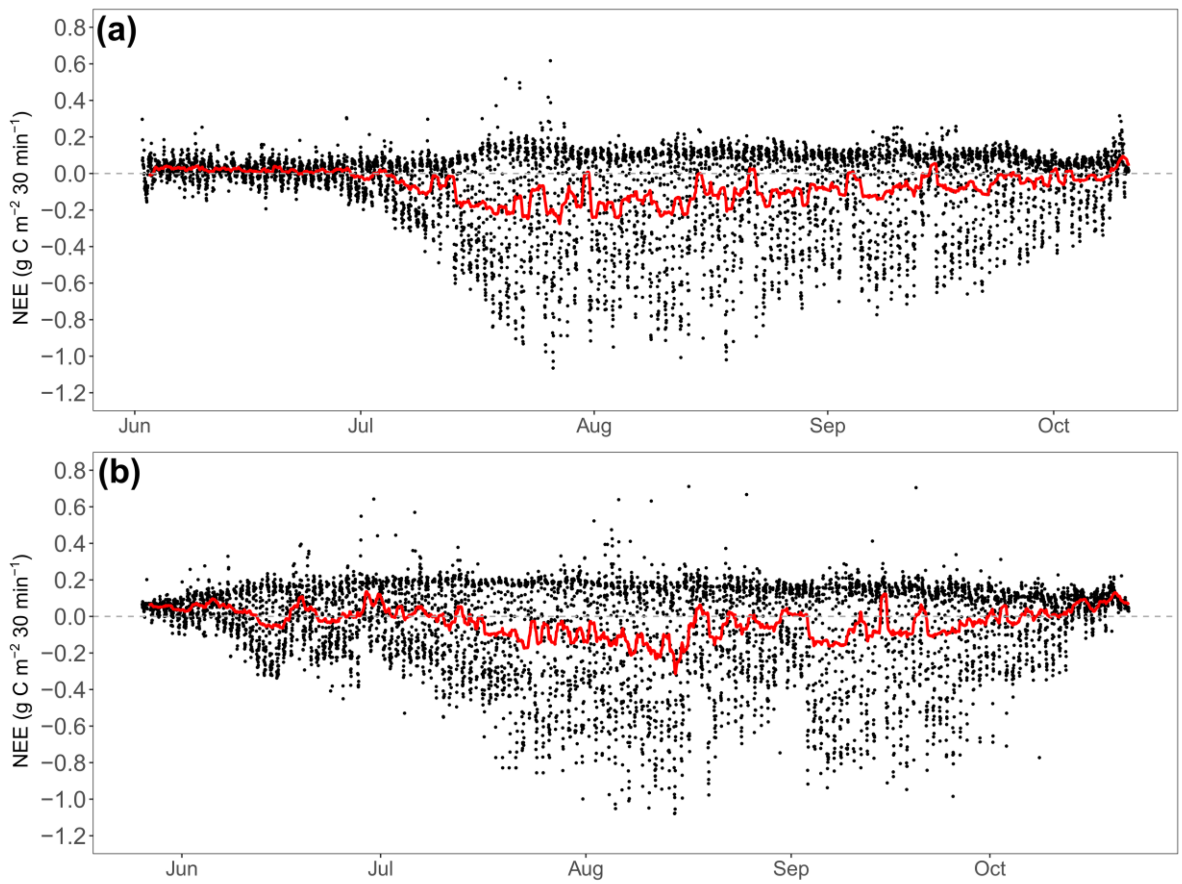

FIGURE 3.3 30-MINUTE FLUXES OF NEE AT (A) THE MINERAL SITE AND (B) PEAT SITE OVER THE MAIZE GROWING SEASONS. THE RED LINE INDICATES THE ROLLING DAILY MEAN.

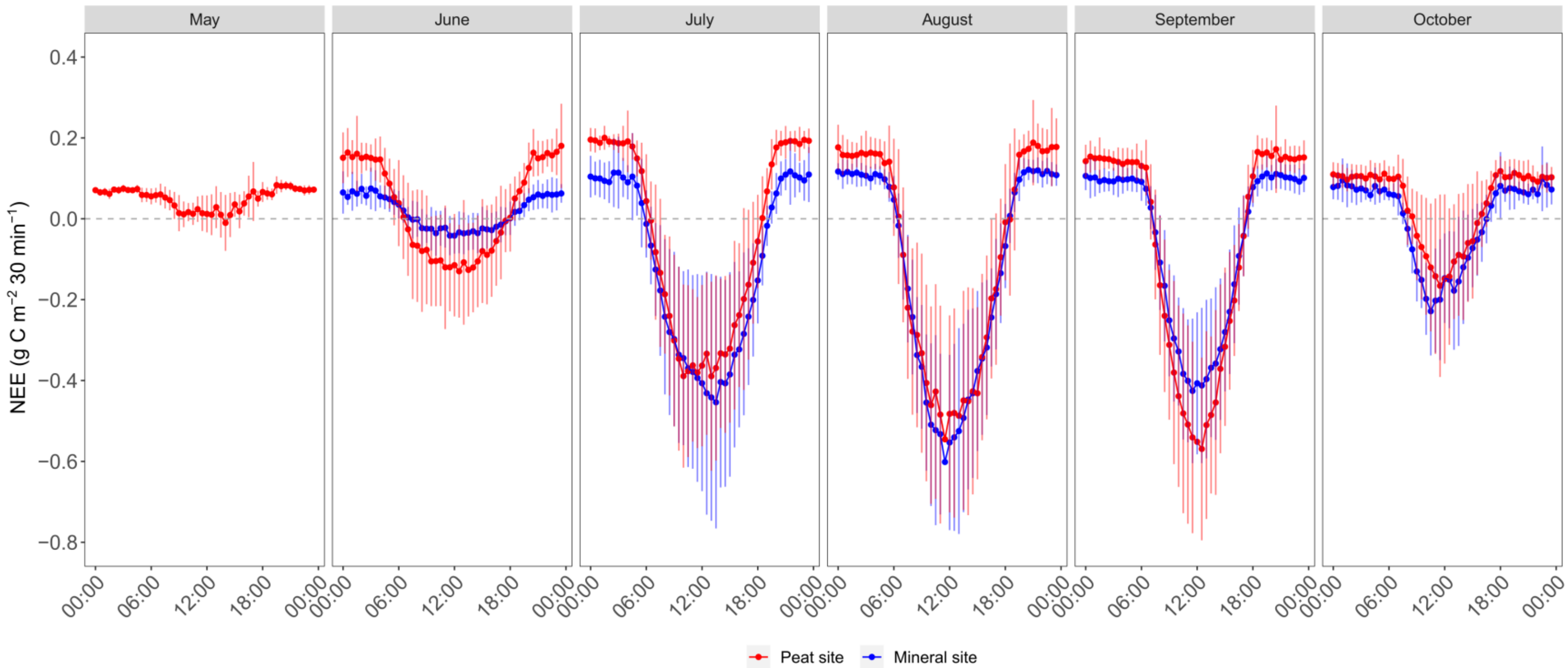

FIGURE 3.4 MEAN DIURNAL NEE AT THE STUDY SITES OVER THE MAIZE GROWING SEASON GROUPED BY MONTH. ERROR BARS REPRESENT STANDARD ERROR OF THE MEAN.

TABLE 3.3 CARBON BUDGET AT THE STUDY SITES ± ROOT SUM SQUARED (ASIDE FROM C_H WHERE ± REPRESENTS STANDARD DEVIATION). THE MICROMETEOROLOGICAL SIGN CONVENTION IS USED FOR NEE AND NEP WHERE POSITIVE VALUES INDICATE C LOSS AND NEGATIVE VALUES INDICATE C GAIN.

	Mineral site (MS)	**Peat site (PS)**
NEP (g C m^{-2})	136 ± 122	290 ± 99
NEP (t C ha^{-1})	1.4 ± 1.2	2.9 ± 1
NEE (g CO_2-C m^{-2})	-429 ± 57	-208 ± 49
GPP (g C m^{-2})	1107 ± 113	1407 ± 129
TER (g C m^{-2})	678 ± 62	1198 ± 100
Yield (t ha^{-1})	12.3	11.3
Maize C content (%)	46	44
CUE_h (g C g C^{-1})	0.51	0.35
C_H (g C m^{-2})	567 ± 65	499 ± 50
C_I (g C m^{-2})	2 ± 0	1 ± 0

3.3.2 Net ecosystem productivity

Cumulative NEP was positive at both sites, showing that C was being lost from both sites under maize cultivation, although C losses from PS (290 ± 99 g C m^{-2} growing season) were over twice those from MS (136 ± 122 g C m^{-2} growing season) (Table 3.3; Figure 3.5). The C_H at MS (567 ± 65 g C m^{-2}) was higher than that at PS (499 ± 50 g C m^{-2}), with yield also being slightly higher at MS, and C_I was minimal at both sites (2 ± 0 g C m^{-2} and MS and 1 ± 0 g C m^{-2} at PS), in the form of seed only (Table 3.3).

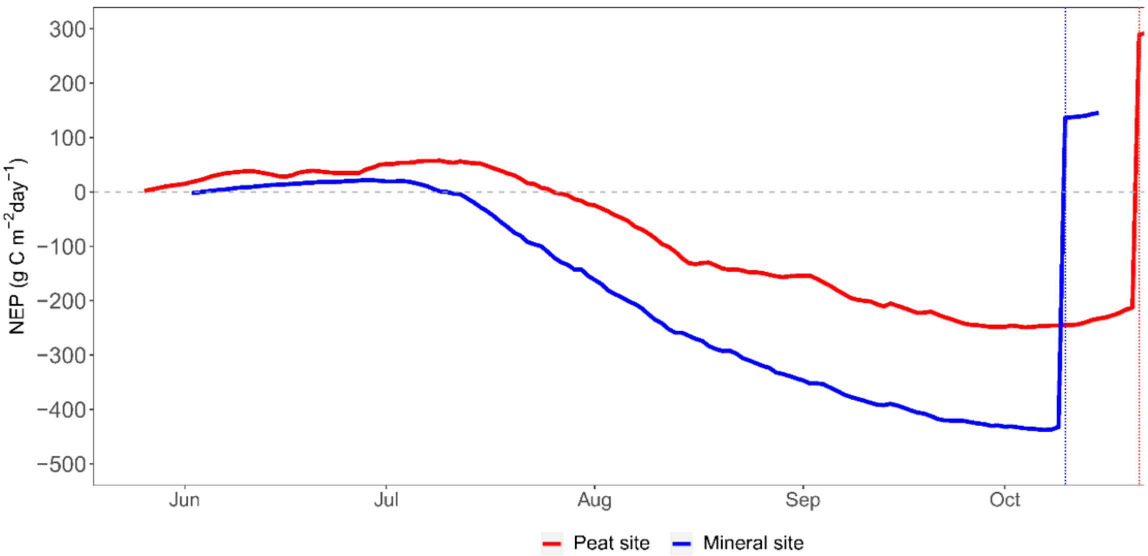

FIGURE 3.5 CUMULATIVE DAILY NEP AT THE STUDY SITES OVER THE MAIZE GROWING SEASON. THE RED AND BLUE DASHED LINES REPRESENT THE MAIZE HARVESTS AT THE PEAT SITE AND MINERAL SITE RESPECTIVELY.

3.4 Discussion

3.4.1 Carbon fluxes

While GPP was higher at PS, more CO_2 was lost to the atmosphere via soil respiration, and so this supports our hypothesis that the CO_2 balance will be higher (more positive) from the maize grown on peat than mineral soil. Given that GPP was similar at both sites, the difference in NEE between sites can be attributed to the fact that TER was nearly twice as high at PS than at MS. The large C store in peat is exposed and rapidly respired following peat drainage and the lowering of the water table due to increased oxygen diffusion, ultimately increasing decomposition of the peat and loss of CO_2 to the atmosphere (Lohila et al., 2003; Evans et al., 2021). Our results corroborate those of Purola and Lehtonen (2002) and Freeman et al. (2022) who found considerably higher rates of CO_2 emission from peatlands used for crop production compared to mineral soils.

This study is among the first to quantify growing season C fluxes of maize grown for bioenergy in the United Kingdom, particularly from bioenergy maize grown on peat. The growing season NEE measured at both study sites sit within the broad range reported throughout the literature (-880 g C m^{-2} from maize grown in the USA; Hollinger et al., 2005 to 64 g C m^{-2} from maize grown in Canada; Eichelmann et al., 2016; Table A2.1). When comparing the growing season NEE of MS in our study with that of other sites in temperate climates with mineral soil, our results are comparable and well within the reported range (Table A2.1). While there are no measurements from maize grown on peat to be compared with those from PS in our study, the growing season NEE from PS is less negative, that is, more of the GPP taken up by the crop was respired as TER, than most sites in temperate climates with mineral soil (Table A2.1).

3.4.2 Net ecosystem productivity

As C_H was greater than NEE, and C_I was minimal at both sites, growing season NEP was positive at both sites, although C losses from PS were over twice those from MS. The negligible contribution of C_I to NEP is observed throughout much of the literature (Table A2.1). The higher C_H at MS is attributed to the higher yield, maize C content and CUE_h at this site compared to PS. The yield at both sites fell within long-term UK averages for whole-crop maize of ~12 t DM ha^{-1} (Macmillan, 2023). The higher CUE_h of the maize grown at MS compared to PS indicates that atmospheric C was converted into new plant biomass more efficiently (Chen et al., 2018; Kim et al., 2022), meaning that less of the CO_2 taken up by the maize during photosynthesis was lost via respiration. Despite PS having lower C_H than MS, it also had a less negative NEE, meaning that PS had a greater loss of C overall, that is, higher NEP.

The NEP of maize during the growing season reported across the literature is highly variable, although most studies report a positive NEP and thus an overall loss of C from the field

(Table A2.1). As well as NEE, the magnitude of C_H is highly variable, ranging from 263 g C m^{-2} for maize grown in China (Liu et al., 2019) to 1083 g C m^{-2} for maize grown in New Zealand (Wall et al., 2020b), and C_I is often zero or negligible in comparison (Table A2.1). Sites with a large C_I can still lose C overall, however, as C_H tends to be larger than NEE, as found by Loubet et al. (2011), Tallec et al. (2013) and Wall et al. (2020). Considering studies from temperate climates only, NEP is generally positive when the whole crop is harvested (i.e., C is lost), whereas NEP is more likely to be negative when only the grain is harvested (i.e., C is accumulated) (Table A2.1), as the C in leaves and stalks is left on the field as crop residue. The NEP of the maize grown at MS in our study (136 g C m^{-2}) is within the broad range reported from sites with mineral soil in temperate climate zones harvesting the whole crop (11 g C m^{-2}; Alberti et al., 2010 to 851 g C m^{-2}; Wall et al., 2020b; Table A2.1), all of which behave as C sources, although to varying magnitudes. For a field to behave as a C sink or to be C neutral, the amount of C remaining in the field must be greater than, or equal to, all other losses of C via exported biomass or TER (Cates and Jackson, 2019). In bioenergy cropping systems, all of the biomass produced is removed for AD, and so very little crop residue is left on the soil surface after harvest. High rates of residue removal, combined with oxidation of the existing SOM (especially in peat soils) can therefore deplete the SOC pool.

3.5 Implications for research and policy

Our results show that growing maize for bioenergy in the UK, especially on peat, is questionable as a climate change mitigation measure due to the ongoing loss of SOC under maize cultivation. Both agri-ecosystems we considered were net C sources once harvested biomass was considered, with emission from peat being two times greater than those of the mineral soil site. There is potential for these losses to exceed the avoided CO_2 emissions from subsequent bioenergy production (Brack and King, 2020). As stated in the UK Government's Biomass Strategy (DESNZ, 2023b), the process of growing biomass for AD should not result in an overall loss of C from an agroecosystem and must reduce CO_2 emissions by at least 60 % relative to fossil fuels once the full production life-cycle is considered. Our data suggest that this may not be possible when growing maize for AD in the UK. There are multiple pathways by which the management practices used to grow maize for AD can cause SOC loss, such as ploughing (Bhattacharyya et al., 2022), residue removal (Raffa et al., 2015; Naylor et al., 2022) and the drainage of peat soils (Evans et al., 2016). Previous research has shown that growing maize is strongly associated with C loss from soil, often to a greater magnitude than other crops such as winter wheat (Ceschia et al., 2010; Poyda et al., 2019; Wall et al., 2020b). Winter wheat has a longer growing season than maize, however, which is likely to be a primary factor controlling the differences in C uptake between the two crops. It is therefore important to consider entire crop rotations and the use of cover crops during fallow periods. It has also been argued that growing maize on productive agricultural land can contribute to food insecurity by reducing the availability of land for growing food crops (Qin et al., 2015; Kiesel et al., 2016), and could also lead to indirect CO_2 emissions as a result of the displacement of food crop production to other areas. If maize is to be grown for use as a bioenergy crop, our results show that it should be grown on mineral soils with a low C content. In addition, good practice would consider growing maize as part of a crop rotation, and with an input of organic materials via organic fertilisers, such as the digestate from the AD plant. Returning digestate from AD will likely be particularly important, as it is C-rich and has a considerable potential to offset C or GHG emissions from vehicles and the AD process itself (Moller, 2015), as well as contributing to a circular economy by reducing waste and enhancing resource efficiency (DESNZ, 2023b). This C input would also offset some of the C removed as harvested biomass and contribute to enhancing the SOC stock (Sun et al., 2023; Yan et al., 2023). Alternatively, growing perennial, rather than annual, bioenergy crops would provide a greater input of C, as these crops often have a greater proportion of their residues left on the soil surface (Ferchaud et al., 2015; Booth and Wentworth, 2023). To avoid SOC loss and compromising food production, bioenergy crops should be grown in addition to, rather than instead of, existing food crops, on land that has a low existing SOC content, with a particular avoidance of peat. If peatlands are to be used for agricultural production they should be managed using methods which aim to minimise C loss, for example by growing food or biomass crops that are tolerant of high water levels (Evans et al., 2021; Freeman et al., 2022).

Further research should consider the impacts of increasing C imports via organic amendments on the NEP of bioenergy maize, and the return of AD digestate on soil health and SOC, to evaluate whether substantially increasing C imports can equate to an overall reduction in SOC loss. As this study only presents data from one growing season, continuing to measure C fluxes from maize grown in the UK would provide a clearer indication of its average NEP and how this is influenced by annual variability in the climate, and over the full crop rotations that characterise agricultural practices in the UK and elsewhere. This would also strengthen the results of our study, as a true comparison between sites requires several years of data, and would help make more robust conclusions on the future management of UK croplands. The two sites in this study received different management, namely in the form of a different planting density, tillage practices, and herbicide inputs. There is the potential for these factors to influence NEP, and so continued research would allow more focus to be placed on the impacts of these management practices. In addition, it would be beneficial to collect data from sites with varying levels of soil C. While growing maize on mineral soils with a low C content may be feasible in the future, the influence of SOM content on NEP is unknown. It is likely that crop N fertilisation will also have a strong impact on the GHG balance as a result of its impact on N_2O emissions. In addition, the low C:N ratio of the soil at both sites may also result in these sites being large sources of N_2O to the atmosphere (Klemedtsson et al., 2005). Thus, future research should measure N_2O emissions in addition to CO_2 fluxes to determine a complete GHG budget associated with growing maize for AD. Finally, it should be considered that our results represent NEP at the field-scale during the maize growing season only, and, while beyond the scope of this study, a life-cycle analysis considering the fate of the crop beyond the farm gate, and accounting for CO_2 emissions associated with the AD process and vehicles, is necessary to fully understand the CO_2 emissions associated with maize production for bioenergy.

CRediT author statement

Isobel L. Lloyd: Conceptualisation, Data curation, Formal analysis, Funding acquisition, Investigation, Visualization, Writing – Original draft, Writing – Review and editing. **Ross Morrison:** Conceptualisation, Data curation, Formal analysis, Funding acquisition, Investigation, Resources, Software, Supervision, Writing – Review and editing. **Richard P. Grayson:** Conceptualisation, Data curation, Investigation, Resources, Supervision, Writing – Review and editing. **Alex M. J. Cumming:** Formal analysis, Investigation, Writing –

Review and editing. **Brenda D'Acunha:** Formal analysis, Investigation, Writing – Review and editing. **Marcelo V. Galdos:** Conceptualisation, Funding acquisition, Supervision, Writing – Review and editing. **Chris D. Evans:** Formal analysis, Investigation, Writing – Review and editing. **Pippa J. Chapman:** Conceptualisation, Funding acquisition, Project administration, Supervision, Writing – Review and editing.

Data availability

Data supporting the findings of this study are available at: https://doi.org/10.5285/9b6c2393-b751-46b4-b139-71ca09321139

Chapter 4 Net ecosystem productivity of a UK cropland over 2.5 years

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Abstract

To combat climate change, agricultural soils must sequester carbon (C) whilst providing sufficient food for the growing human population. Despite this being widely recognized, there is a significant lack of data on the extent of C losses and gains between croplands and the atmosphere associated with the growth of different crops, particularly in the UK. In response to this, the eddy covariance technique was used to measure net ecosystem exchange (NEE) of carbon dioxide (CO_2) and the net ecosystem productivity (NEP) calculated of a UK cropland over 2.5-years, which included the growing seasons of maize, winter wheat and vining pea. Net ecosystem productivity showed the cropland was losing C during the maize growing season (136 g C m^{-2}), but was acting as a C sink during the winter wheat and vining pea growing seasons (-148 g C m^{-2} and -154 g C m^{-2} respectively). Over the complete 2.5-year measurement period, which included fallow periods when there was no crop in the ground, the cropland was a net C source (208 g C m^{-2}) to the atmosphere. This highlights the importance of measuring NEE and NEP during non-productive fallow periods as well as crop growing seasons when estimating cropland NEP. For agri-ecosystems to accrue C, the amount of C added to the system must be greater than all other losses of C as exported biomass and the ecosystem respiration. Increasing C imports by adding organic fertilisers, retaining a greater proportion of crop residues in the field, and/or growing cover crops during fallow periods have the potential to reduce C losses from agri-ecosystems in the UK.

4.1 Introduction

Since the 1940s the proportion of land in the UK dedicated to agricultural production has rapidly expanded to provide food security for a growing human population (Robinson and Sutherland, 2002). Of the 17.2 million ha of agricultural land in the UK, approximately 30 % (5.16 million ha) is used for crop production (DEFRA, 2022a). In conjunction with this agricultural expansion, agricultural management practices have intensified as producers strive to achieve maximum crop yields. These practices include frequent deep tillage, high rates of biomass removal and the growth of crops for non-consumption purposes such as bioenergy production (de Graaff et al., 2019; Schils et al., 2022). Intensive agricultural management practices contribute to the depletion of the soil organic carbon (SOC) pool (Eze et al., 2018) and the resultant emission of carbon dioxide (CO_2) to the atmosphere (Ussiri and Lal, 2009; Bhattacharyya et al., 2022). Carbon (C) losses from these practices are often not compensated for by sufficient additions of C via organic fertiliser (Peng et al., 2021). Intensive agricultural land use and management is therefore responsible for the majority of the ~133 Pg C lost from the top 2 m of global soil over the past 200 years (Sanderman et al., 2017).

In the UK, winter wheat is the most common crop, with winter wheat alone accounting for around 40 % of the country's cropping area (Harkness et al., 2020), and maize and vining pea are commonly grown break crops. Winter wheat yields in the UK average around 8 t ha^{-1}, which is more than double the global average of 3.5 t ha^{-1} (Knight et al., 2012; Slater et al., 2022). In addition, the amount of UK cropland used to grow maize has recently rapidly increased, by 120 % between 2015 and 2021 (DEFRA, 2021c). This increase is mainly attributed to its use in bioenergy production; maize is a favoured bioenergy crop as it has a high biogas output when anaerobically digested (Herrmann, 2013; Bowman and Woroniecka, 2020). In 2020, 75,000 ha of land was used to grow maize for bioenergy (DEFRA, 2021c), mostly in the mid and south of the UK (AHDB, 2018), which is equivalent to 1.5 % of the arable land area in England. There is debate surrounding the use of productive UK croplands to grow non-food crops such as maize for bioenergy production, as this presents the potential to negatively affect food security and increase the reliance on imported food (Kline et al., 2016). In 2023 the UK Government introduced a recommendation to move away from the use of food crops, including maize, for bioenergy production to reduce the pressure on food prices (DESNZ, 2023b). Several studies have found that croplands lose C during maize production but accumulate C under winter wheat (e.g., Buysse et al., 2017 in Belgium; Poyda et al., 2019 in Germany). Around 35,000 ha of UK land, mainly in the east of the country, is used every year to grow vining peas (Ashworth, 2023). Peas are legumes which fix nitrogen (N) into the soil (Jakobsen, 1985) and have a short growing season (Maier et al., 2022b), usually between 3 and 4 months. They therefore have a low requirement for N fertiliser and are a popular break crop between cereals to prevent the spread of pests and diseases (Lavergne et al., 2021).

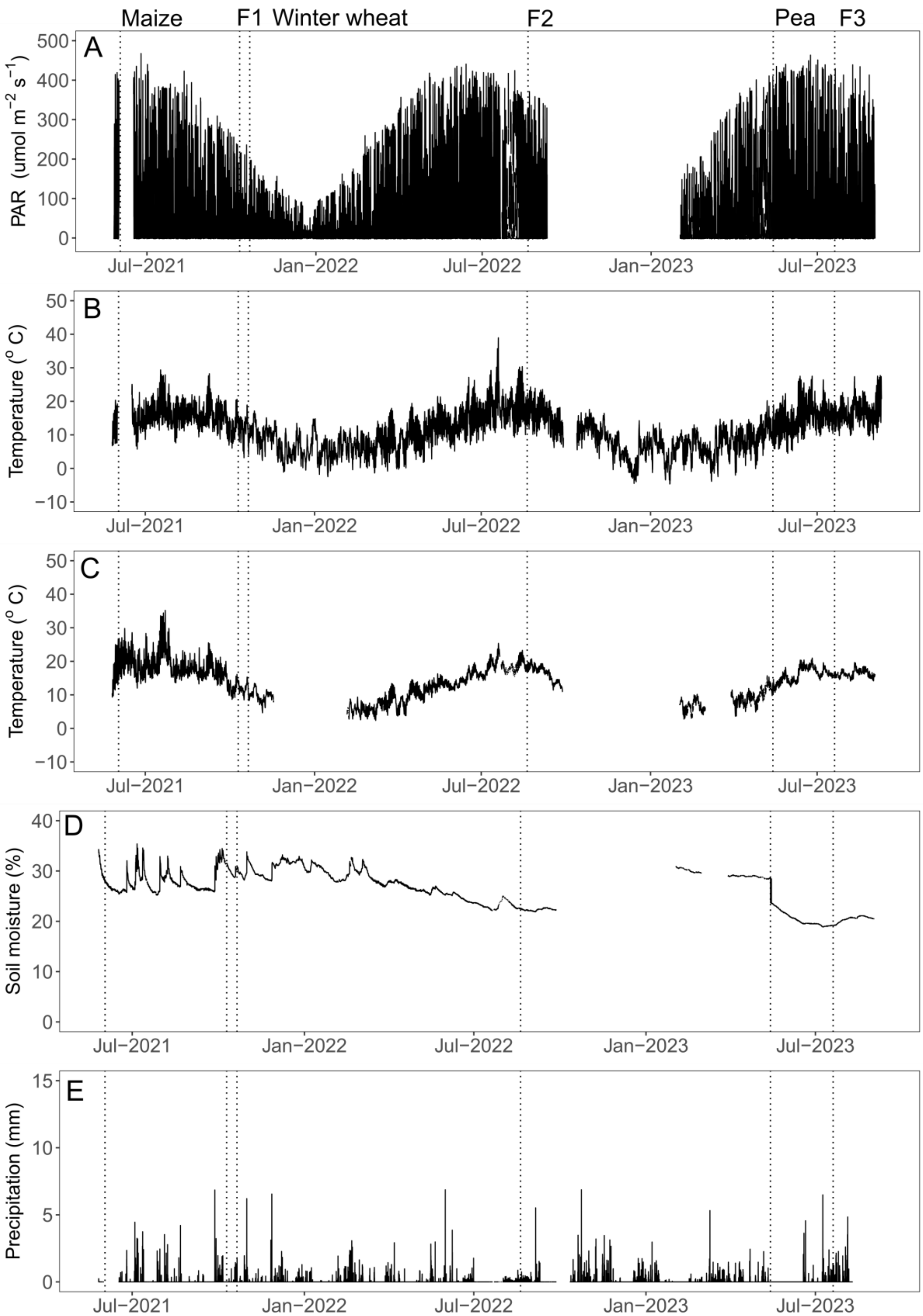
Despite winter wheat, maize and vining pea being common in the UK, there is limited data on the net ecosystem exchange (NEE) and net ecosystem productivity (NEP) of these crops during their growing seasons, and the impact of the fallow periods between these crops on soil C fluxes. This knowledge is crucial for a comprehensive understanding of the current C balance of different commonly grown crops in the UK, information that is critical to facilitate a transition to food production systems that have low CO_2 emissions and that sequester C in agricultural soils. This study aims to begin to address this knowledge gap by determining the impact of crop type on the C source or sink strength of a cropland in the UK. The objectives were to: (i) quantify CO_2 fluxes from a cropland over 2.5-years, calculating NEE for the entire measurement period and during the growing seasons of maize, winter wheat and vining pea; and (ii) estimate the C source or sink strength of the cropland over the 2.5-year measurement period, and during each crop growing season and fallow period, by accounting for lateral C fluxes to determine NEP.

4.2 Methods

4.2.1 Study site

This study was conducted in a crop field (CF) at the University of Leeds Research Farm in Tadcaster, UK, a commercial farm that also supports scientific research. The soil is predominantly a Calcaric Endoleptic Cambisol (IUSS, 2022), 50-90 cm deep, and is underlain by dolomitic limestone (Holden et al., 2019). The farm has a temperate oceanic climate with mild winters and warm summers (Beck et al., 2018); average annual temperature is 9.5 ± 1 °C (Met Office, 2019) and average annual precipitation is 639 ± 142 mm (Met Office, 2006). The crop field (53°51'58.64"N, 1°19'11.08"W; elevation 49 m; 10.4 ha) has been under continuous arable cultivation with conventional tillage since 1994, with a rotation of mainly winter wheat and spring or winter barley, with oilseed rape as a break crop. Soil properties of the field are summarized in Table 3.1 (Chapter 3, see data for Mineral Site).

An eddy covariance (EC) flux tower with associated meteorological and soil sensors was installed in CF in 2021. Measurements from CF over three crop growing seasons (2021-2023), and their associated fallow periods, were used to assess the influence of crop type on agricultural soil C fluxes. The three crops were maize (*Zea mays*), winter wheat (*Triticum aestivum*) and vining pea (*Pisum sativum L.*). Average daily air temperature and total precipitation over the crop growing seasons are presented in Table A3.1; air temperature, soil temperature, soil moisture, photosynthetically active radiation (PAR) and precipitation over the measurement period are shown in Figure 4.1. Detailed management information for CF over the measurement period is presented in Table A4.1.



Maize
F1
Winter wheat
F2
Pea
F3
A
PAR (umol m^{-2} s^{-1})
B
Temperature (^{o} C)
C
Temperature (^{o} C)
D
Soil moisture (%)
E
Precipitation (mm)
Jul−2021
Jan−2022
Jul−2022
Jan−2023
Jul−2023

FIGURE 4.1 (A) PHOTOSYNTHETICALLY ACTIVE RADIATION (PAR), (B) AIR TEMPERATURE, (C) SOIL TEMPERATURE, (D) SOIL MOISTURE AND (E) PRECIPITATION MEASURED OVER EACH CROP GROWING SEASON IN CF. GAPS INDICATE MISSING DATA AND DOTTED LINES INDICATE THE START AND END OF THE MAIZE, WINTER WHEAT AND VINING PEA GROWING SEASONS AND THE FALLOW PERIODS (F1, F2 AND F3).

TABLE 4.1 MANAGEMENT IN CF OVER THE 822-DAY MEASUREMENT PERIOD.

	Date	**Management**
Maize	Spring 2021	Fertiliser (N26+5SO3): 50 kg N ha^{-1}, 9.6 kg S ha^{-1}
	16/04/2021	Herbicide (Amega Duo): 2.1 L ha^{-1} Application aid (Phase II): 0.5 L ha^{-1} Application aid (Spryte Aqua): 0.5 L ha^{-1}
	06/05/2021	Herbicide (Pendimethalin): 3.3 L ha^{-1} Herbicide (Glyphosate): 2 L ha^{-1}
	18/05/2021 19/05/2021	Non-inversion tillage: 20-25 cm
	02/06/2021	Planted maize (Fieldstar variety) using precision drill: 110,000 seeds ha^{-1} Fertiliser (Di-ammonium phosphate): 125 kg ha^{-1} (of which 22.5 kg N ha^{-1} and 57.5 kg P ha^{-1})
	10/10/2021	Harvest: 12.3 t ha^{-1} dry matter
F1	20/10/2021	Non-inversion tillage: 25 cm
Winter wheat	21/10/2021	Non-inversion tillage: 25 cm Planted winter wheat (Extase variety) using precision drill: 440 seeds m^{-2}
	10/11/2021	Herbicide (Flufenacet + pendimethalin): 4 L ha^{-1}
	01/02/2022 21/03/2022	Fertiliser (Pig slurry): 30 m^{-3} ha^{-1} (of which 87 kg N ha^{-1}, 54.9 kg P ha^{-1}, 61.8 kg K ha^{-1} and 450 kg C ha^{-1})
	16/04/2022	Fertiliser (N26+5SO3): 120 kg N ha^{-1}, 23 kg S ha^{-1}
	26/04/2022	Fungicide (Bixafen, fluopyram + prothioconazole): 0.9 L ha^{-1} Plant growth regulator (Chlormequat chloride): 2.2 L ha^{-1}
	14/05/2022	Herbicide (Pyroxsulam + floraulam): 265 g ha^{-1}
	20/05/2022	Fungicide (Fenpicoxamid + prothioconazole): 1.5 L ha^{-1} Plant growth regulator (Mepiquat chloride + 2-chloroethylphosphoric acid): 1 L ha^{-1}

	20/08/2022	Harvest: 15.5 t ha^{-1} dry matter (10.3 t ha^{-1} grain, 5.2 t ha^{-1} straw)
F2	October 2022	Non-inversion tillage: 25 cm
Vining pea	14/05/2023	Planted vining pea (Noroit variety) using 6 m rapid drill: 145 seeds m^{-2}
	15/05/2023	Herbicide (Nirvana): 3.5 L ha^{-1} Herbicide (Sirtaki): 0.15 L ha^{-1} Application aid (Grounded AD): 0.2 L ha^{-1}
	17/06/2023	Herbicide (Tropotox): 1.8 L ha^{-1} Herbicide (Benta): 1.8 L ha^{-1}
	21/06/2023	Insecticide (Teppeki): 0.4 kg ha^{-1}
	20/07/2023	Harvest: 1.1 t ha^{-1} dry matter (pods only)

4.2.2 Measurement of CO_2 fluxes

Turbulent fluxes of CO_2 (μmol m^{-2} s^{-1}) and sensible and latent heat fluxes (H; LE; W m^{-2}) were measured using the EC technique; the EC set up was as described for Mineral Site in Section 3.2.2 (Chapter 3). The maximum flux footprint radius was 440 m, with a mean peak distance of 43 m and an average 90 % contribution of 119 m (Figure A3.1). The total monitoring period was 822 days (02/06/2021-01/09/2023), although data is only reported for 695 days (02/06/2021-26/09/2022 and 02/02/2023-01/09/2023) due to a period of instrument failure between 27/09/2022-01/02/2023. The measurement period encompassed three crop growing seasons (maize: 131 days (02/06/2021-10/10/2021), winter wheat: 304 days (21/10/2021-20/08/2022) and vining pea: 68 days (14/05/2023-20/07/2023)) and three fallow periods (F1: 11 days (11/10/2021-20/10/2021), F2: 266 days (21/08/2021-13/05/2023) and F3: 43 days (21/07/2023-01/09/2023)).

4.2.3 Calculation of CO_2 fluxes

Flux data processing, including the calculation of CO_2 fluxes, quality control and gap-filling was conducted as described in Section 3.2.3 (Chapter 3). During quality control, data were removed when friction velocity (u*; m s^{-1}) < 0.1. Gap-filled NEE accounted for 27 % of the overall dataset. Between 27/09/2022 and 01/02/2023 (128 days) there was a prolonged period of instrument failure, meaning that no CO_2 fluxes were recorded for the majority of F2 – equivalent to 16 % of the total 822-day monitoring period. This period of missing data was considered too large to be gap-filled.

Net ecosystem exchange is calculated as the difference between gross primary productivity (GPP) and total ecosystem respiration (TER) as shown in Equation 4.1 (Smith et al., 2010); following gap-filling, NEE was partitioned into GPP and TER (Reichstein et al., 2016). The micrometeorological sign convention is used for NEE, where a positive value indicates the ecosystem is losing C and a negative value indicates the ecosystem is accumulating C (Baldocchi et al., 2003).

$$NEE = TER - GPP \qquad \text{(Equation 4.1)}$$

4.2.4 Ancillary measurements

Additional micrometeorological measurements were recorded for the calculation of turbulent fluxes, as described for Mineral Site in Section 3.2.4 (Chapter 3).

4.2.5 Energy balance

The degree of energy balance closure (EBC) is used to assess the quality of EC data at a given site (Aubinet et al., 2001; Wilson et al., 2002). It compares the sum of H and LE measured by EC, with energy balance terms measured by other means (i.e., net radiation (Rnet) and soil heat flux (G)). Over the 695-day measurement period in CF, turbulent fluxes accounted for 74 %, 72 % and 45 % of the available energy in 2021, 2022 and 2023 respectively (Figure 4.2). The amount of variance (as measured by R^2 values) for all years are within the typical range of EC measurements (i.e., 0.7-0.9) (Wilson et al., 2002; Foken, 2008; Wagle et al., 2018).

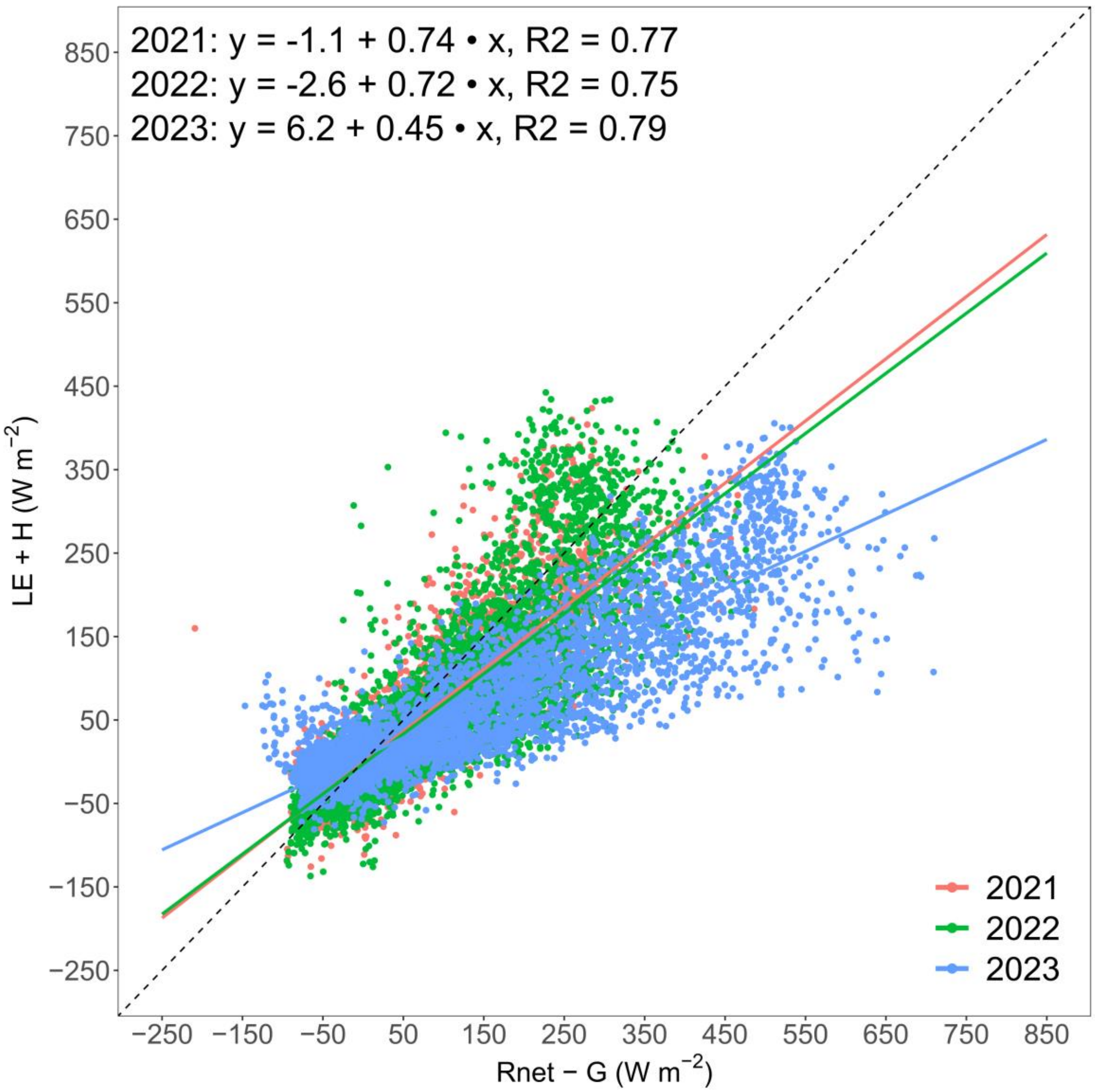


FIGURE 4.2 ENERGY BALANCE IN CF OVER THE 695-DAY MEASUREMENT PERIOD (02/06/2021-01/09/2023) SPLIT BY YEAR, WHERE H IS SENSIBLE HEAT FLUX, LE IS LATENT HEAT FLUX, RNET IS NET RADIATION AND G IS SOIL HEAT FLUX.

4.2.6 Net ecosystem productivity and crop carbon use efficiency

To estimate the C source or sink strength of the agroecosystems, NEP was calculated according to Equation 4.2. Exports of C (C_H) were in the form of harvested vegetation; the entire maize crop (i.e., whole-crop maize) and winter wheat crop were harvested, whereas for vining pea only the pods were harvested and crop residues were left on the field. The C removed via harvested vegetation was calculated by analyzing the C content of biomass samples taken from the field on the day of harvest, and scaling this to the reported yield (as in Abraha et al., 2018 and Poyda et al., 2019). As the aim of this study is to assess NEP at the field scale, it is assumed that all C within the removed biomass was converted to atmospheric CO_2 on leaving the field (Eichelmann et al., 2016; Morrison et al., 2019) via anaerobic digestion (AD) for bioenergy in the case of maize, or respiration from humans that consumed the winter wheat and vining pea. We acknowledge that the AD process involves further C fluxes – in some cases the digestate is returned to the field, although not the case here. A full life-cycle analysis, beyond the field boundary, is beyond the scope of this study, however. Carbon imports (C_I) were in the form of seed (calculated according to Yue et al., 2023) and organic amendments. As in Evans et al. (2021), the micrometeorological sign convention is used for NEP, where a positive value indicates C loss and a negative value indicates C gain.

$$NEP = NEE + C_H - C_I \qquad \text{(Equation 4.2)}$$

The C use efficiency of harvested material (CUE_h) is a measure of how effectively atmospheric C is converted into new plant material (Chen et al., 2018) and is calculated according to Equation 4.3 (Kim et al., 2022).

$$CUE_h = \frac{C_H}{GPP} \qquad \text{(Equation 4.3)}$$

4.3 Results and discussion

4.3.1 Carbon fluxes

The crop field exhibited *in-situ* net CO_2 uptake as NEE during all crop growing seasons; with winter wheat having the most negative cumulative NEE (-648 g C m^{-2}), followed by maize (-429 g C m^{-2}) and vining pea (-193 g C m^{-2}) (Figure 4.3; Figure 4.4; Table 4.2). Over the 2.5-year measurement period the field had an overall *in-situ* net CO_2 uptake as NEE of -897 g C m^{-2} (Table 4.2).

The pattern of NEE over the maize and winter wheat growing seasons was as expected and similar to that observed amongst the literature (Anthoni et al., 2004; Hollinger et al., 2005; Adviento-Borbe et al., 2007; Moureaux et al., 2008; Gebremedhin et al., 2012; Liu et al., 2019; Franco-Luesma et al., 2020b; Niu et al., 2022). The field behaved as a CO_2 source at the start of the maize growing season and was a CO_2 sink from crop emergence onwards, with the greatest uptake occurring between July and September. For winter wheat, CF behaved as a CO_2 source from October to March, as the winter wheat was dormant and there was very little photosynthesis, and then switched to a CO_2 sink between March and June, and then back to a CO_2 source in July due to senescence of the crop. Vining pea had the greatest CO_2 uptake in June and continued to behave as a CO_2 sink until the crop was harvested.

The difference in cumulative NEE between the three crops can be attributed to the large variation between their GPP and TER values, and differences in the length of the crop growing seasons. Gross primary productivity and TER were highest for winter wheat, although the difference between these two values was the greatest of all the crops, meaning that it had the greatest CO_2 uptake (i.e., most negative NEE). The mean daily NEE of maize and vining pea were more negative than that of winter wheat however – -3 g C m^{-2} day^{-1} for maize and vining pea and -2 g C m^{-2} day^{-1} for winter wheat (Table 4.2) – which reflects the considerable period of dormancy at the start of the winter wheat growing season. The considerably higher total TER from winter wheat compared to maize and vining pea can also be attributed to this long period of dormancy at the start of the winter wheat growing season where photosynthesis was limited due to the low leaf area, and so overall TER was not balanced by GPP (Liu et al., 2019). Winter wheat had the longest growing season of the three crops studied and thus the greatest amount of time in which to photosynthesise following emergence (Prescher et al., 2010), hence its higher GPP. Several studies comparing NEE between maize and winter wheat have found that winter wheat has a greater CO_2 uptake than maize as a result of its longer growing season (Prescher et al., 2010; Tallec et al., 2013; Wang et al., 2015; Lv et al., 2022). Data on C fluxes from crop rotations containing peas is limited, with no existing measurements from peas grown in the UK. Of the crops in our study, vining pea had the lowest total GPP and TER which can be attributed to its very short growing season in comparison to maize and winter wheat, and it was also grown in drought conditions, which is likely to have affected its productivity relative to if it had been grown in a normal year. The vining pea, however, did still show CO_2 uptake. This contradicts Ceschia et al. (2010) and Lopez-Garrido et al. (2014) who propse that NEE would be positive (i.e., CO_2 emission) or C neutral during the pea growing season due to its low capacity to photosynthesise as a result of its low leaf area and short growing season.

During the three fallow periods, TER was greater than GPP, which resulted in an overall *in-situ* emission of CO_2 (Table 4.2). Tillage events occurred during F1 and F2 to both prepare the soil for planting and to prevent the emergence of weeds and volunteer crops. These events will have disturbed the soil structure, exposing SOC and oxidizing it to CO_2 (McGinn and Akinremi, 2001; Al-Kaisi and Yin, 2005; Moureaux et al., 2006; Reicosky and Archer, 2007). The considerably lower NEE of F1 (22 g C m^{-2}) compared to F2 (183 g C m^{-2}) is due to F1 being shorter than F2 (11 days compared to 138), and thus a shorter fallow period and less soil respiration. Furthermore, the NEE value for F2 will be higher than that reported, as around half of the CO_2 fluxes during this period were not measured. The large emission of CO_2 during F3 (170 g C m^{-2}), despite this period only being 43 days, is a result of TER being considerably higher than GPP, as the vining pea residues were decomposing rapidly on the soil surface following the harvest event. This is reflected in the daily TER values, which show mean daily TER was over twice as high for vining pea as it was for winter wheat (5 g C m^{-2} day^{-1} compared to 2 g C m^{-2} day^{-1}) (Table 4.2) due to this rapid residue decomposition. The CO_2 emission during these fallow periods contributes to increasing the NEP of CF. Amongst the literature, NEE tends to be measured over crop growing seasons only, and there are considerably fewer reports of NEE during fallow periods or over entire crop rotations. Davis et al. (2010) report NEE during fallow periods to range between 0.5 and 1 g C m^{-2} day^{-1}, and Liu et al. (2019) find an average of 1 g C m^{-2} day^{-1} NEE during fallow periods, values which are considerably lower than the mean daily NEE measured during the fallow periods in CF in this study. Emissions of CO_2 may be reduced during fallow periods by growing cover crops as the field would be able to photosynthesise (Steenwerth and Belina, 2008; Ruis et al., 2017; Jian et al., 2020; Rigon and Calonego, 2020), however it is important to then consider the fate of the cover crop, as this CO_2 uptake may be counteracted by emissions from cover crop residue decomposition on the soil surface (Nilahyne et al., 2019; Blanco-

Canqui et al., 2022). Sanz-Cobena et al. (2014) and Nguyen and Kravchenko (2021) observed elevated CO_2 fluxes during periods when cover crops were grown relative to when they were not, with Liebig et al. (2010) noting that CO_2 emissions may be lower in cropping systems that include fallow periods rather than cover crops due to lower C_I and thus no priming effect.

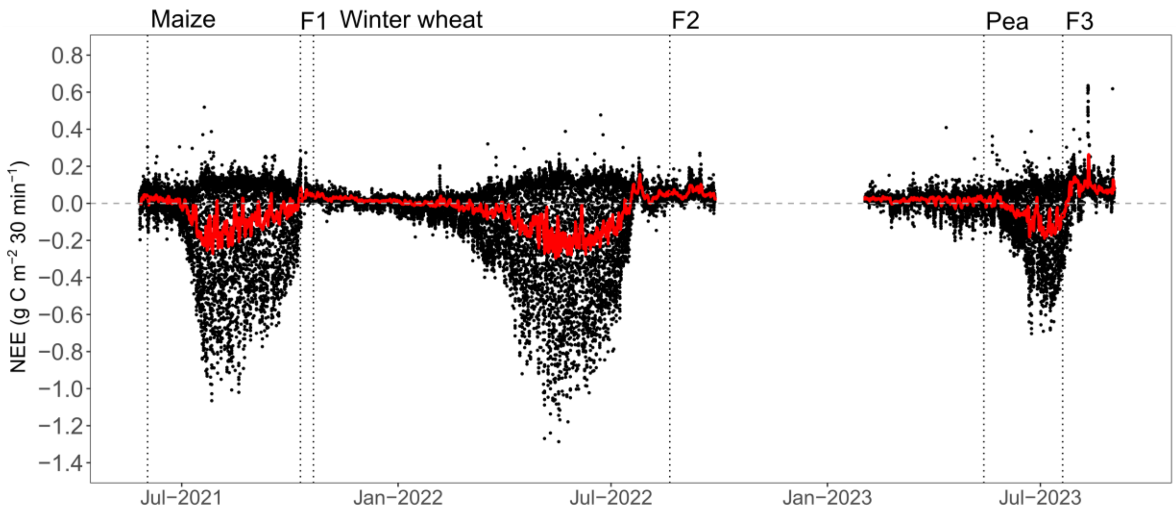


FIGURE 4.3 30-MINUTE FLUXES OF NEE OVER THE 2.5-YEAR MEASUREMENT PERIOD. DOTTED LINES INDICATE THE START AND END OF THE MAIZE, WINTER WHEAT AND VINING PEA GROWING SEASONS AND THE FALLOW PERIODS (F1, F2 AND F3). THE RED LINE INDICATES THE ROLLING DAILY MEAN.

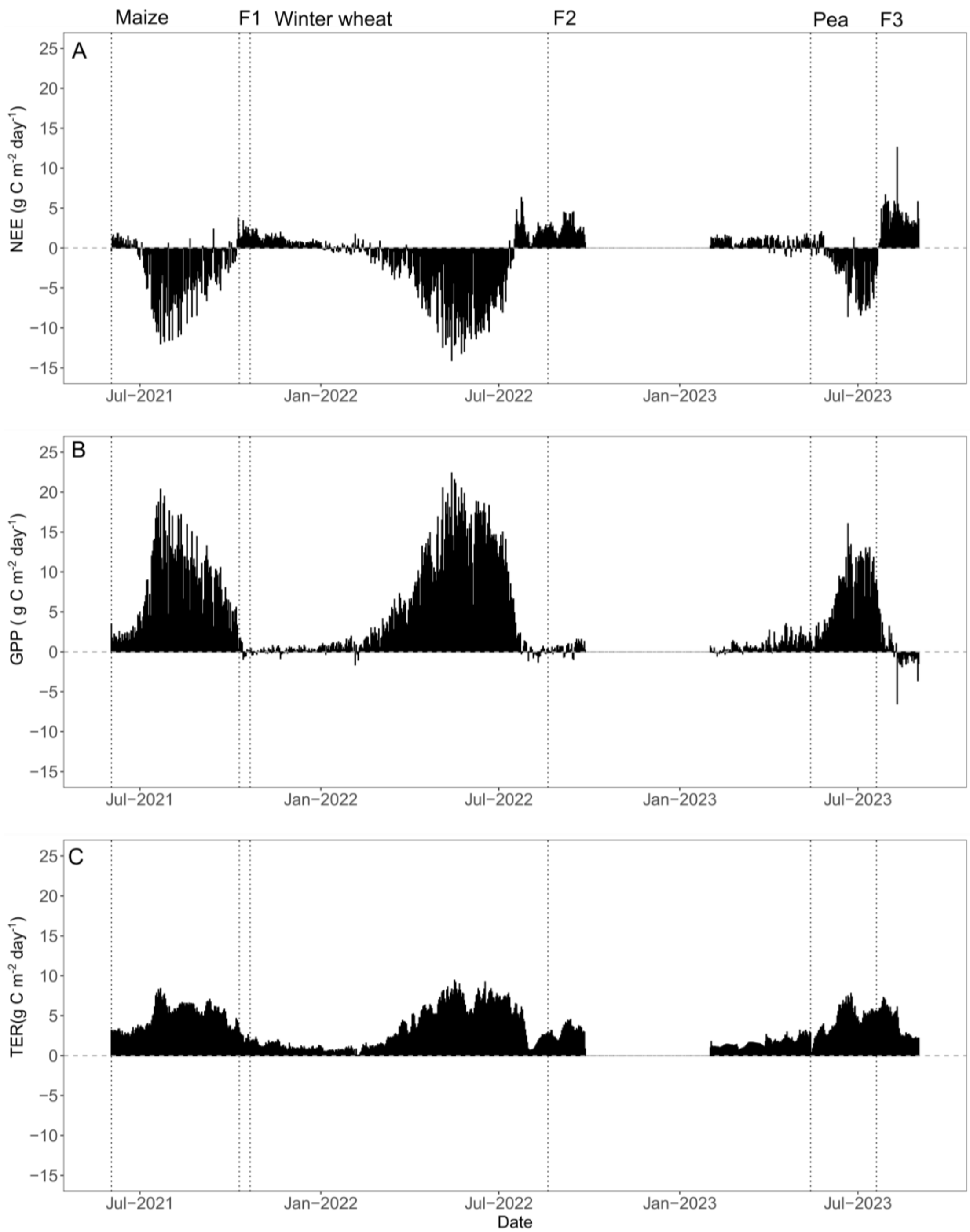


FIGURE 4.4 DAILY (A) NEE, (B) GPP AND (C) TER FOR THE CROP GROWING SEASONS AND FALLOW PERIODS OVER THE 822-DAY MONITORING PERIOD (02/06/2021-01/09/2023). DOTTED LINES INDICATE THE START AND END OF THE MAIZE, WINTER WHEAT AND VINING PEA GROWING SEASONS AND THE FALLOW PERIODS (F1, F2 AND F3). NOTE THAT THE PERIOD OF NO DATA IN F2 IS THE 128-DAY PERIOD IN WHICH FLUXES WERE NOT MEASURED.

TABLE 4.2 CARBON BUDGET IN CF OVER THE 695-DAY MEASUREMENT PERIOD (02/06/2021-01/09/2023) ± ROOT SUM SQUARED (ASIDE FROM C_H WHERE ± REPERSENTS STANDARD DEVIATION OF BIOMASS CARBON CONTENT UPSCALES TO REPORTED BIOMASS OFFTAKE, AND C_I WHERE ± REPRESENTS STANDARD DEVIATION OF PIG SLURRY CARBON CONTENT). F1, F2 AND F3 REPRESENT THE FALLOW PERIODS. NOTE THAT DATA IN F2 (*) WAS MEASURED OVER 138 DAYS AS THERE WAS A LARGE PERIOD OF MISSING DATA BETWEEN 27/09/2022 AND 01/02/2023 (128 DAYS). THIS IS REFLECTED IN THE NUMBER OF MEASUREMENT DAYS FOR THE TOTAL PERIOD. THE MICROMETEOROLOGICAL SIGN CONVENTION IS USED FOR NEE AND NEP WHERE POSITIVE VALUES INDICATE C LOSS AND NEGATIVE VALUES INDICATE C GAIN.

	Maize	**F1**	**Winter wheat**	**F2**	**Vining pea**	**F3**	**Total period**
Measurement period	02/06/2021-10/10/2021	11/10/2021-20/10/2021	21/10/2021-20/08/2022	21/08/2022-13/05/2023 *	14/05/2023-20/07/2023	21/07/2023-01/09/2023	02/06/2021-10/09/2023
Number of measurement days	131	11	304	138	68	43	695
NEP (g C m^{-2})	136 ± 122	22 ± 7	-148 ± 48	183 ± 20	-154 ± 34	170 ± 29	208 ± 261
NEP (t CO_2-equivalent ha^{-1})	5 ± 5	1 ± 0	-5 ± 2	7 ± 1	9 ± 1	6 ± 1	8 ± 10
Mean daily NEP (g C m^{-2} day^{-1})	1 ± 1	2 ± 1	-0.5 ± 0	1 ± 0	-2 ± 1	4 ± 1	0.2 ± 0.5
NEE (g C m^{-2})	-429 ± 58	22 ± 7	-648 ± 83	183 ± 20	-193 ± 34	169 ± 29	-897 ± 112
NEE (t CO_2-equivalent ha^{-1})	-16 ± 2	1 ± 0	-24 ± 3	7 ± 1	-7 ± 1	6 ± 1	-33 ± 4
Mean daily NEE (g C m^{-2} day^{-1})	-3 ± 0	2 ± 1	-2 ± 0	1 ± 0	-3 ± 0.5	4 ± 1	-1 ± 0
TER (g C m^{-2})	678 ± 62	24 ± 7	1031 ± 76	293 ± 27	313 ± 40	181 ± 30	2490 ± 112
Mean daily TER (g C m^{-2} day^{-1})	5 ± 0	2 ± 1	3 ± 0	2 ± 0	5 ± 1	4 ± 1	4 ± 0
GPP (g C m^{-2})	1107 ± 113	2 ± 3	1679 ± 150	110 ± 14	506 ± 70	12 ± 16	3388 ± 200
Mean daily GPP (g C m^{-2} day^{-1})	8 ± 1	0.2 ± 0	6 ± 0.5	1 ± 0	7 ± 1	0.3 ± 0	5 ± 0
Yield (t DM ha^{-1})	Whole-crop	-	Straw Grain	-	Pods	-	-

	12.3		5.2	10.3		1.1		
C content of harvested biomass (%)	46	-	40		-	40	-	-
CUE_h (g C g C^{-1})	0.51	-	0.37		-	0.09	-	-
C_H (g C m^{-2})	567 ± 65	-	616 ± 6		-	45 ± 0	-	-
C_I (g C m^{-2})	2 ± 0	-	116 ± 41		-	6 ± 0	-	-

4.3.2 Net ecosystem productivity

Cumulative NEP was positive over the maize growing season (136 g C m^{-2}) with CF behaving as a C source. In contrast, cumulative NEP was negative during the winter wheat and vining pea growing seasons (-148 g C m^{-2} and -154 g C m^{-2}, respectively), with CF behaving as a C sink (Table 4.2; Figure 4.5). This corroborates the results of Buysse et al. (2017) and Poyda et al. (2019) who found a crop field to behave as a C source during the maize growing season and as a C sink during the winter wheat growing season. However, over the 2.5-year measurement period, the cropland was losing C (Table 4.2; Figure 4.5), which highlights the importance of including fallow periods in C budget calculations so as not to overestimate C uptake capacity of agri-ecosystems.

During the maize growing season, C_H was greater than NEE, and C_I was negligible, resulting in a positive NEP (Table 4.2). Likewise, C_I was negligible for vining pea (seed only), however C_H for vining pea was over four times smaller than its NEE. This is because only the pea pods were removed at harvest and the remaining aboveground biomass C was left in the field as crop resuide. Winter wheat also had less C removed from the field than was added; the C_H of grain and straw was slightly less than the CO_2 uptake as NEE, and C_I was considerable (organic fertiliser plus seed). The results of published studies show that agri-ecosystems usually behave as C sources over the maize growing season, however there is a tendency for NEP to be lower, or even negative, when only maize grain is removed compared to when the whole-crop is harvested (Table A3.2), as C_H is lower. For winter wheat there is a less obvious pattern, with most published studies finding winter wheat to behave as a C sink, although this conclusion is typically based on systems where only grain is harvested (Table A3.2). As in this study, many farmers harvest winter wheat residues as straw as well as grain, as it is highly valuable for use as feed and bedding for livestock. Despite this, agri-ecosystems where both straw and grain are harvested are presented considerably less throughout the literature. Similar to our results, Aubinet et al. (2009), Schmidt et al. (2012) and Tallec et al. (2013) measured NEP of wheat agri-ecosystems where both the grain and straw were removed, and found the fields were behaving as C sinks (Table A3.2). This highlights the potential for agri-ecosystems to behave as C sinks even when straw is removed in addition to the grain, although it is important to be aware that these are growing season measurements only. It should also be noted that the removal of straw for livestock feed may increase non-CO_2 emissions such as methane (CH_4) via enteric fermentation, with the amount of CH_4 release being partially dependent on livestock diet (Beauchemin et al., 2009). Multiple studies have observed that retaining crop residues on the soil surface can increase the SOC pool (Raffa et al., 2015; Zhan et al., 2019; Zhao et al., 2020) as more C is kept within an agri-ecosystem (i.e., C_H is reduced). Crop residue retention can increase CO_2 emission from a field, however, as the decomposition of residues, either on the soil surface or after being ploughed into the soil, can increase soil microbial activity and thus respiration (Brye et al., 2006; Sainju et al., 2010). Winter wheat residue has a particularly high C:N ratio and decomposes slowly when left on the soil surface, so CO_2 emissions can be higher when residues are retained during fallow periods as they decompose over that time (Gebremedhin et al., 2012; Veeck et al., 2022). Increasing C_I to a field, either via the addition of organic amendments or decreasing C_H by retaining more crop residues on the soil surface, therefore has considerable potential to increase soil C sequestration and reduce C loss (i.e., decrease NEP). This will vary depending on the amount of residues retained (Jans et al., 2010), however, and it will be crucial that C is being added to the system rather than being transferred between sites, which is not a form of C sequestration.

Whilst growing season measurements allow us to compare C dynamics between different crops, it is important to consider the system as a whole and to account for C fluxes during fallow periods. When summing the fluxes measured during the three crop growing systems, NEP for CF would be -166 g C m^{-2}, which shows an overall C uptake. However, when summing all fluxes measured during the 2.5-year/695-day measurement period (i.e., crop growing seasons plus fallow periods), NEP for CF was 208 g C m^{-2} showing an overall C loss (Table 4.2). There is no crop growth during fallow periods and thus no opportunity for the field to photosynthesise and offset the constant TER with GPP; furthermore any crop residues from the previously harvested crop are left on the soil surface, also contributing to TER as they decompose (Veeck et al., 2022). Net ecosystem exchange was positive during all fallow periods which increased the NEP of the field overall. Therefore, these fallow periods cannot be ignored; fallow periods accounted for 28 % of the data collected during the 695-day measurement period in CF. Reporting the NEP of a cropland based on the C fluxes measured during crop growing seasons only can therefore be misleading, as there is huge potential for the C sequestration rate to be overestimated if fluxes during fallow periods are ignored. Ideally, the C budget of the entire crop rotation should be measured to fully understand the C losses or gains associated with an agri-ecosystem. It should be noted that a large proportion of the data in F2 is missing (128 days out of a total of 266 days) and so we provide a conservative estimate for the field. For the field to be C neutral during the measurement period, providing NEE and C_H remain the same, C_I would have to be increased by at least 208 g C m^{-2}, or 2.08 t C ha^{-1}. This could be achieved by increasing organic inputs to add C via OM (Hijbeek et al., 2019; Li et al., 2023; DEFRA, no date b) or retaining crop residues on the soil surface (Stella et al., 2019; Haas et al., 2022; Aditi et al., 2023). The feasibility of adding this much C via organic amendments will depend on the type of amendment, and its C content. There is the potential for crop residue retention to result in elevated CO_2 emissions however, as the residues decompose on the soil surface or enhance the decomposition of older SOC by facilitating increased soil microbial activity (Nilahyane et al., 2019; Blanco-Canqui et al., 2022). Alternatively, an effort could be made to reduce C_H by decreasing the amount of biomass removed from the field as harvested vegetation, however this is unlikely to present a feasible option as the demand for food is expected to increase with the growing global population (HYDE, 2017).

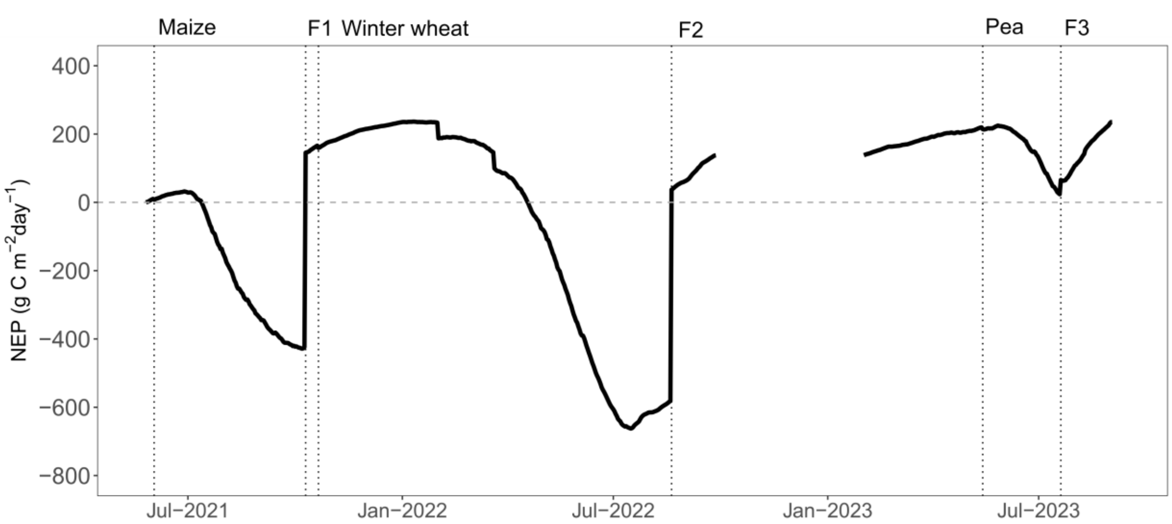


FIGURE 4.5 CUMULATIVE DAILY NEP IN CF OVER THE 822-DAY MONITORING PERIOD. DOTTED LINES INDICATE THE START AND END OF THE MAIZE, WINTER WHEAT AND VINING PEA GROWING SEASONS AND THE FALLOW PERIODS (F1, F3 AND F3).

4.3.3 Limitations and implications for research

The C dynamics of an agri-ecosystem and its potential to behave as a C sink or source are strongly influenced by crop type and management practices. The results from the 695-day measurement period in CF show that the inclusion of fallow periods in a crop rotation, whether left bare or covered with crop residues, has a clear impact on the NEP of an agri-ecosystem. Across the literature, C fluxes during fallow periods are reported considerably less than those measured during crop growing seasons, however they are highly influential on the NEP of a cropland, as shown by Davis et al. (2010) and Liu et al. (2019). Subsequently, there is a clear need for measurements of NEE in croplands to be extended beyond crop growing seasons and include fallow emissions. During the fallow periods in CF there was a large C loss from the field, and so future research also should explore the extent to which increasing C_I, as organic fertiliser or retained crop residues, or growing cover crops during fallow periods, can decrease NEP. Here, we present data for only three crops and three fallow periods, which is not the entire rotation of the field, and so further research should strive to measure the NEP of croplands over entire crop rotations to fully understand the C dynamics associated with long-term agricultural management. As only one growing season of each crop is presented, the study does not account for any potential variation of the NEE and NEP of these crops with varying climate conditions over time or in different areas of the UK. Furthermore, the crops were grown as part of a rotation in the same field, and so no impact of soil type can be shown. There is considerable potential for the NEE and NEP of an agri-ecosystem to be affected by the climate conditions and soil type (Dilustro et al., 2005; Jager et al., 2011; Shakoor et al., 2021) (see also Chapter 2), and so there is a clear need to measure C fluxes from these commonly grown crops both over time and across the UK to place the results of this study into the wider context. To reduce the negative impact of agriculture on the environment, non-CO_2 greenhouse gas emissions must also be considered – these being methane (CH_4) and nitrous oxide (N_2O) which are emitted in large quantities from the agricultural sector. Further work should aim to measure these fluxes in addition to CO_2 to account for any trade-offs associated with the use of certain management practices, including the choice of crop grown. This could be achieved by using greenhouse gas flux chambers at sites where EC is used to measure NEE.

In addition, the period of missing data during F2 (128 days) cannot be ignored when discussing the limitations of the dataset. Whilst short gaps in NEE data can be gap-filled using established methods, as was done for the remainder of the dataset, the large section of missing data between 27/09/2022 and 01/02/2023 is too large to gap-fill in this way. During F2, CF was fallow with the soil surface exposed as no crops were growing and there were no crop residues covering the soil surface. Multiple tillage events also occurred during this time. As both leaving soil exposed and tilling the soil is known to cause an emission of CO_2 (Lal, 2016; Tanveer et al., 2018; Daryanto et al., 2019), it is highly likely that CO_2 would have been emitted during F2 and thus the reported NEE value is likely to be underestimating the fluxes from the field. Amongst the literature, methods for filling in longer gaps have been utilised however are not yet established or standardised. One such example is the use of linear regression to estimate CO_2 fluxes based on air temperature and PAR; measurements recorded during similar site conditions (i.e., fallow) would be used to create a model showing the response of CO_2 to air temperature and PAR, and this model would be applied to the air temperature and PAR recorded during the period of missing data to estimate CO_2 fluxes (Lucas-Moffat et al., 2022). Alternatively, the mean daily NEE recorded by the flux tower for the remainder of the study period could be calculated and this multiplied by the number of days of missing data (Keane et al., 2019). Table 4.3 shows NEE and NEP for the 2.5-year period as reported in this study, and estimated using the two aforementioned methods. Compared to the -897 g C m^{-2} and 208 g C m^{-2} reported in this study for NEE and NEP respectively, the values estimated using the linear regression method are higher, suggesting greater C loss, and those estimated by the mean daily method are lower, suggesting greater C uptake. This highlights the difficulty associated with estimating large periods of missing data; CO_2 fluxes are highly dependent on the climate and soil conditions and the management practices used, so it is difficult to provide an accurate estimate of CO_2 fluxes based on other periods of data unless the environmental conditions and management practices are near identical.

TABLE 4.3 NET ECOSYSTEM EXCHANGE AND NET ECOSYSTEM PRODUCTIVITY IN CF OVER THE 2.5-YEAR MEASUREMENT PERIOD USING THREE APPROACHES TO GAP-FILLING THE 128-DAY PERIOD OF MISSING DATA (27/09/2022-01/02/2023). 'NON-GAP-FILLED' refers to the approach taken in Chapter 4 where the gap in the data was not filled. 'Linear regression' refers to the use of linear regression to predict the NEE values (as in Lucas-Moffat et al., 2022) based on data measured between 11/10/2021-21/10/2021, 21/08/2022-26/09/2022 and 02/02/2023-13/05/2023 (149 days, when field conditions were similar to those during the period of missing data) and added to the measured NEE. 'Mean daily' refers to the calculation of mean daily NEE across the entire 2.5-year measurement period (excluding the period of missing data), this value being multiplied by the number of days of missing data (128) as in Keane et al. (2019) and added to the measured NEE.

	NEE (g C m^{-2})	**NEP (g C m^{-2})**
Non-gap-filled	-897	208
Linear regression	-743	361
Mean daily	-1062	42

4.4 Conclusions

A comprehensive understanding of the extent of C loss or gain as a result of the growth of different crops and associated management practices in the UK is critical for reducing C emissions and combatting climate change. This information will support policymakers to make evidence-based decisions on how to best support farmers to adapt their management practices to reduce their greenhouse gas emissions. This study measured the NEP of a cropland over 2.5-years, encompassing three crop growing seasons and three fallow periods. Of the crops grown in the rotation, maize behaved as a C source over its growing season (136 g C m^{-2}), whereas winter wheat and vining pea were C sinks (-148 g C m^{-2} and -154 g C m^{-2} respectively) during their growing seasons. When considering the fallow periods in between crops in addition to the growing seasons, the cropland was a C source (208 g C m^{-2}) over the 2.5-year study period. In order for the cropland to behave as a C sink, the amount of C added to the field must be greater than the amount exported as harvested biomass. The demand for food crops will continue to grow with the global population, and so reducing the amount of exported biomass, and thus C, from agri-ecosystems is an unlikely solution. Increasing additions of C via organic fertiliser, by returning crop residues, and by growing cover crops during fallow periods, will therefore be required to offset some of the exported C, and to increase the C sink activity of the cropland soil.

CRediT author statement

Isobel L. Lloyd: Conceptualisation, Data curation, Formal analysis, Funding acquisition, Investigation, Visualization, Writing – Original draft, Writing – Review and editing. **Ross Morrison:** Conceptualisation, Resources, Software, Supervision, Writing – Review and editing. **Richard P. Grayson:** Conceptualisation, Data curation, Supervision, Writing – Review and editing. **Marcelo V. Galdos:** Conceptualisation, Supervision, Writing – Review and editing. **Pippa J. Chapman:** Conceptualisation, Funding acquisition, Project administration, Supervision, Writing – Review and editing.

Data availability

Data supporting the findings of this study are available at: https://doi.org/10.5285/11f9dd8a-6dac-40e0-b756-05e1f32171f8

Chapter 5 Comparing net ecosystem productivity of neighbouring arable and pasture systems over one year

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Abstract

There is an urgent need to adopt farming systems that sequester carbon (C) in agricultural soils to mitigate climate change, achieve net zero targets and improve soil health. To achieve this, there is a need to understand the C fluxes associated with agricultural management practices, however in the UK measurements of C fluxes from croplands and managed grasslands are lacking. To provide an indication of how C fluxes differ between UK croplands and managed grasslands, we used the eddy covariance technique to measure net ecosystem exchange (NEE) of carbon dioxide (CO_2) and calculated the net ecosystem productivity (NEP) of a cropland and neighbouring cut and grazed pasture in the UK over one year. Over the same period, annual NEP showed the cropland to have a small net C uptake (-26 g C m^{-2}) and the managed grassland to be a source of C (311 g C m^{-2}). For both agri-ecosystems to accumulate C, the amount of C added into the systems must be greater than the C removed as harvested and grazed biomass and the ecosystem respiration. This could be achieved by growing cover crops during fallow periods in croplands, and increasing the addition of organic fertilisers to croplands and managed grasslands.

5.1 Introduction

The use of intensive agricultural management practices, such as frequent deep tillage, high rates of biomass removal, intensive grazing and the conversion of grassland to cropland to increase crop yields and the output of animal-derived products has, and continues to contribute to global soil carbon (C) loss (Sanderman et al., 2017; de Graaff et al., 2019; Schils et al., 2022). This C loss depletes the soil organic carbon (SOC) pool (Eze et al., 2018) and results in an emission of carbon dioxide (CO_2) to the atmosphere (Ussiri and Lal, 2008; Bhattacharyya et al., 2022). Since the early 1800s, an estimated ~133 Pg C has been lost from the top 2 m of global soil, with much of this attributed to agricultural practices, creating a soil C debt (Sanderman et al., 2017). It has been widely recognized that a shift in food and farming systems is required to reverse this soil C debt by sequestering C in agricultural soils, which will only be achieved by adopting appropriate management practices that facilitate soil C sequestration (Padarian et al., 2022; Thamarai et al., 2024).

It is estimated that around 71 % of the UK's land area, equivalent to 17.2 million ha, is currently used for agriculture (DEFRA, 2022a). Approximately 30 % of this (5.16 million ha) is used for crop production and 60 % (10.32 million ha) as managed grassland (DEFRA, 2022a). Winter wheat is one of the most commonly grown crops in the UK, with average yields for the country more than double the global average (Harkness et al., 2020). The winter wheat growing season is typically followed by a fallow period, where no crops are grown, until September or October when the next crop is planted (Adil et al., 2022; Li et al., 2024). During the fallow period the soil is often left bare and multiple tillage events can occur, which can encourage higher yields (Zhong et al., 2023) but also soil erosion and soil C loss as CO_2 (Curtin et al., 2000). Agriculturally managed grasslands are used for livestock grazing and growing vegetation for fodder (Felten et al., 2013; Abraha et al., 2018). Most managed grasslands in the UK are cut for silage and/or grazed by livestock, such as sheep and cattle, to both reduce the cost of feed and maintain pasture height. Croplands have been found to have lower soil C stocks than grasslands (Blair, 2018; Guillaume et al., 2022; Wall et al., 2023a); the 2007 Countryside Survey estimated the average SOC stock in UK arable land to be 43 t ha^{-1}, compared to 61 t ha^{-1} for improved grassland and 62.4 t ha^{-1} for neutral grassland (Countryside Survey, 2007). The conversion of cropland to managed grassland therefore has the potential to increase SOC storage and sequester C back into agricultural soil (Guo and Gifford, 2002; Lugato et al., 2014), contributing to a reduction of the soil C debt, although the effects are likely to be observed over the long term rather than on a short term basis (Gosling et al., 2017).

The wide-scale implementation of agricultural management practices that reduce soil CO_2 emission and facilitate soil C sequestration is heavily reliant on a sound understanding of the extent of C losses and sequestration associated with certain management practices, which in turn requires robust measurements from existing agricultural systems. The extent of these C losses from croplands and managed grasslands in the UK are relatively unknown however. This study aims to contribute to addressing this knowledge gap by determining the impact of land use on the C source or sink strength of agricultural soils in the UK. The objectives were: (i) to quantify CO_2 fluxes, as net ecosystem exchange (NEE), from a neighbouring cropland and managed grassland over one year; and (ii) compare the C source or sink strength of the two fields by calculating net ecosystem productivity (NEP). This research will provide a direct evaluation of the impacts of land use on NEE and NEP; the fact that the cropland and managed grassland are neighbouring sites, and thus have identical climate and soil conditions, means that these factors can be discounted when considering the impacts on NEE and NEP in favour of a focus on land management.

5.2 Methods

5.2.1 Study sites

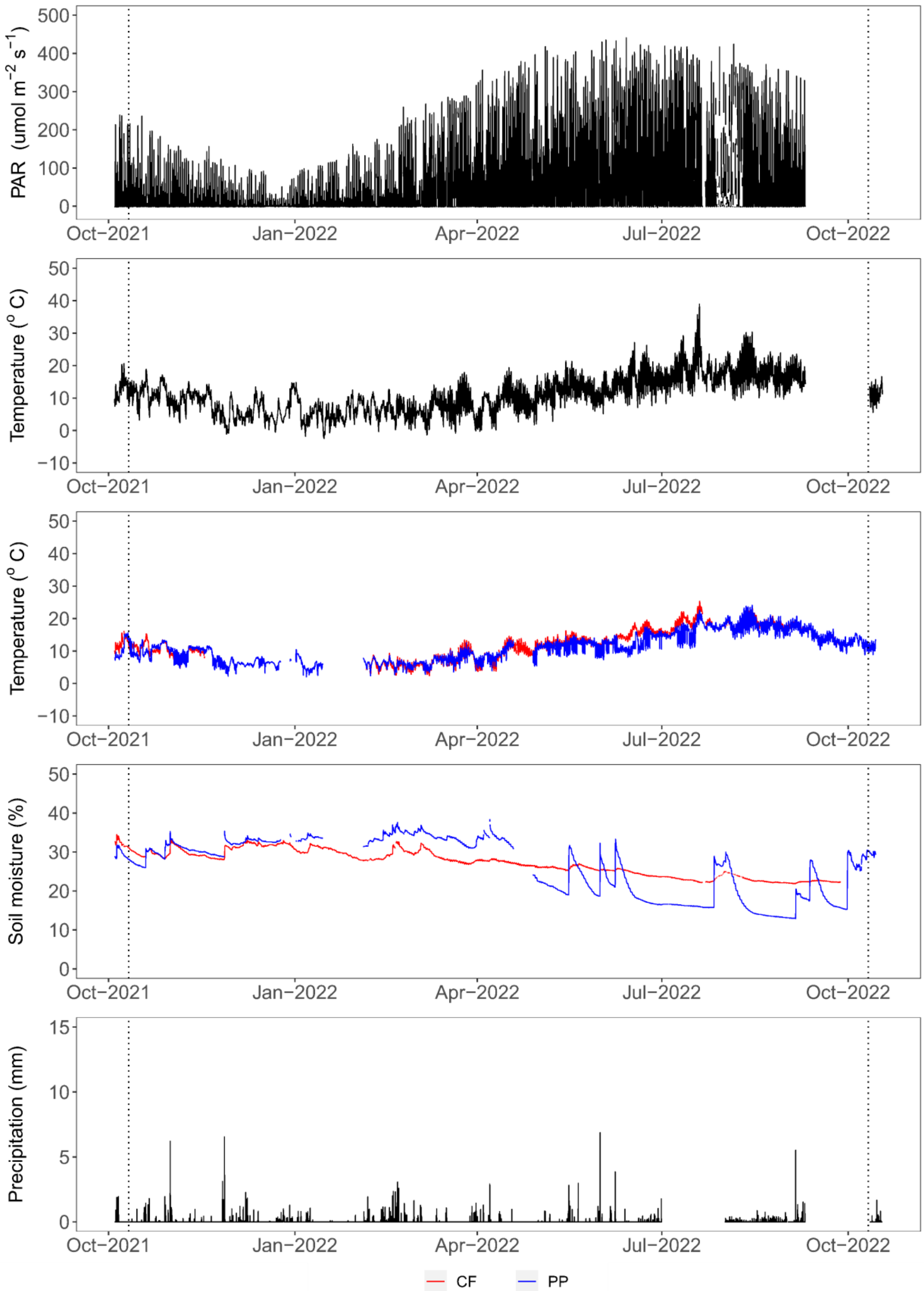
The two sites in this study are neighbouring fields at the University of Leeds Research Farm in Tadcaster, UK, a commercial farm that also supports scientific research. The soil is predominantly a Calcaric Endoleptic Cambisol (IUSS, 2022), 50-90 cm deep, and is underlain by dolomitic limestone (Holden et al., 2019). The farm has a temperate oceanic climate with mild winters and warm summers (Beck et al., 2018); average annual temperature is 9.5 ± 1 °C (Met Office, 2019) and average annual precipitation is 639 ± 142 mm (Met Office, 2006). The crop field (CF) (53°51’56.26”N, 1°19’28.22”W; elevation 49 m; 10.4 ha) has been under continuous arable cultivation with conventional tillage since 1994, with a rotation of mainly winter wheat, and spring or winter barley and oilseed rape as break crops. The permanent pasture (PP) (53°51’58.64”N; 1°19’11.08”W; 46 m elevation; 3.05 ha) has been used to grow grass for silage since 2012. The predominant grass species is perennial ryegrass (*Lolium perenne*). During the spring and summer months PP is periodically grazed by sheep, and typically receives one silage cut via mechanical harvest in the summer. Soil properties of both fields are summarised in Table 5.1.

Eddy covariance (EC) flux towers with associated meteorological and soil sensors were installed in both fields in 2021. Measurements from CF and PP over a twelve month period (11/10/2021-10/10/2022) were used to assess the influence of agricultural land use on soil C fluxes. Over this period, PP was periodically grazed by sheep and silage was harvested in July 2022 with a yield of 80 bales weighing 200 kg dry matter (DM) each. Detailed management information for CF during the one-year measurement period is presented in Table A4.1. Average daily air temperature over the twelve-month period was 11 °C and total precipitation was 481 mm (Figure 5.1).

TABLE 5.1 SOIL INFORMATION FOR EACH FIELD (MEAN ± STANDARD DEVIATION, N=9, FOR TOPSOIL 0-30 CM).

	CF	**PP**
Soil type [a]	Calcaric Endoleptic Cambisol	Calcaric Endoleptic Cambisol
Soil texture [b]	Clayey loam	Clayey loam, sandy loam
Organic matter content (%)	6.7 ± 0.6	7.5 ± 1.7
pH ($CaCl_2$)	6.9 ± 0.2	6.8 ± 0.1
Bulk density (g cm^{-3})	1.3 ± 0.1	1.1 ± 0.1
Total carbon (g kg^{-1})	39.5 ± 9	27.7 ± 7.8
Total organic carbon (g kg^{-1})	22.9 ± 4.9	26.4 ± 6.2
Total nitrogen (g kg^{-1})	2.3 ± 0.6	2.3 ± 0.8
C:N ratio	10:1	11:1
Plant available nitrogen (g kg^{-1})	0.013 ± 0	< 0.01 ± 0

[a] Data obtained from World Reference Base for Soil Resources (IUSS, 2022); [b] UK Soil Observatory (UKRI, 2021)



PAR (umol m^{-2} s^{-1})
500
400
300
200
100
0
Oct−2021
Jan−2022
Apr−2022
Jul−2022
Oct−2022
Temperature (^{o} C)
50
40
30
20
10
0
−10
Soil moisture (%)
Precipitation (mm)
15
5
CF
PP

FIGURE 5.1 (A) PHOTOSYNTHETICALLY ACTIVE RADIATION (PAR), (B) AIR TEMPERATURE, (C) SOIL TEMPERATURE, (D) SOIL MOISTURE AND (E) PRECIPITATION MEASURED DURING THE ONE-YEAR MEASUREMENT PERIOD IN CF AND PP. GAPS INDICATE MISSING DATA AND DOTTED LINES SHOW THE START AND END OF THE MEASUREMENT PERIOD.

5.2.2 Measurement of CO_2 fluxes

The EC technique was used to measure turbulent fluxes of CO_2 (μmol m^{-2} s^{-1}) and sensible and latent heat fluxes (H, LE; W m^{-2}) (Moncrieff et al., 1997; Baldocchi et al., 2003); the EC set up for CF was as described for Mineral Site in Section 3.2.2 (Chapter 3) and was identical in PP. In CF the maximum flux footprint radius was 440 m, with a mean peak distance of 43 m and an average 90 % contribution of 119 m (Figure A4.1). In PP the maximum flux footprint radius was 200 m, with a mean peak distance of 45 m and an average 90 % contribution of 123 m (Figure A4.2). Data collected between 11/10/2021 and 10/10/2022 (365 days) were used to compare C fluxes between CF and PP.

5.2.3 Calculation of CO_2 fluxes

Flux data processing, including the calculation of CO_2 fluxes, quality control and gap-filling was conducted as described in Section 3.2.3 (Chapter 3). During quality control, data were removed when friction velocity (u*; m s^{-1}) < 0.1 in CF and < 0.12 in PP. Gap-filled NEE accounted for 30 % and 37 % of the overall dataset in CF and PP respectively.

Net ecosystem exchange is calculated as the difference between gross primary productivity (GPP) and total ecosystem respiration (TER) as shown in Equation 5.1 (Smith et al., 2010); following gap-filling, NEE was partitioned into GPP and TER (Reichstein et al., 2016). The micrometeorological sign convention is used for NEE, where a positive value indicates the ecosystem is losing C and a negative value indicates the ecosystem is accumulating C (Baldocchi et al., 2003).

$$NEE = TER - GPP \quad \text{(Equation 5.1)}$$

5.2.4 Ancillary measurements

Additional micrometeorological measurements were recorded in CF and PP for the calculation of turbulent fluxes, with the set up as described for Mineral Site in Section 3.2.4 (Chapter 3).

5.2.5 Energy balance

The degree of energy balance closure (EBC) is used to assess the quality of EC data at a given site (Aubinet et al., 2001; Wilson et al., 2002). It compares the sum of H and LE measured by EC, with energy balance terms measured by other means (i.e., net radiation (Rnet) and soil heat flux (G)). Over the one-year measurement period, turbulent fluxes accounted for 71 % and 54 % of the available energy in CF and PP respectively (Figure 5.2). The amount of variance (as measured by R^2 values) for each field are within the typical range of EC measurements (i.e., 0.7-0.9) (Wilson et al., 2002; Foken, 2008; Wagle et al., 2018).

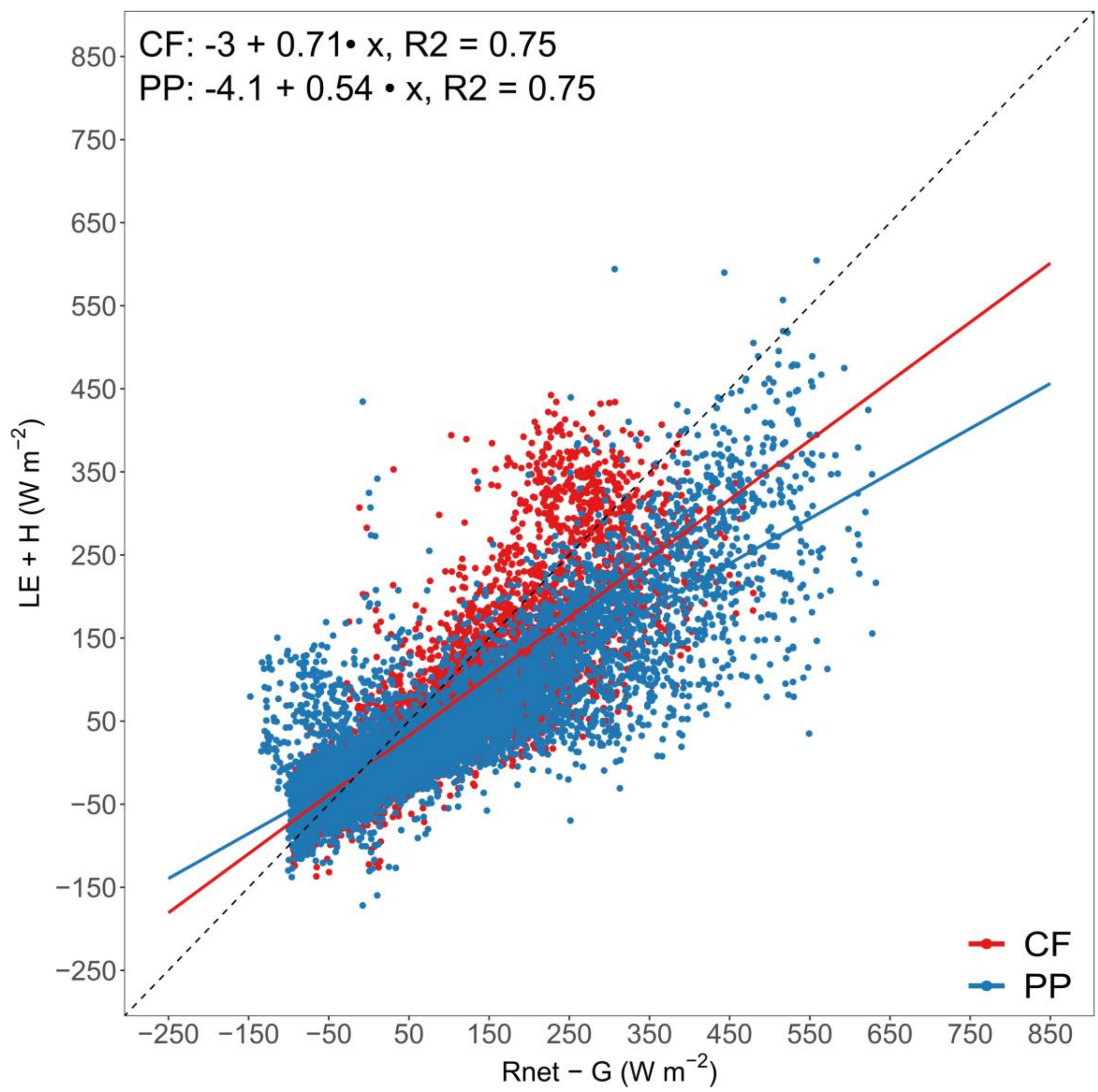


FIGURE 5.2 ENERGY BALANCE IN CROP FIELD (CF) AND PERMANENT PASTURE (PP) OVER THE ONE-YEAR MEASUREMENT PERIOD (11/10/2021-10/10/2022), WHERE H IS SENSIBLE HEAT FLUX, LE IS LATENT HEAT FLUX, RNET IS NET RADIATION AND G IS SOIL HEAT FLUX.

5.2.6 Net ecosystem productivity and crop carbon use efficiency

To estimate the C source or sink strength of the agroecosystems, NEP was calculated according to Equation 5.2. Exports of C (C_H) were in the form of harvested vegetation (i.e., when winter wheat was harvested in CF or when grass was cut for silage in PP) and grazed vegetation (i.e., during grazing events in PP).

In CF, C_H as harvested vegetation (the entire winter wheat crop) was calculated by analyzing the C content of winter wheat samples taken from the field on the day of harvest and scaling this to the reported yield (as in Abraha et al., 2018 and Poyda et al., 2019). In PP, C_H as harvested vegetation was calculated by analyzing the C content of grass samples taken from the field on the day of the silage cut and scaling this to the reported yield; the yield from the silage cut in PP was reported as 80 bales weighing approximately 200 kg DM each. We acknowledge that this carries the assumption that all 80 bales had exactly the same weight, and thus C_H via the silage cut is likely to be slightly under- or over-estimated as a result of this assumption. Conversations with staff at the University of Leeds Research Farm confirmed that the yield measured during this study was similar to those in previous years, however, and so are aligned with what is expected at this site. Exclusion cages were used to determine C_H via sheep grazing in PP. Prior to sheep entering the field, six 1 m^2 exclusion cages were erected to prevent sheep grazing in certain areas. After grazing events, grass samples were taken from inside and outside of the exclusion cages using a 0.5 m^2 quadrat. The samples were dried and the difference in weight between the grass from inside and outside of the exclusion cages was determined as the amount of vegetation removed from the field via grazing. This method was adapted from Hunt et al. (2016) and Laubach et al. (2023); these studies used a plate meter and took grass samples before and after grazing events respectively to determine the quantity of vegetation removed, whereas we used exclusion cages. The C content of the grass was analysed and scaled to the amount of vegetation removed, as done by Hunt et al. (2016) and Laubach et al. (2023). We acknowledge that this method relies on the assumption that the grass grew and was grazed at an even rate across the field, which may result in a slight over- or under-estimation of C_H via grazing livestock as it is unlikely that the grass growth and grazing rates were uniform and that the grass grew in a similar manner in the grazed and ungrazed areas due to different inputs (i.e., the grazed areas received livestock excreta) and the grass in the grazed areas being pulled up by the livestock. As the aim of the study is to assess NEP at the field scale, it is assumed that all C within the removed biomass from CF and PP was converted back to atmospheric CO_2 on leaving the field (Eichelmann et al., 2016; Morrison et al., 2019), via respiration from humans and livestock that consumed the biomass, and humans that consumed the livestock products. We acknowledge that not all of the C exported will be converted back to CO_2 – i.e., some will be returned to the field by grazing animals as dung – and some will be lost as methane (CH_4) via enteric livestock fermentation, however we were unable to account for these fluxes in this study.

Carbon imports (C_I) were in the form of seed (calculated according to Yue et al., 2023), organic amendments and excreta from grazing livestock. We assumed the addition of C via livestock excreta to be 37 % of the C ingested via sheep grazing, as in Skinner (2008; 2013). This assumption was made as more specific information required to calculate C deposited via excreta, such as the non-organic matter digestability and the number of grazing days, as in de la Motte et al. (2016) and Rutledge et al. (2017), was unavailable.

As in Evans et al. (2021), the micrometeorological sign convention is used for NEP, where a positive value indicates C loss and a negative value indicates C gain.

$$NEP = NEE + C_H - C_I \quad \text{(Equation 5.2)}$$

The C use efficiency of harvested material (CUE_h) is a measure of how effectively atmospheric C is converted into new plant material (Chen et al., 2018) and is calculated according to Equation 5.3 (Kim et al., 2022).

$$CUE_h = \frac{C_H}{GPP} \quad \text{(Equation 5.3)}$$

5.3 Results and discussion

5.3.1 Carbon fluxes

Over the one-year measurement period, CF exhibited *in-situ* net CO_2 uptake as NEE (-526 g C m^{-2}) whereas PP had a small CO_2 loss as NEE (37 g C m^{-2}) (Figure 5.3; Figure 5.4; Table 5.2). A diurnal pattern was observed in both fields, with maximum CO_2 uptake occurring in the middle of the day (Figure 5.5).

The difference between the NEE of the two fields can be attributed to the fact that GPP and TER were nearly equal in PP (1394 g C m^{-2} and 1431 g C m^{-2} respectively), whereas in CF GPP (1700 g C m^{-2}) was considerably higher than TER (1175 g C m^{-2}) (Table 5.2; Figure 5.4). Gross primary productivity was considerably higher in CF due to intense CO_2 uptake during the winter wheat growing season, which was triggered by rapid crop growth after nitrogen (N) fertilisation – average daily GPP was 1 g C m^{-2} higher during the 7 days following the second fertilisation event on 21/03/2022 compared to the 7 days before this fertilisation event. A difference in GPP before and after the first fertilisation event was not noticeable, as the crop was not well established at this point in the growing season. Nitrogen fertilisation did not occur in PP and so CO_2 uptake was less intense in PP. Although similar between the two sites, TER was 22 % higher in PP than CF. This can be attributed to the fact that PP had a higher soil organic matter (SOM) and SOC content (7.5 % and 26.4 g kg^{-1} compared to 6.7 % and 22.9 g kg^{-1} in CF) and also had more continuous vegetation cover, and thus more living roots, during the measurement period. The decomposition of SOM and utilization of root exudates as a substrate by soil microorganisms would have increased soil microbial activity and respiration (Kruse et al., 2013; Kotroczo et al., 2023), which most likely explains the higher TER from PP. In addition, grazing livestock were sometimes present in the field, with EC also capturing the CO_2 emitted via livestock respiration (Senapati et al., 2014; Rutledge et al., 2015), although the grazing intensity was low and so livestock respiration will be a small proportion of the CO_2 emission.

In CF, the magnitude of diurnal NEE was highest between April and June when winter wheat was growing vigorously (Figure 5.5). The CO_2 uptake in PP was also greatest during this time, which was when the field was not grazed, and the grass was left to grow before the harvest event in July (Figure 5.5). Similar to our findings, Skinner (2008) and Myrgiotis et al. (2022) observed that CO_2 uptake by managed grasslands was greatest in spring. The overall CO_2 uptake and magnitude of diurnal NEE decreased considerably following harvest events in both fields – August in CF and July in PP (Figure 5.3; Figure 5.5). Cutting events have been shown to decrease CO_2 uptake as leaf area index, and thus the ability for plants to photosynthesise, is reduced (Klumpp et al., 2004; Prescher et al., 2010; Zeeman et al., 2010; Jerome et al., 2012). Following a grass harvest event, the CO_2 uptake capacity of the field typically increases as vegetation re-establishes and photosynthesis resumes (Aires et al., 2008; Wall et al., 2019; 2020b), as observed in our study (Figure 5.3; Figure 5.5). Cardenas et al. (2022) report the only measured values of annual NEE from a cut and grazed grassland in the UK. Whilst C_H and C_I are not reported for the calculation of NEP, the CO_2-equivalent values of NEE are reported. The NEE of PP (1.4 t CO_2-equivalent ha^{-1}) is well within the range reported by Cardenas et al. (2022) which ranged from -5.4 t CO_2-equivalent ha^{-1} to 6.17 t CO_2-equivalent ha^{-1}. This range of values can be attributed to the difference in livestock stocking density, number of cuts and amount of harvested material between the sites and between the study years in Cardenas et al. (2022), and highlights the need for more measurements to account for inter-annual variability.

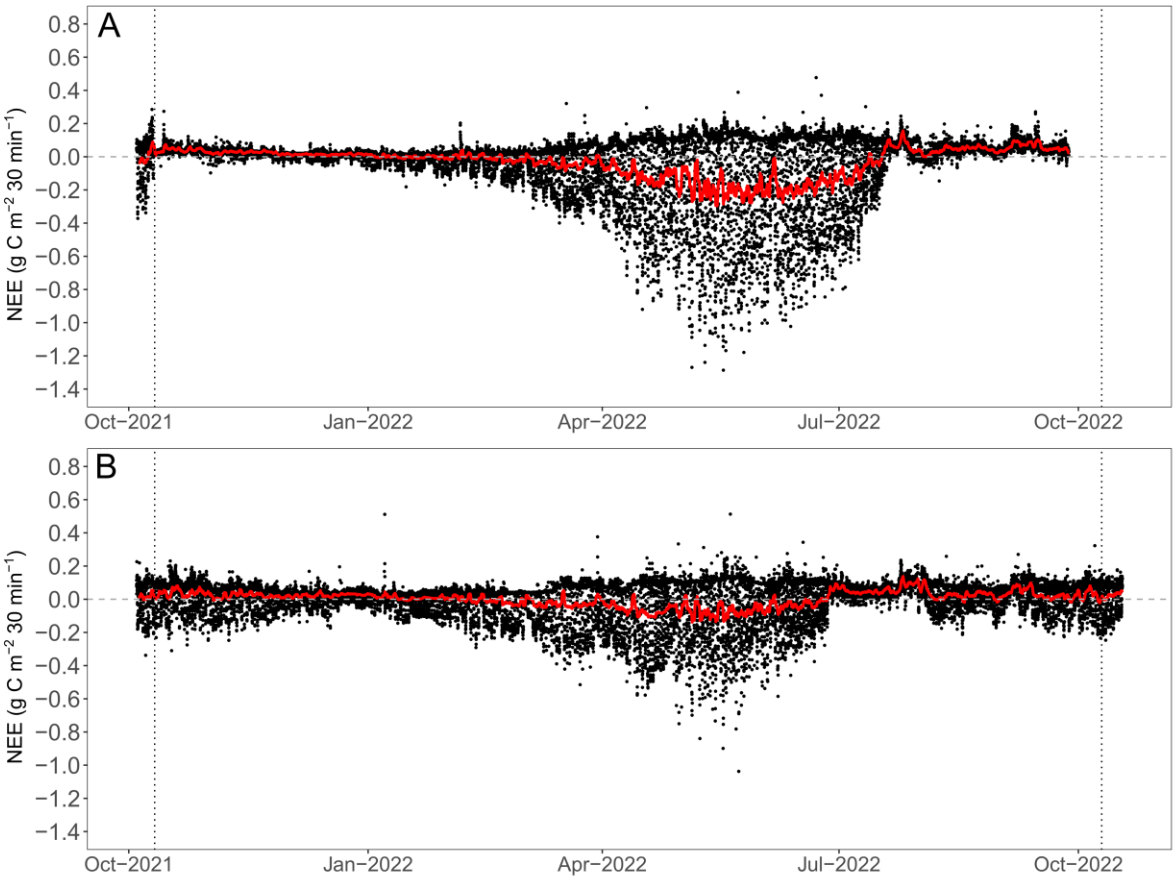
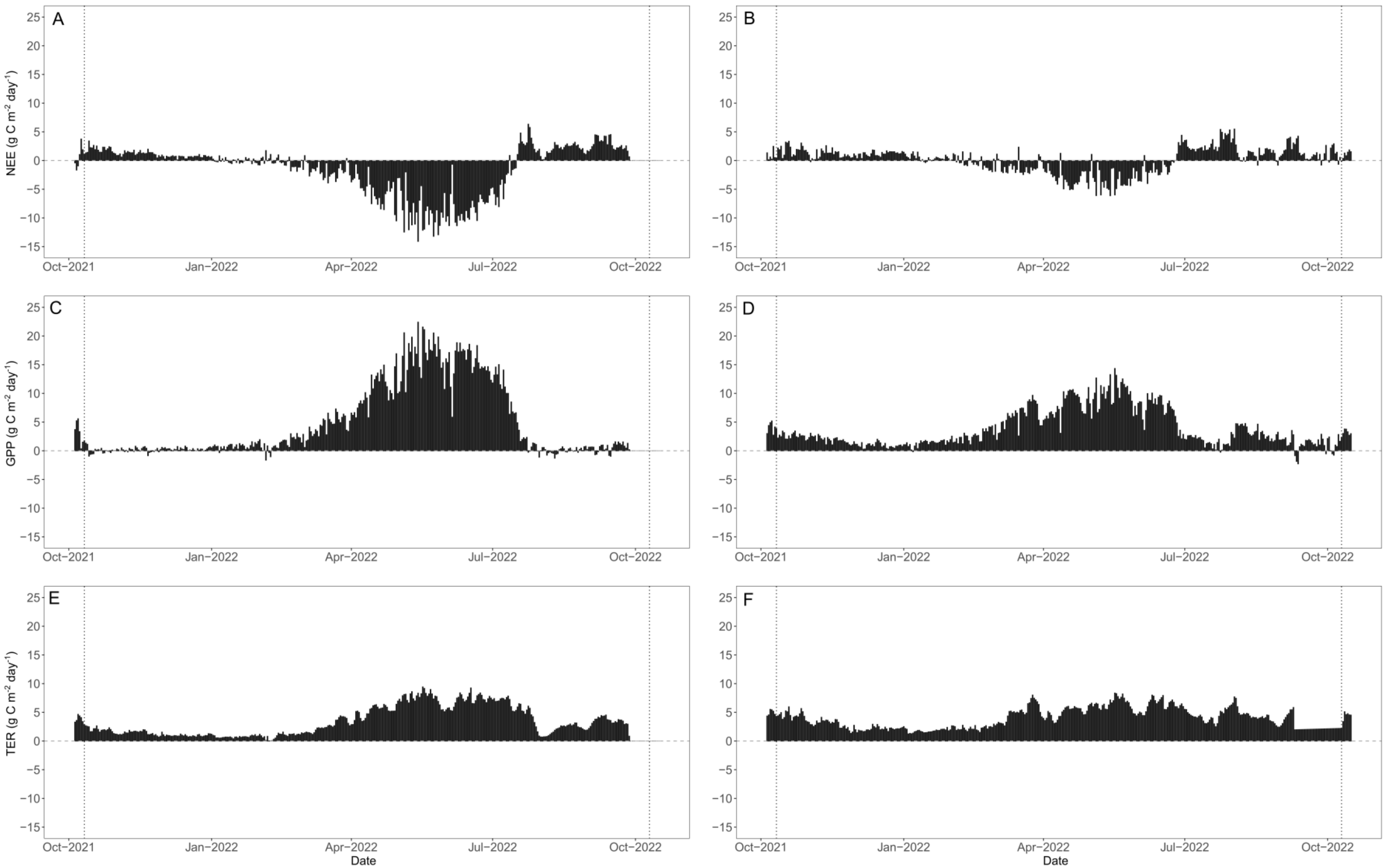


FIGURE 5.3 30-MINUTE FLUXES OF NEE IN (A) CF AND (B) PP. DOTTED LINES INDICATE THE START AND END OF THE ONE-YEAR MEASUREMENT PERIOD. THE RED LINE INDICATES THE ROLLING DAILY MEAN.



A
B
C
D
E
F
NEE (g C m-2 day-1)
GPP (g C m-2 day-1)
TER (g C m-2 day-1)
25
20
15
10
5
0
−5
−10
−15
Oct−2021
Jan−2022
Apr−2022
Jul−2022
Oct−2022
Date

FIGURE 5.4 DAILY NEE, GPP AND TER IN CF (A, C, E) AND PP (B, D, F) OVER THE ONE-YEAR MEASUREMENT PERIOD (11/10/2021-10/10/2022). DOTTED LINES INDICATE THE START AND END OF THE ONE-YEAR MEASUREMENT PERIOD.

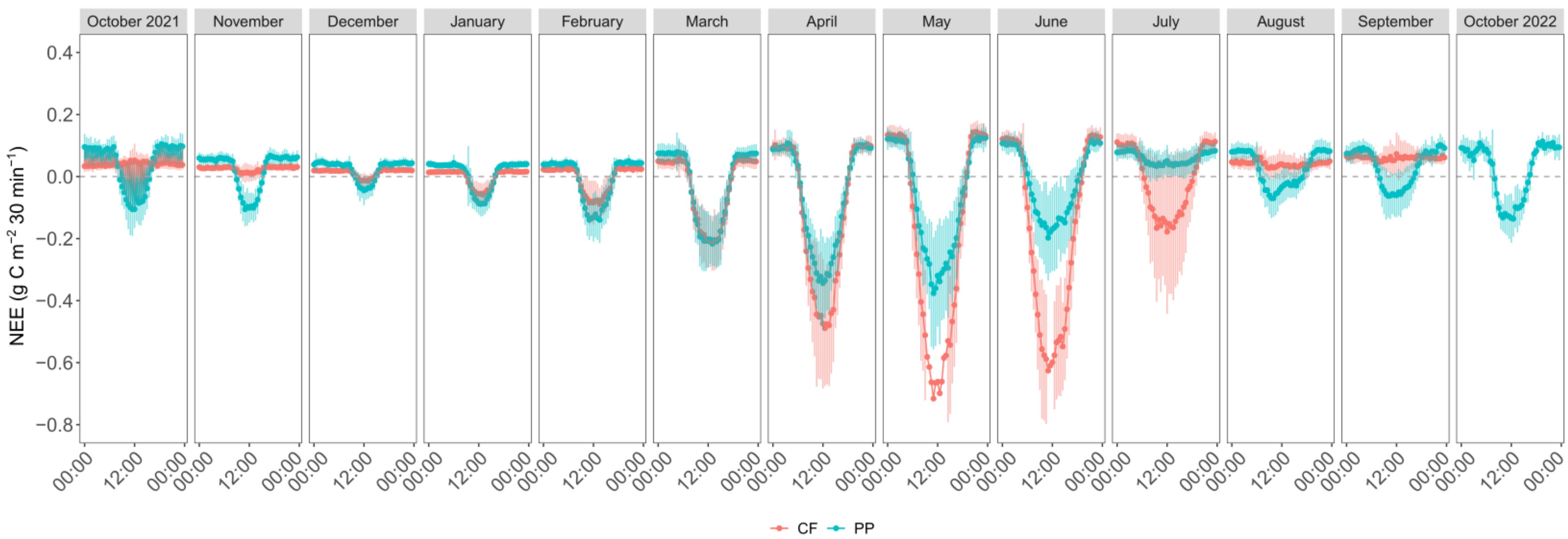


FIGURE 5.5 MEAN DIURNAL NEE IN CF AND PP OVER THE ONE-YEAR MEASUREMENT PERIOD (11/10/2021-10/10/2022). ERROR BARS REPRESENT STANDARD ERROR OF THE MEAN.

TABLE 5.2 CARBON BUDGET IN CF AND PP OVER THE ONE-YEAR MEASUREMENT PERIOD (11/10/2012-10/10/2022; 365 DAYS) ± ROOT SUM SQUARED (ASIDE FROM C_H WHERE ± REPRESENTS STANDARD DEVIATION OF BIOMASS CARBON CONTENT UPSCALED TO REPORTED BIOMASS OFFTAKE, AND C_I WHERE ± REPRESENTS STANDARD DEVIATION OF SPIG SLURRY CARBON CONTENT). THE MICROMETEOROLOGICAL SIGN CONVENTION IS USED FOR NEE AND NEP WHERE POSITIVE VALUES INDICATE C LOSS AND NEGATIVE VALUES INDICATE C GAIN.

	CF	**PP**
NEP (g C m^{-2})	-26 ± 132	311 ± 215
NEP (t CO_2-equivalent ha^{-1})	-1 ± 5	11 ± 8
NEE (g C m^{-2})	-526 ± 85	37 ± 42
NEE (t CO_2-equivalent ha^{-1})	-19 ± 3	1 ± 2
TER (g C m^{-2})	1175 ± 79	1431 ± 82
GPP (g C m^{-2})	1700 ± 150	1394 ± 96
CUE_h (g C g C^{-1})	0.36	0.22
C_H (g C m^{-2})	616 ± 6	313 ± 173
C_I (g C m^{-2})	116 ± 41	39 ± 0

5.3.2 Net ecosystem productivity

Over the one-year measurement period, C was being lost from the managed grassland, with PP having a positive cumulative NEP (311 g C m^{-2}; 11 t CO_2-equivalent ha^{-1}), whereas CF had a small C uptake (-26 g C m^{-2}; -1 t CO_2-equivalent ha^{-1}) (Table 5.2; Figure 5.6; Figure 5.7).

An overall loss of C from agricultural systems has been reported by multiple studies, as most of the C fixed by vegetation during photosynthesis is removed by mechanical harvest or grazing events or via respiration from grazing animals (Skinner, 2008; Chang et al., 2015; Carozzi et al., 2022; Niu et al., 2022). The managed grassland lost C overall as C_H was considerably higher than NEE and C_I was small in comparison, whereas the cropland was C neutral as the sum of its NEE and C_I was near equal to C_H (Table 5.2). Much of the literature suggests that a conversion of cropland to managed grassland would reduce C losses (e.g., Guo and Gifford, 2002; Lugato et al., 2014), however our results show that this is not necessarily always the case, as PP had a greater C loss than CF. An emission of C from managed grasslands has the potential to offset the C sink behavior of other ecosystems (Chang et al., 2021). The C_I to CF (116 g C m^{-2}) was in the form of seed (8 g C m^{-2}) and organic fertiliser (108 g C m^{-2}) which is more than double the C_I to PP (39 g C m^{-2}) which was added via excreta from grazing livestock. It has been proposed that C_I to croplands is generally lower than to grasslands (Janzen et al., 2022; De Rosa et al., 2023), however we show here that this is not always the case as C_I is dependent on orgnic fertiliser use and livestock grazing intensity. The C_H from CF (616 g C m^{-2}) was nearly twice that from PP (313 g C m^{-2}). For CF and PP to behave as C sinks, the amount of C added must be greater than all other losses of C as exported biomass and TER, as highlighted by Cates and Jackson (2019). In croplands, C_I can be increased by adding organic amendments (Lal, 2016) and reducing the length of time that soil is bare for, which could involve growing cover crops during fallow periods (Steenwerth and Belina, 2008; Ruis et al., 2017; Jian et al., 2020). In managed grasslands, this may be achieved by increasing grassland productivity to increase C_I to the soil through plant roots, which can be achieved by increasing fertilisation, seeding with high-yielding species and increasing species diversity (Cong et al., 2014; Moxley et al., 2014; Rutledge et al., 2017).

The NEP measured in PP is higher than many of the NEP values reported across much of the the literature for cut and grazed grasslands in temperate climates; annual NEP ranges from -249.4 g C m^{-2} (Laubach et al., 2023) to 337 g C m^{-2} (Skinner, 2013) (Table A4.2). The C_H from PP is close to the average of that reported in the literature (298 g C m^{-2}), however NEE is positive, and so, as most of the NEE values reported by the literature are negative, the NEP of PP is relatively high. There are comparatively fewer published studies to compare the results from CF with; only one published study could be found containing annual NEE, C_I and C_H for winter wheat followed by a fallow period in a temperate climate (Poyda et al., 2019). Annual NEP reported by Poyda et al. (2019) ranges from -328.9 g C m^{-2} to 382.2 g C

m^{-2} (Table A4.2); although Poyda et al. (2019) only harvested grain, and grain and straw are harvested in this study, the NEP of CF fits well within the reported range.

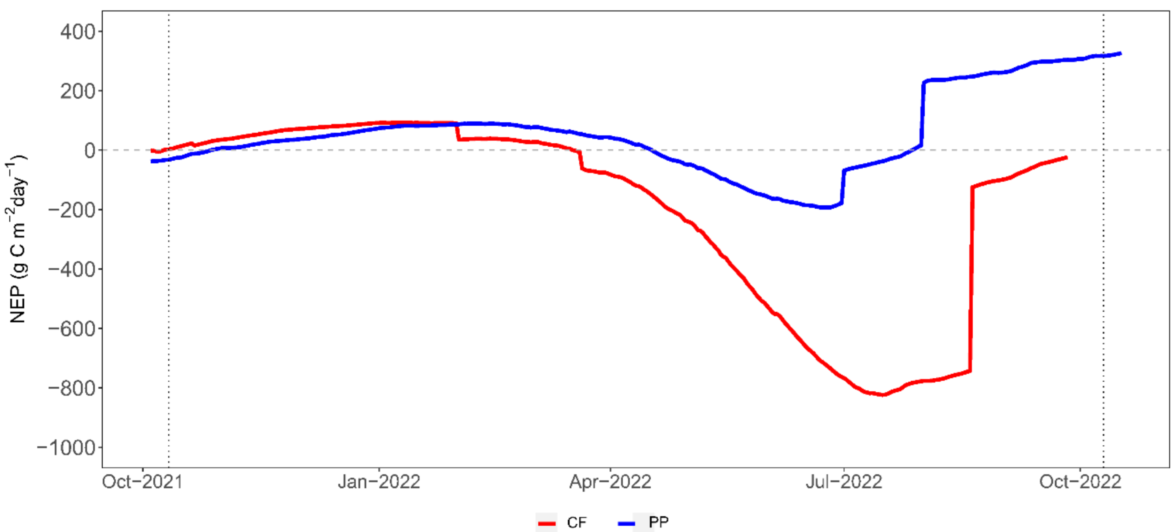


FIGURE 5.6 CUMULATIVE DAILY NEP IN CF AND PP. DOTTED LINES INDICATE THE START AND END OF THE ONE-YEAR MEASUREMENT PERIOD.

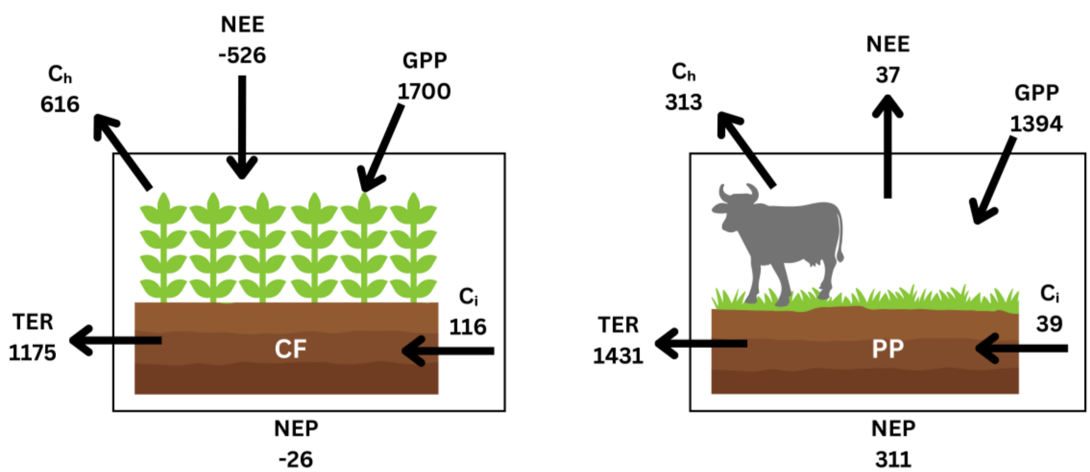


FIGURE 5.7 ANNUAL CARBON BALANCE OF CROP FIELD (CF) AND PERMANENT PASTURE (PP). NOTE THAT ALL UNITS ARE G C M^{-2}.

5.3.3 Limitations and implications for research

The results of this study highlight the strong influence of agricultural management practices on the potential of an agroecosystem to behave as a C sink or source. Over a one-year period, the cut and grazed pasture lost C, whereas the neighbouring cropland was C neutral, thus contradicting the concept that the conversion of cropland to grassland would increase C sequestration and reduce C loss. Only one year of data for two fields in the UK are presented here, however, so we are unable to identify how NEP varies on an inter-annual basis, in response to varying climate conditions and management practices. Similarly, soil type has been observed to affect NEP, with more clayey soils observed to have lower CO_2 emissions (Maia et al., 2019; Prout et al., 2022), however CF and PP had the same soil type, so it is therefore not possible to determine the effects of soil type on NEP in this study. There is a clear need for more data from agricultural sites across the UK, with different soil types, climate conditions, and management practices to identify the influences on annual NEP in croplands and managed grasslands over multiple years. Agricultural grasslands are managed in a wide range of ways, from intensive to extensive management, and so it is crucial that all meta-data on the management and environmental conditions of the site is reported to compare and interpret results (Gosling et al., 2017). Non-CO_2 greenhouse gas emissions must also be considered when aiming to reduce the negative impacts of agricultural management on the environment – these being methane (CH_4) and nitrous oxide (N_2O) – and could be considered by using greenhouse gas flux chambers in conjunction with EC.

The presentation of our data provides the first indication of annual NEP from a neighbouring cropland and managed grassland in the UK, however more data measured over multiple years across the country is required to compile a dataset that is representative of the UK. This information will be essential for policymakers to make evidence-based decisions and recommend the most suitable management practices for farmers to increase soil C sink activity. When measuring soil C fluxes on an annual basis, the start and end dates of the measurement period will determine the NEP values reported. In this study, for example, the 365-day measurement period (11/10/2021-10/10/2022) encompasses the winter wheat growing season and the fallow period following winter wheat harvest in CF. If the measurement period were to start one week earlier it would have included the harvest of the preceeding maize crop, and so C_H from CF would be considerably larger than reported. This further highlights the importance of measuring C fluxes over multiple years to account for variations in NEP as a result of the different management practices implemented on annual timescales.

There are greater uncertainties associated with the calculation of C_H and C_I in managed grasslands compared to croplands, and so we have greater confidence in the C_H and C_I values reported for CF than PP. Unlike C_H via harvest, where the C content of the biomass can be upscaled to the reported yield, C_H via grazing is difficult to ascertain and multiple methodologies have been suggested to derive this value. In this study, we compared grass samples from inside and outside exclusion cages in PP to determine the amount ingested by grazing livestock and multiplied this by the C content of the grass. The exclusion cage method assumes the rate of grass growth and grazing to be uniform across the field, however this is unlikely to have been the case. Furthermore, the nature of the grass growth inside and outside of the exclusion cages is likely to have been different due to these differences in management; the grass outside of the exclusion cages will have received excreta from the grazing livestock and the motion of the animals pulling up the grass to consume are likely to have stimulated growth compared to the non-grazed grass. Due to these assumptions and differences, C_H via grazing will be slightly over- or under-estimated. In addition, we assumed the proportion of C returned to the field as livestock excreta to be 37 % as in Skinner (2008; 2013). This proportion is also highly variable throughout the literature (Section 2.4.2), meaning that C_I to PP in our study may also be somewhat over- or under-estimated provided this assumption has been used. The calculation of C_H from both CF and PP is based on yield reported by the farmer, which again has some associated error. The 80 bales of silage exported from PP, for example, are unlikely to have weighed exactly 200 kg DM each, which introduces further uncertainty to the C_H values reported.

5.4 Conclusions

An understanding of the C loss or gain associated with agricultural production systems in the UK will be critical for a transition to more sustainable food systems which reduce C loss and ideally facilitate soil C sequestration. This information will be critical for UK policymakers to support farmers in making the most appropriate land management decisions to reduce their negative environmental impacts whilst still achieving good yields and income. This study measured and compared the annual NEP of a cropland and cut and grazed grassland. Over the one-year study period, the cropland, although close to C neutral, had a small net C uptake (-26 g C m^{-2}), whereas the managed grassland was a source of C (311 g C m^{-2}). An increase in C_I would increase the C sink capacity of the cropland, and would offset some of the C losses from the cut and grazed grassland. This could be achieved by increasing inputs of organic fertiliser, by returning crop residues to the field and/or by growing cover crops during fallow periods. Whilst the results presented here are the first NEP data of a cropland and managed grassland in the UK on mineral soil, they are not sufficient to base UK-wide conclusions on. More data is required to understand the inter-annual variability of NEP as a result of management practices in the UK, and the influence of soil and climate conditions.

CRediT author statement

Isobel L. Lloyd: Conceptualisation, Data curation, Formal analysis, Funding acquisition, Investigation, Visualization, Writing – Original draft, Writing – Review and editing. **Ross Morrison:** Conceptualisation, Resources, Software, Supervision, Writing – Review and editing. **Richard P. Grayson:** Conceptualisation, Data curation, Supervision, Writing – Review and editing. **Marcelo V. Galdos:** Conceptualisation, Supervision, Writing – Review and editing. **Pippa J. Chapman:** Conceptualisation, Funding acquisition, Project administration, Supervision, Writing – Review and editing.

Data availability

Data supporting the findings of this study are available at: https://doi.org/10.5285/11f9dd8a-6dac-40e0-b756-05e1f32171f805e1f32171f8 and https://doi.org/10.5285/c94b7b70-ab7e-4415-9b99-7f4a10e97c1c

Chapter 6 Nitrous oxide and methane fluxes from plasma-treated pig slurry applied to winter wheat

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Abstract

The use of livestock waste as an organic fertiliser releases significant greenhouse gas emissions, exacerbating climate change. Innovative fertiliser management practices, such as treating slurry with plasma induction, have the potential to reduce losses of carbon and nitrogen to the environment. The existing research on the effectiveness of plasma-treated slurry at reducing nitrous oxide (N_2O) and methane (CH_4) emissions, however, is not comprehensive, although must be understood if this technology is to be utilised on a large scale. A randomised block experiment was conducted to measure soil fluxes of N_2O and CH_4 from winter wheat every two hours over an 83-day period using automated chambers. Three treatments receiving a similar amount of plant-available N were used: (1) inorganic fertiliser (IF); (2) pig slurry combined with inorganic fertiliser (UPS); (3) plasma-treated pig slurry combined with inorganic fertiliser (TPS). Cumulative N_2O fluxes from TPS (1.14 g N m^{-2}) were greater than those from UPS (0.32 g N m^{-2}) and IF (0.13 g N m^{-2}). A diurnal pattern in N_2O fluxes was observed towards the end of the experiment for all treatments, and was driven by increases in water-filled pore space and photosynthetically active radiation and decreases in air temperature. Cumulative CH_4 fluxes from UPS (3.2 g C m^{-2}) were considerably greater than those from IF (-1.4 g C m^{-2}) and TPS (-1.4 g C m^{-2}). The greenhouse gas intensity of TPS (0.2 g CO_2-eq kg grain^{-1}) was over twice that of UPS (0.07 g CO_2-eq kg grain^{-1}) and around six times that of IF (0.03 g CO_2-eq kg grain^{-1}). Although treating pig slurry with plasma induction considerably reduced CH_4 fluxes from soil, it increased N_2O emissions, resulting in higher non-CO_2 emissions from this treatment. Life-cycle analysis will be

required to evaluate whether the upstream manufacturing and transport emissions associated with inorganic fertiliser usage are outweighed by the emissions observed following the application of treated pig slurry to soil.

6.1 Introduction

Nitrogen (N) is one of the most limiting nutrients for crop growth in agricultural soils, so organic (i.e., animal manure and slurry) and inorganic (i.e., synthetic) N fertilisers are applied to provide a supply of N to support crop growth and achieve high yields (Lu et al., 2021). Organic fertilisers also provide a source of other plant nutrients, enhance soil carbon (C) content, and are increasingly being seen as part of an on-farm circular economy within the agricultural sector. The use of fertilisers in agriculture results in significant emissions of greenhouse gases (GHGs) to the atmosphere. Agriculture is responsible for 13 % global carbon dioxide (CO_2) emissions, 50 % global methane (CH_4) emissions, and 60 % global nitrous oxide (N_2O) emissions (Macharia et al., 2020). Nitrous oxide and CH_4 are of particular concern, as they have global warming potentials 273 and 27.9 times greater than CO_2 respectively (Smith et al., 2021) and continue to exacerbate climate change (Mikhaylov et al., 2020). Agricultural N_2O emissions primarily originate from the use of inorganic and organic N fertilisers, which has increased markedly over the last 60 years (Rudaz et al., 1999; Cameron et al., 2013; Lu et al., 2021). Between 2016 and 2019, animal farming in the European Union produced more than 1.4 billion tonnes of manure annually, and over 90 % of this was directly re-applied to soils (Koninger et al., 2021). Fertiliser application, particularly organic fertiliser, can also increase CH_4 emissions; CH_4 is often produced during organic fertiliser storage, as the C supply and storage conditions facilitate methanogenesis, dissolving CH_4 into the fertiliser and releasing it upon application to soil (Rochette and Cote, 2000; Bastami et al., 2016).

There is an urgent need to minimise the negative impacts of agriculture on the environment, with the aim to achieve net zero GHG emissions becoming increasingly critical (Sakrabani et al., 2023). Despite the implementation of strategies which aim to reduce environmental N pollution (i.e., Nitrate Vulnerable Zones (UK Government, 2021b) and 4R Nutrient

Stewardship – right source, rate, time and place (Nutrient Stewardship, 2017)), GHG emissions from agriculture, particularly N_2O, remain high (Tian et al., 2020). To reduce GHG emissions from fertiliser use, crop N use efficiency (NUE) – the efficiency at which applied N is assimilated by plants (Sharma and Bali, 2018) – must be improved. Given the push to increase the use of livestock waste as fertiliser and build soil C, a range of practices and innovative technologies are promoted to reduce GHG emissions from fertiliser use and improve NUE. One such example of this is the treatment of organic fertilisers, such as pig slurry, with plasma induction. This treatment primarily aims to reduce losses of the non-GHG ammonia (NH_3) by ionising air to form reactive nitrogen gas which is absorbed into the slurry, creating an N-rich slurry (Nyang'au et al., 2024). This process lowers the pH of the slurry and reduces the potential for NH_3 emissions (Nyang'au et al., 2024). An increase in the N content of the plasma-treated slurry means the product has the potential to replace synthetic inorganic fertiliser and has been shown to increase yields compared to untreated slurry (Mousavi et al., 2022; Cottis et al., 2023), as well as reducing both CH_4 and NH_3 emissions during storage (Graves et al., 2018). Whether the beneficial gains of increasing the amount of inorganic N available for immediate plant uptake are counterbalanced by other N losses upon application to the soil, such as N_2O to the atmosphere, however, are unknown. Numerous studies have investigated the impacts of fertiliser application on GHG fluxes, mainly N_2O, from agricultural soils (Inselsbacher et al., 2010; Mateo-Marin et al., 2020; Adelekun et al., 2021). The overarching consensus is that soils amended with organic fertiliser have higher N_2O and CH_4 emissions than those amended with inorganic fertiliser (Thangarajan et al., 2013; Walling and Vaneeckhaute, 2020; He et al., 2023). The effects of using plasma-treated slurry as an organic fertiliser on soil N_2O and CH_4 emissions is relatively unknown, however, and most of the existing research on plasma-treated organic waste has focused on the effects of plasma-treated cattle slurry on crop yield, soil biota and NH_3 emissions (Mousavi et al., 2022; 2023; Cottis et al., 2023). If plasma-treated pig slurry is to become a potential solution to reduce non-CO_2 GHG emissions, it will be necessary to explore the extent to which it can achieve this relative to non-treated pig slurry and inorganic fertiliser.

The aim of this study was therefore to determine the effects of treating pig slurry with plasma induction on N_2O and CH_4 fluxes and crop yield when applied as an organic fertiliser. This was achieved by carrying out the following objectives: (1) measure and analyse the response of N_2O and CH_4 fluxes to the application of inorganic and organic fertilisers, including plasma-treated and non-treated pig slurry; (2) compare winter wheat yield and its GHG intensity as a result of the fertiliser treatment used; and (3) quantify and explain the controls on the diurnal variation of N_2O and CH_4 fluxes during the main winter wheat growth phase. Treating pig slurry with plasma induction has been proven to reduce NH_3 emissions as a result of acidification, creating an N-enriched product which has a higher content of inorganic N. Furthermore, a reduction in the pH of the slurry may prevent methanogenesis and thus CH_4 formation during slurry storage, and thus potentially following application. Therefore, our first hypothesis is that non-CO_2 GHG emissions will be lower from the plasma-treated pig slurry compared to the non-treated pig slurry. Based on the existing research on GHG emissions and the impact of fertiliser type, our second hypothesis is that N_2O and CH_4 emissions will be higher from winter wheat treated with organic fertilisers (i.e., plasma-treated and non-treated pig slurry treatments) compared to inorganic fertiliser, as a result of increasing C and N availability to soil microorgansims, thus increasing their activity.

6.2 Materials and methods

6.2.1 Field site and experimental design

The University of Leeds Research Farm is a commercial mixed arable and livestock farm near Tadcaster, UK. It has a temperate climate with mild winters and warm summers (Beck et al., 2018). The soil is a well-drained, loamy calcareous Cambisol (Cranfield University, 2018), with a depth of 0.5-0.9 m (Holden et al., 2019). Soil properties of the study site are summarised in Table A5.1. Between 1992 and 2021 mean annual temperature ± standard deviation was 9.5 ± 1 °C (Met Office, 2019) and mean annual precipitation was 639 ± 142 mm (Met Office, 2006). During the study period (20/03/2022-13/06/2022), drought conditions and record maximum temperatures were experienced in the UK (Turner, 2022) (Figure A5.1); total precipitation was 112 mm and average daily air temperature was 10.7 °C (527 mm lower and 1.2 °C higher than the annual average). On 21/10/2021, winter wheat

(WW) (*Triticum aestivum*), Extase variety, was sown at a density of 440 seeds m^{-2} in an arable field (53º51'56.26"N 1º19'28.22"W; elevation 49 m; 10.4 ha). In February 2022, prior to the application of any fertiliser, a randomised block experiment was set up consisting of nine plots (2 x 0.5 m) and neighbouring areas for the placement of nine GHG measurement chambers. Circular collars (0.5 m diameter) were inserted into the soil to a depth of 0.1 m and Eosense eosAC-LT chambers (Eosense, Canada) with an internal volume of 0.072 m^3 were attached one month prior to fertiliser application. This allowed the soil to return to steady state conditions prior to the commencement of GHG measurements (Charteris et al., 2020).

Three fertiliser treatments (each with three replicates) were compared (Table A5.2): three applications of inorganic fertiliser (IF); two applications of pig slurry followed by two applications of inorganic fertiliser (UPS); and two applications of plasma-treated pig slurry followed by two applications of inorganic fertiliser (TPS). Each plot and its neighbouring GHG chamber received the same fertiliser treatment; fertiliser was applied to the plots and chambers in split applications, the rates based on recommendations from MANNER-NPK (ADAS, 2013). All fertiliser treatments were applied by hand; granular fertiliser was evenly distributed onto the soil surface and slurry was applied with a watering can, taking care to apply slurry only to the soil surface and not on WW leaves. The treatments were applied with the intention of all plots receiving a total of 220 kg available N ha^{-1}. Following analysis of the fertilisers, it was confirmed that the IF and UPS treatments received a total of 220 kg available N ha^{-1}, whereas the TPS treatment received 253 kg available N ha^{-1}. More detail on application types, rates and dates are shown in Table A5.2. For UPS and TPS, pig slurry was collected from an on-farm indoor pig facility and for TPS the pig slurry was then treated using plasma induction. The plasma treatment process uses electricity to ionise air and create nitrogen oxide gas, which combines with free NH_3 to form involatile ammonium nitrate, thus reducing NH_3 emissions and increasing the amount of inorganic N potentially available for immediate plant uptake upon application to the crop (Graves et al., 2018; Nyang'au et al., 2024). This may in turn reduce the amount of N available for conversion to N_2O, thus reducing N_2O emissions, however this is highly dependent on the environmental conditions and the crop type and growth stage. The plasma induction process also prevents the conditions which facilitate methanogenesis and reduces the pH of the slurry, reducing CH_4 production in storage and thus CH_4 emissions upon application (Tooth et al., 2021). The nutrient composition of the organic fertiliser treatments is shown in Table 6.1. The IF treatment received no inputs of phosphorous or potassium, whereas the UPS and TPS treatments did (Table 6.1), however this is unlikely to have limited the growth of wheat as the soil has a phosphorus index of 3 in the top 10 cm, and thus is not limited in the soil (Table A5.1).

TABLE 6.1 NUTRIENT COMPOSITION OF EACH OF THE APPLIED ORGANIC FERTILISER TREATMENTS (UPS = UNTREATED PIG SLURRY, TPS = TREATED PIG SLURRY); [A] ANALYSIS OF TREATMENTS USED IN THE EXPERIMENT, [B] AVERAGE OF ANALYSIS OF OTHER UPS (N=3) AND TPS (N=3) SAMPLES CONDUCTED PRIOR TO THE EXPERIMENT.

	UPS (Application 1)	**UPS (Application 2)**	**TPS (Application 1)**	**TPS (Application 2)**
Dry matter (kg DM t^{-1}) [a]	54.8	89.6	19.6	20.9
pH [a]	7.09	7.15	4.92	4.97
Total Kjeldahl nitrogen (% w/w) [a]	0.34	0.39	0.3	0.29
Ammonium-nitrogen (mg kg^{-1}) [a]	2055	2207	1488	1443
Nitrate-nitrogen (mg kg^{-1}) [a]	68	21.9	1108	1331
Total phosphorus (mg kg^{-1}) [a]	932	1630	499	572
Total potassium (mg kg^{-1}) [a]	1940	2096	1716	1969
Total nitrogen (mg kg^{-1}) [a]	3470	3920	4110	4230
Organic matter (%) [b]	1.14 ± 0.9		1.1 ± 0.6	
Total organic carbon (%) [b]	0.66 ± 0.5		0.64 ± 0.4	

Soil moisture and temperature were measured in each plot at a depth of 0.05 m using TEROS 11 moisture and temperature sensors (METER Group Inc., USA), with measurements logged at 15-minute intervals. Soil moisture and bulk density were used to calculate water-filled pore space (WFPS) according to Equation 6.1, adapted from De and Toor (2015):

$$WFPS(\%) = \left((\theta g \times Bd) \div \left(1 - (Bd - Pd)\right)\right) \times 100 \quad \text{(Equation 6.1)}$$

where θg is soil moisture (%), Bd is bulk density (g cm^{-3}) and Pd is particle density (g cm^{-3}) (assumed to be 2.65 g cm^{-3} for arable soils (Schjonning et al., 2017)).

6.2.2 GHG sampling and crop yield measurements

Fluxes of N_2O, CH_4 and CO_2 were measured from each chamber every 120-minutes between 20/03/2022 and 13/06/2022 using a Picarro G2508 GHG analyser (Picarro, USA), resulting in 9288 discrete sampling points over 83-days. The analyser uses cavity ring-down spectroscopy to measure GHG fluxes; the measurement range of N_2O is 0.3-200 ppm, of CH_4 is 1.5-12 ppm and of CO_2 is 180-5000 ppm (Picarro, no date). Chamber measurements were planned to continue until harvest, however extreme temperatures caused instrument failure, so GHG measurements ceased ~6 weeks before harvest. An Eosense eosMX-P multiplexer (Eosense, Canada) and eosLink-AC software (Eosense, Canada) allowed each chamber to be sampled in turn. Chambers were programmed to close (i.e., sample) for 7-minutes each on a continuous loop sequence. On 25/04/2022, vertical extensions (0.7 m height) were attached between the chamber collar and lid to accommodate the growing crop, increasing the internal chamber volume to 0.209 m^3. The accumulation time of the chambers was then increased from 7 to 10-minutes in accordance with the increased chamber volume.

Winter wheat was harvested from within chamber collars and from a 0.5 m^2 quadrat within each neighbouring plot on 27/07/2022. Harvesting was carried out by hand, cutting the stems 0.1 m above the soil surface. The harvested WW was weighed before and after drying at 60 ˚C for 24-hours to determine its moisture content. At harvest the winter wheat had an average moisture content ± standard deviation of 13.2 ± 3.2 %. The dried winter wheat was threshed using a HALDRUP LT-21 laboratory thresher (HALDRUP, Germany), providing grain, chaff and stalk samples which were ground and analysed for C and N content using a Vario EL Cube elemental analyser (Elementar, UK) according to Pella (1990a; 1990b). Separately, filtration and digestion methods were used to calculate grain N content (Ministry of Agriculture, Fisheries and Food, 1973) which was multiplied by 5.7 to calculate grain protein content (Sosulski and Imafidon, 1990; Ma et al., 2019). Harvest index, or total WW biomass as grain, was calculated according to Equation 6.2 (Amanullah and Inamullah, 2016):

$$Harvest\ index\ (\%) = (grain\ yield\ \div total\ DM\ yield) \times 100 \qquad \text{(Equation 6.2)}$$

6.2.3 Data processing

Greenhouse gas fluxes were calculated using bespoke software for the Eosense chamber system (eos-AnalyzeMX/AC V3.5.0, Eosense, Canada); a linear fit was adjusted to the raw concentration of CO_2 by identifying the start and end of each measurement, which was then used to calculate fluxes of all gases for each sampling point (Petrakis et al., 2017; Barba et al., 2019). Outliers were identified using a modified version of the method by Elbers et al. (2011) which quantifies the uncertainty of CO_2 fluxes based on the threshold detection value (u*), statistical screening, measurement errors, and uncertainties associated with flux calculations. Measurements of CO_2, and associated N_2O and CH_4, identified as outliers (261 sampling points) were then removed. Gaps in the data, either due to instrument failure during the measurement period or as a result of outlier removal were then gap-filled. Missing N_2O and CH_4 data between 20/03/2022 and 13/06/2022 were gap-filled using linear interpolation and missing daytime and night-time CO_2 data between 20/03/2022 and 13/06/2022 were gap-filled separately using linear regression (Dorich et al., 2020; Lucas-Moffat et al., 2022). Thirty-three percent of the data were gap-filled. Complete gap-filled data were analysed using The R Language and Environment for Statistical Computing V4.1.3 (R Core Team, 2021). As one flux measurement was made per chamber every 2-hours, measurements were converted from µmol m^{-2} $s^{-}1$ (CO_2) or nmol m^{-2} s^{-1} (N_2O and CH_4) to g C m^{-2} (CO_2 and CH_4) or g N m^{-2} (N_2O) and daily averages were calculated. Cumulative CO_2, N_2O and CH_4 fluxes were converted to CO_2-equivalent (g m^{-2} day^{-1}) by multiplying these gases by their GWP; 273 for N_2O and 27.9 for CH_4 (Smith et al., 2021).

Greenhouse gas intensity (GHGI) was calculated according to Equation 6.3, adapted from Mosier et al. (2006) and Guo et al. (2022):

$$GHGI\ (kg\ CO_2 equivalent\ kg\ grain^{-1}) = E_D\ \div Y \qquad \text{(Equation 6.3)}$$

where E_D is the cumulative CO_2-equivalent emissions from each fertiliser treatment over the measurement period (i.e., $N_2O + CH_4$; kg CO_2-equivalent ha^{-1}) and Y is grain yield from each fertiliser treatment plot (kg ha^{-1}).

Throughout the paper, GHGIs are based on emissions recorded during the measurement period of this study; we acknowledge that these will not be GHGIs for the entire WW growing season.

Nitrogen use efficiency is the percentage of total N recovered by a plant at harvest (Scottish Government, 2023b); NUE of the whole-crop (NUE_{total}) and grain (NUE_{grain}) were calculated according to Equation 6.4 and Equation 6.5:

$$NUE_{total}\ (\%) = (N\ output\ \div N\ input) \times 100 \qquad \text{(Equation 6.4)}$$

where N output is N content of whole-crop (kg N ha^{-1}) and N input is total N added via fertiliser (kg N ha^{-1}).

$$NUE_{grain}\ (\%) = (N\ output\ \div N\ input) \times 100 \qquad \text{(Equation 6.5)}$$

where N output is N content of grain (kg N ha^{-1}) and N input is total N added via fertiliser (kg N ha^{-1}).

Normality tests were conducted using the Shapiro-Wilk method. Tests for statistically significant differences of mean daily and mean cumulative GHG emissions between each fertiliser treatment were conducted using Kruskal-Wallis and Wilcoxon tests as all data followed a non-normal distribution. Tests for significant differences of average WW dry matter (DM) yield, grain yield, total and grain C and N content, and grain protein content between each treatment were conducted using Kruskal-Wallis and Wilcoxon or ANOVA and Tukey tests dependent on the normality of the data. Multiple linear regression (MLR) was used to investigate the impact of environmental factors (i.e., precipitation, air temperature, soil temperature (0.05 m), WFPS and photosynthetically active radiation (PAR)) on N_2O and CH_4 fluxes for each treatment. Prior to conducting MLR, a correlation matrix was used to assess for collinearity between the environmental variables. There was strong collinearity between soil temperature and air temperature (0.77); MLR showed a higher R^2 value when air temperature was included compared to when soil temperature was included, so soil temperature was removed from MLR to remove the potential effects of collinearity. When considering the dataset excluding the 0-7 days after the first two fertiliser applications, the R^2 value was higher when soil temperature was included compared to when air temperature was included, so for this analysis air temperature was removed from MLR.

6.3 Results

Cumulative N_2O fluxes were highest from TPS and lowest from IF, and cumulative CH_4 fluxes were highest from UPS and lowest from IF (Table 6.1; Figure 6.2; Figure A5.2). Despite lower CH_4 fluxes from TPS compared to UPS, N_2O fluxes were highest from TPS, meaning that total CO_2 equivalent fluxes were highest from TPS compared to UPS, disproving our first hypothesis. Our second hypothesis is proven by the IF treatment having lower non-CO_2 GHG emissions than the organic fertiliser treatments (i.e., TPS and UPS). The response of the non-CO_2 fluxes to the fertiliser treatments is discussed in more detail below. Cumulative CO_2 fluxes were highest from UPS and lowest from IF, and were significantly different between UPS and IF but not between UPS and TPS or IF and TPS (Table A5.3). Further results on CO_2 fluxes, including mean daily and cumulative CO_2 fluxes, and diurnal CO_2 fluxes for each treatment over each WW growth stage are presented in Figures A5.3, A5.4 and A5.5. These data are not presented as main results as non-CO_2 GHG fluxes are the focus of this study. CO_2-equivalent fluxes of N_2O and CH_4 were highest from TPS and lowest from IF (Table 6.2; Figure A5.2).

TABLE 6.2 MEAN DAILY AND MEAN CUMULATIVE FLUXES, AND MEAN GHGI OVER THE 83-DAY MEASUREMENT PERIOD ± STANDARD DEVIATION (SD) FOR EACH FERTILISER TREATMENT (IF = INORGANIC FERTILISER, UPS = UNTREATED PIG SLURRY, TPS = TREATED PIG SLURRY). ACROSS EACH ROW, DIFFERENT LETTERS INDICATE SIGNIFICANT DIFFERENCES IN THE VARIABLE OF INTEREST BETWEEN FERTILISER TREATMENTS.

		IF	UPS	TPS
N_2O	Mean daily ± SD (g N m^{-2} day^{-1})	0.002 ± 0 a	0.004 ± 0 b	0.013 ± 0 a
	Mean cumulative ± SD (g N m^{-2})	0.13 ± 0 a	0.32 ± 0.1 a	1.14 ± 0.1 a
	Mean daily 0-7 days after first two fertiliser applications ± SD (g N m^{-2} day^{-1})	0.004 ± 0 a	0.013 ± 0 b	0.068 ± 0 c
CH_4	Mean daily ± SD (g C m^{-2} day^{-1})	-0.0003 ± 5.8e-05 a	0.0004 ± 0.0006 b	-0.0003 ± 0.0001 a
	Mean cumulative ± SD (g C m^{-2})	-1.4 ± 0.3 a	3.2 ± 1.4 a	-1.4 ± 0.6 a
	Mean daily 0-7 days after first two fertiliser applications ± SD (g C m^{-2} day^{-1})	-0.0002 ± 0 a	0.004 ± 0.4 b	-0.0001 ± 0 a
CO_2-eq (N_2O + CH_4)	Mean cumulative ± SD (g CO_2-eq m^{-2})	34.2 ± 7.6 a	88.8 ± 14.3 a	311.7 ± 34.9 a
	Mean GHGI ± SD (kg CO_2-eq kg $grain^{-1}$)	0.03 ± 0.005 a	0.07 ± 0.02 a	0.2 ± 0.02 a

6.3.1 N_2O fluxes

Cumulative N_2O fluxes were highest from TPS and lowest from IF and were not significantly different between treatments (Table 6.2; Figure A5.2). Nitrous oxide fluxes increased with increasing WFPS and the application of untreated pig slurry and treated pig slurry (P = <0.05), and decreased with increasing PAR (P = <0.05) (Figures A5.6 and A5.7). When treated pig slurry was applied, significant interactions were observed between N_2O fluxes, WFPS, air temperature and PAR (P = <0.05) (Figure A5.7). Precipitation did not significantly influence N_2O fluxes (P = 0.42). Mean daily N_2O fluxes were highest from TPS and lowest from IF and were significantly different between IF and UPS (P = 0.004) and IF and TPS (P = 0.03) but not between UPS and TPS (P = 0.82) (Table 6.2). Nitrous oxide fluxes increased following the first fertiliser application to TPS and following the second fertiliser applications to UPS and TPS, peaking one day after application and decreasing over five to fourteen days before returning to pre-fertilisation levels (Figure 6.1; Figure 6.2).

Nitrous oxide fluxes from TPS and UPS did not respond to the third and fourth fertiliser applications, which were in the form of inorganic fertiliser and contained less N than the previous two applications which were in the form of organic fertiliser (Figure 6.1; Figure 6.2; Table 6.1). Nitrous oxide fluxes from IF did not respond to any of the fertiliser applications (Figure 6.1; Figure 6.2). When considering N_2O fluxes from within seven days of the first two fertiliser applications only (i.e., when organic fertilisers were added to TPS and UPS) (Figure 6.3), mean daily N_2O fluxes were highest from TPS and lowest from IF and were significantly different between all treatments (P = <0.05) (Table 6.2).

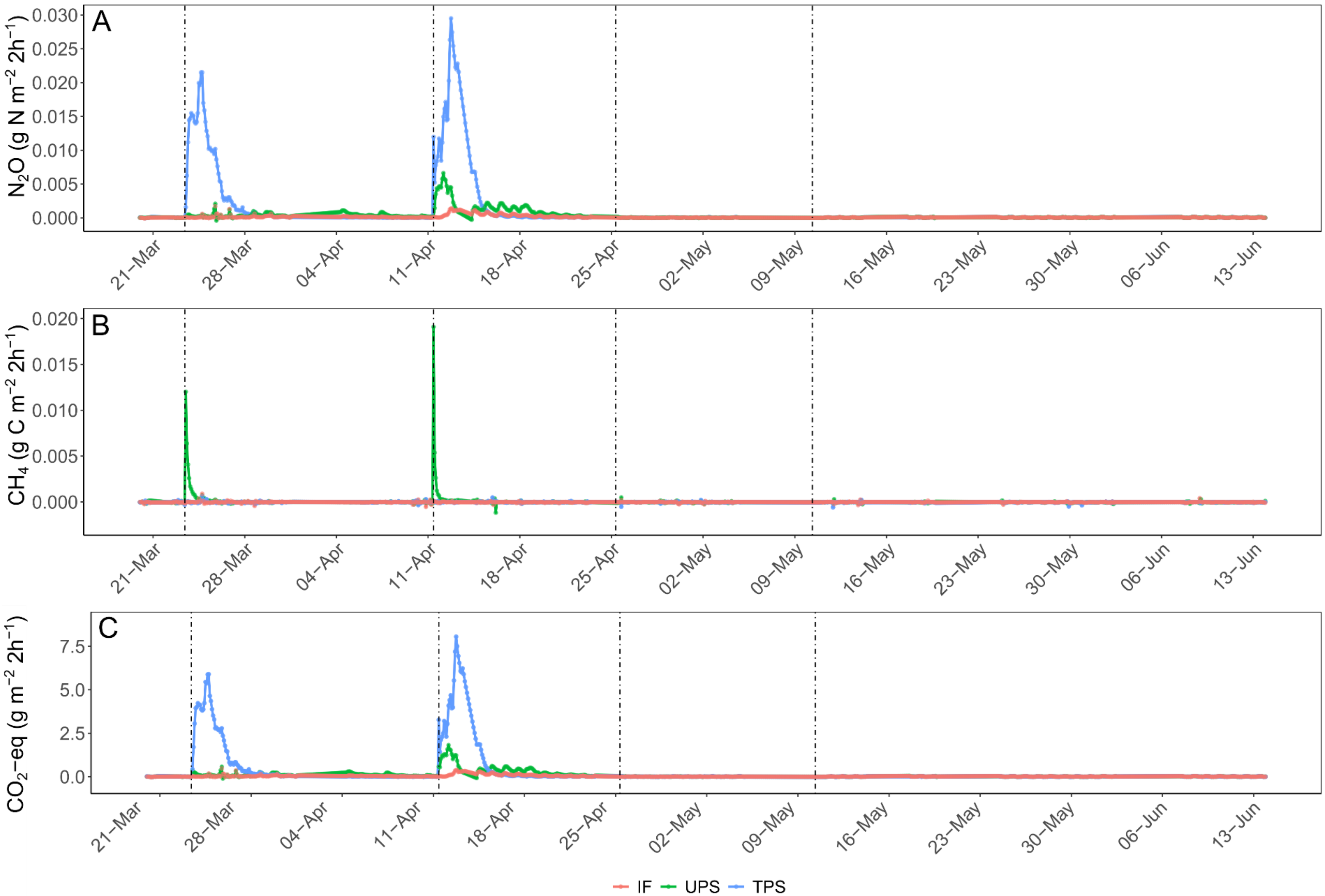
A
B
C
N2O (g N m−2 2h−1)
CH4 (g C m−2 2h−1)
CO2−eq (g m−2 2h−1)
0.000
0.005
0.010
0.015
0.020
0.025
0.030
0.0
2.5
5.0
7.5
21−Mar
28−Mar
04−Apr
11−Apr
18−Apr
25−Apr
02−May
09−May
16−May
23−May
30−May
06−Jun
13−Jun
IF
UPS
TPS

Figure 6.1 2-hour fluxes of (A) N_2O, (B) CH_4 and (C) CO_2-equivalent fluxes of N_2O and CH_4 for each fertiliser treatment (IF = inorganic fertiliser, UPS = untreated pig slurry, TPS = treated pig slurry). Each data point represents the mean of three chambers used per treatment and vertical dashed lines represent the split applications of fertilisers. Error bars have been removed to aid visualisation.

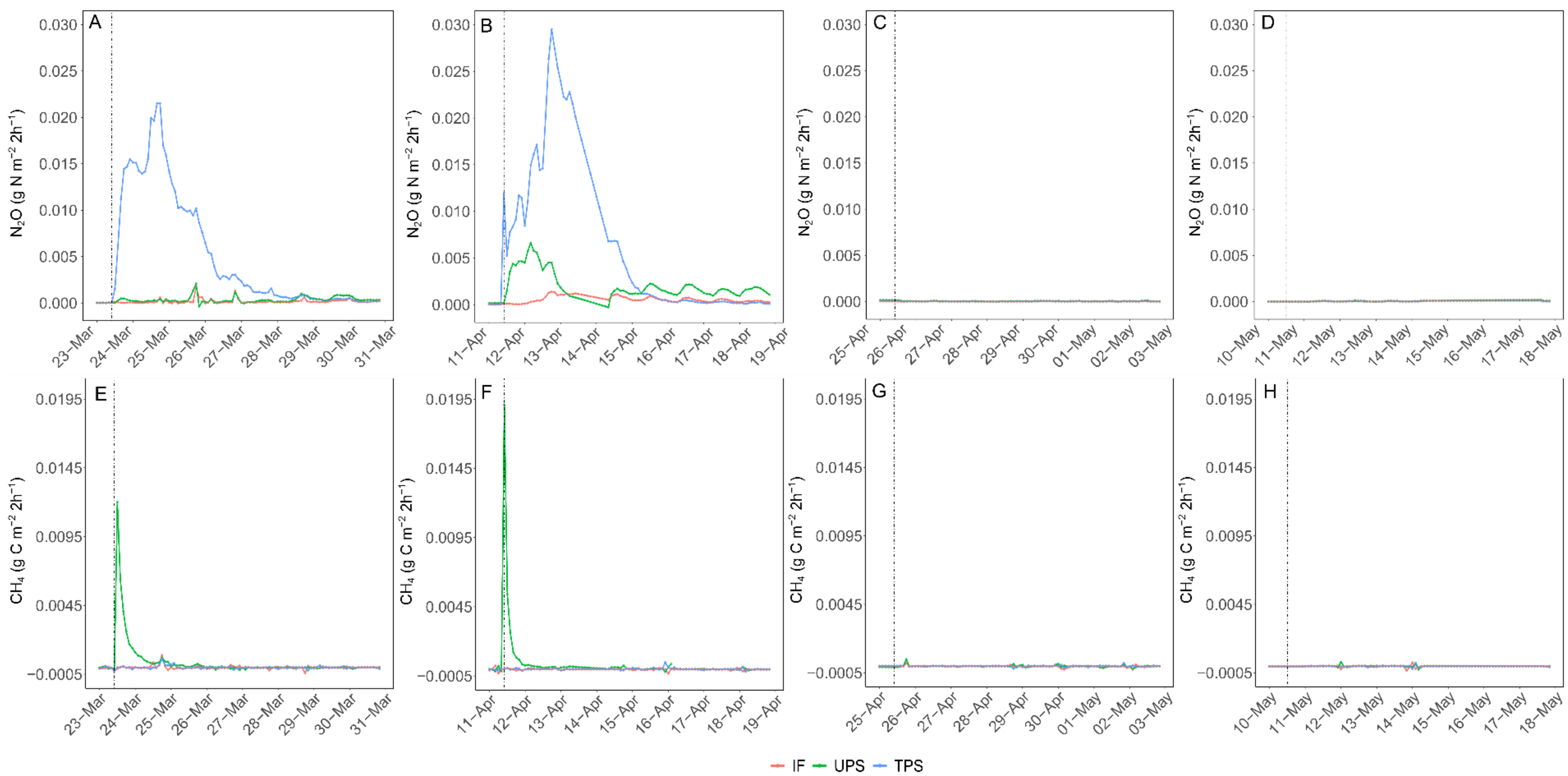

FIGURE 6.2 2-HOUR FLUXES OF (A, B, C, D) N_2O AND (E, F, G, H) CH_4 DURING THE FIRST 7 DAYS OF EACH FERTILISER APPLICATION FOR EACH FERTILISER TREATMENT (IF = INORGANIC FERTILISER, UPS = UNTREATED PIG SLURRY, TPS = TREATED PIG SLURRY). EACH DATA POINT REPRESENTS THE MEAN OF THREE CHAMBERS USED PER TREATMENT AND VERTICAL DASHED LINES REPRESENT THE SPLIT APPLICATIONS OF FERTILISERS. ERROR BARS HAVE BEEN REMOVED TO AID VISUALISATION.

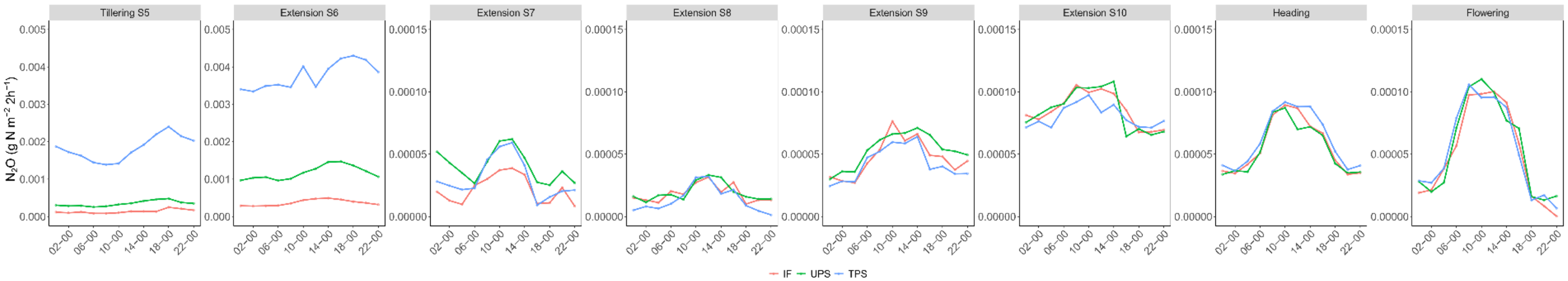

FIGURE 6.3 MEAN 2-HOUR FLUXES OF N_2O FOR EACH FERTILISER TREATMENT (IF = INORGANIC FERTILISER, UPS = UNTREATED PIG SLURRY, TPS = TREATED PIG SLURRY) FOR EACH WINTER WHEAT GROWTH STAGE OVER THE MEASUREMENT PERIOD. EACH DATA POINT REPRESENTS THE MEAN OF THREE CHAMBERS USED PER TREATMENT. ERROR BARS HAVE BEEN REMOVED TO AID VISUALISATION. THE DATES OF EACH GROWTH STAGE, AND THE AVERAGE DAILY AIR TEMPERATURE AND TOTAL RAINFALL PER WINTER WHEAT GROWTH STAGE ARE SHOWN IN TABLE A5.5.

Diurnal variations in N_2O fluxes were identified throughout the measurement period, apart from within 0 to 7 days of the first two fertiliser applications (i.e., when organic fertilisers were applied to UPS and TPS and thus N_2O flux activity was at its maximum). Therefore, to better understand the controls on the diurnal fluxes of N_2O, data from days 0 to 7 after the first two fertiliser applications were excluded from further analysis. Following this removal, an increase in WFPS and PAR were found to increase N_2O fluxes; however N_2O fluxes decreased with increasing soil temperature (Figure A5.8). There was no significant effect of precipitation on N_2O fluxes (P = >0.05). Significant interactions (P = <0.05) were identified between pig slurry application and several environmental variables and N_2O fluxes (Table A5.4). There was no clear diurnal trend in N_2O fluxes observed at Tillering S5 and Extension S6, although the magnitude of N_2O flux was higher from TPS compared to IF and UPS at these growth stages (Figure 6.3). From Extension S7 onwards a slight diurnal trend in N_2O fluxes became prevalent for all treatments and became more pronounced from Extension S10 onwards – fluxes increased during the day and decreased at night, with the highest fluxes observed between 10:00 and 12:00 (Figure 6.3).

6.3.2 CH_4 fluxes

Cumulative CH_4 fluxes were highest from UPS and lower from IF and TPS and were not significantly different between treatments (Table 6.1; Figure A5.2). Methane fluxes increased with increasing WFPS, PAR, air temperature and pig slurry application (P = <0.05) (Figure A5.6; Figure A5.7). There was no significant influence of precipitation on CH_4 fluxes (P = 0.24). Mean daily CH_4 fluxes were highest from UPS and lower from IF and TPS but were not significantly different between treatments (P = >0.05) (Table 6.1). Methane fluxes from UPS peaked immediately after the first and second fertiliser applications and remained elevated for less than 24-hours before returning to pre-fertilisation levels (Figure 6.1; Figure 6.2). Methane fluxes did not respond to the third and fourth fertiliser applications which were in the form of inorganic fertiliser (Figure 6.1; Figure A5.2; Table A5.2). Methane fluxes from IF and TPS remained low for the entire measurement period and did not respond to any fertiliser applications (Figure 6.1; Figure 6.2; Table A5.5). When considering CH_4 fluxes from 0 to 7 days of the first two fertiliser applications only (Figure 6.2), mean daily CH_4

fluxes were higher from UPS than IF and TPS but were not significantly different between treatments (*P* = >0.05) (Table 6.1). There was no clear diurnal trend in CH_4 fluxes for any of the treatments at any of the WW growth stages (Figure 6.4).

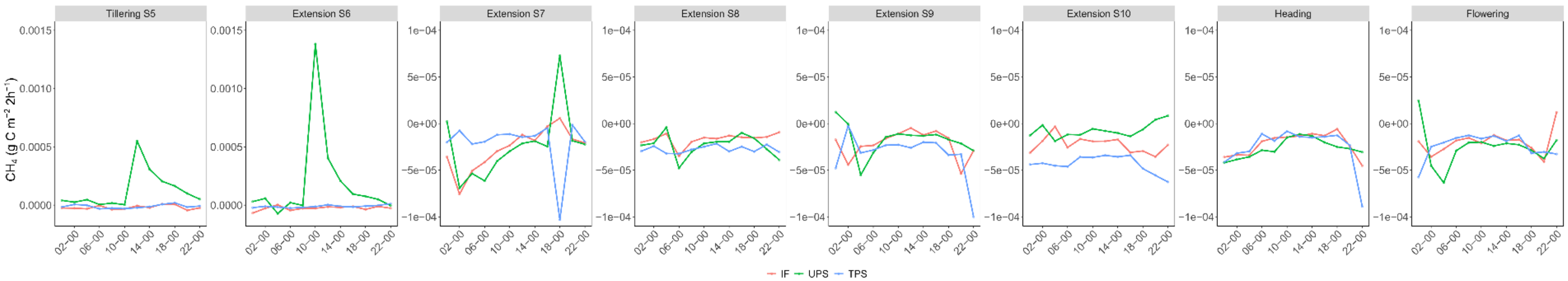

FIGURE 6.4 MEAN 2-HOUR FLUXES OF CH_4 FOR EACH FERTILISER TREATMENT (IF = INORGANIC FERTILISER, UPS = UNTREATED PIG SLURRY, TPS = TREATED PIG SLURRY) FOR EACH WINTER WHEAT GROWTH STAGE OVER THE MEASUREMENT PERIOD. EACH DATA POINT REPRESENTS THE MEAN OF THREE CHAMBERS USED PER TREATMENT. ERROR BARS HAVE BEEN REMOVED TO AID VISUALISATION. THE DATES OF EACH GROWTH STAGE, AND THE AVERAGE DAILY AIR TEMPERATURE AND TOTAL RAINFALL PER WINTER WHEAT GROWTH STAGE ARE SHOWN IN TABLE A5.5.

6.3.3 Yield response

The average total WW DM yield did not vary significantly between treatments (Table 6.2) and ranged from 22.75 ± 1.31 t ha^{-1} (UPS) to 25.21 ± 3.68 t ha^{-1} (TPS), which is slightly higher than that reported for the entire field (22.1 ± 3.4 t ha^{-1}). Winter wheat grain yield ranged from 13 ± 1.2 t ha^{-1} (UPS) to 14.5 t ha^{-1} (TPS), which is slightly higher than that reported for the entire field (12.9 t ha^{-1}). At harvest, the harvest index was similar between treatments (Table 6.2). Dry matter yield, total C and N content, grain yield, grain C and N content, and grain protein content were not significantly different between any of the treatments (*P* = >0.05); NUE_{total} and NUE_{grain} were highest for IF and lowest for TPS and were not significantly different between any of the treatments (Table 6.2). Mean GHGI was highest from TPS and lowest from IF (Table 6.1) and was not significantly different between treatments (*P* = 0.1).

TABLE 6.3 SEED PLANTING DENSITY, TOTAL BIOMASS AND CROP YIELD, HARVEST INDEX, WHOLE-CROP AND GRAIN C AND N CONTENT, TOTAL C AND N REMOVED IN WHOLE-CROP AND GRAIN, PROPORTION OF TOTAL CROP N IN GRAIN, GRAIN PROTEIN CONTENT, NITROGEN USE EFFICIENCY OF TOTAL BIOMASS (NUE_{TOTAL}) AND GRAIN YIELD (NUE_{GRAIN}), AND THE PROPORTION OF APPLIED N LOST AS N_2O-N FOR EACH TREATMENT (IF = INORGANIC FERTILISER, UPS = UNTREATED PIG SLURRY, TPS = TREATED PIG SLURRY) ± STANDARD DEVIATION (SD) WHERE APPROPRIATE. NOTE THAT WHOLE-CROP REFERS TO THE ENTIRE HARVESTED PLANT (I.E., CHAFF, GRAIN AND STALK). SAMPLES TAKEN FROM PLOTS USING A 0.5 M^2 QUADRAT (N=3). ACROSS EACH ROW, THE SAME LETTERS INDICATE NO SIGNIFICANT DIFFERENCE IN THE VARIABLE OF INTEREST BETWEEN FERTILISER TREATMENTS.

Fertiliser treatment	IF	UPS	TPS
Planting density ± SD (seeds m^2)	383.33 ± 137.7	400 ± 114.6	341.67 ± 104.1
Total biomass yield ± SD (t DM ha^{-1})	23.76 ± 1.5 a	22.75 ± 1.3 a	25.21 ± 3.7 a
Grain yield ± SD (t ha^{-1})	13.05 ± 0.9 a	12.98 ± 1.2 a	14.84 ± 2.7 a
Harvest index ± SD (%)	54.92 ± 1.1 a	57 ± 1.7 a	58.66 ± 2.3 a
Whole-crop C content ± SD (%)	40.71 ± 0 a	40.58 ± 0.2 a	40.57 ± 0.1 a
Total C removed in whole-crop (t ha^{-1})	9.67 ± 0.6 a	9.23 ± 0.5 a	10.23 ± 1.5 a
Grain C content ± SD (%)	39.06 ± 0.7 a	38.80 ± 0.4 a	38.84 ± 0.8 a

Whole-crop N content ± SD (%)	0.78 ± 0.1 a	0.79 ± 0 a	0.78 ± 0.1 a
Total N removed in whole-crop (t ha^{-1})	0.18 ± 0 a	0.18 ± 0 a	0.2 ± 0 a
Grain N content ± SD (%)	1.29 ± 0.1 a	1.23 ± 0.1 a	1.22 ± 0.1 a
Total N removed in grain (t ha^{-1})	0.17 ± 0 a	0.16 ± 0 a	0.18 ± 0 a
% of total crop N in grain	90.78 ± 1.9 a	88.93 ± 9.6 a	93.21 ± 11.6 a
Grain protein content ± SD (%)	6.17 ± 0.6 a	6.64 ± 0.8 a	5.97 ± 0.7 a
NUE_{total} (%)	83.64 ± 3.7 a	81.69 ± 1 a	77.81 ± 17.4 a
NUE_{grain} (%)	75.89 ± 2.4 a	72.63 ± 7.6 a	71.89 ± 15 a
% of applied N lost as N_2O-N	0.6 ± 0.1 a	0.9 ± 0.1 a	4 ± 0.5 a

6.4 Discussion

6.4.1 Plasma treatment of pig slurry increased N_2O emissions

The large peaks of N_2O following the two applications of treated pig slurry are responsible for TPS having the highest cumulative N_2O emissions. Similarly, the smaller N_2O peak following the second application of pig slurry to UPS is responsible for this treatment having the second highest cumulative N_2O emissions relative to IF. Elevated N_2O fluxes following N fertiliser application are well-documented and are often attributed to fertiliser N becoming available for conversion to N_2O shortly after application, as there is competition between plant uptake and soil microbes for the N (Ma et al., 2013; Officer et al., 2015). Many studies have observed higher N_2O emissions from crops fertilised with organic fertiliser, or a combination of organic and inorganic fertiliser, compared to those amended with inorganic fertiliser only (Pelster et al., 2012; Ball et al., 2014; Yang et al., 2015). Organic fertilisers have a higher labile C content which is easily decomposed by soil microorganisms and releases mineralizable N for the production of N_2O (Hangs and Schoneau, 2022); this is likely to have caused the higher N_2O emissions from TPS and UPS compared to IF. Furthermore, the pig slurry and treated pig slurry had a higher content of fine solids than the inorganic fertiliser; fine solids block soil pores and restrict oxygen movement through soil, which creates favourable conditions for N_2O production (Chadwick et al., 2000). We found that the plasma induction process increased the nitrate-N content of the pig slurry; the higher content of inorganic N combined with the C in the pig slurry is likely to be responsible for the higher N_2O emissions (Shurpali et al., 2016; Li et al., 2022) from TPS compared to UPS. Mousavi et al. (2023) found that the nitrification potential of plasma-treated pig slurry was higher than that of other fertilisers due to its higher volatile organic C content, which reduces NH_3 immobilisation, and so may also explain the higher N_2O emissions from TPS. Denitrification is highly influenced by pH, with denitrification being slowed or even inhibited at lower pH levels (Liu et al., 2010; Olaya-Abril et al., 2021). At lower pH, the transformation of N_2O to nitrogen gas is inhibited, meaning that the N_2O is available to be emitted from the soil (Liu et al., 2010; Olaya-Abril et al., 2021). The lower pH of the treated pig slurry relative to the untreated pig slurry (Table 6.1) may therefore also explain the higher N_2O emissions from TPS. It should be noted that the amount of available N applied to TPS was slightly higher than to UPS and IF which may have contributed to its higher N_2O emission, although because the N_2O emissions from TPS are so much higher than the other two treatments, it is highly unlikely that this discrepancy is the only reason.

A higher soil moisture content can restrict aeration and reduce soil oxygen concentration, creating favourable conditions for denitrification and N_2O emission (Westphal et al., 2018; Kostyanosvky et al., 2019; Li et al., 2022). This can explain the higher N_2O emissions from TPS and UPS, as the relationship between N_2O and WFPS was higher for these treatments than IF, and WFPS appeared highest at TPS. The lack of response of N_2O fluxes to the applications of inorganic fertiliser across all treatments is explained by the drought conditions experienced during the study. The inorganic fertilisers were applied in the form of solid granules (application 1) or a small volume of liquid (subsequent applications), which did not wet the soil enough to stimulate N_2O emissions. Verdi et al. (2019) also found low N_2O emissions from a dry soil when solid inorganic fertiliser was added. The volume of liquid applied as pig slurry and treated pig slurry was greater, and thus wetted up the soil more, inducing N_2O emission.

6.4.2 Plasma treatment of pig slurry decreased CH_4 emissions

The immediate peaks in CH_4 fluxes following the two applications of pig slurry are responsible for UPS having the highest total CH_4 fluxes. Methane is produced during pig slurry storage as the conditions and C content of the slurry facilitate methanogenesis; the CH_4 is dissolved into the pig slurry and then volatilised and emitted to the atmosphere following slurry application (Rochette and Cote, 2000; Bastami et al., 2016). Severin et al. (2015) also measured higher CH_4 emissions from crops amended with pig slurry. The small CH_4 uptake by IF and TPS is not unexpected, as methanotrophy occurs in well-drained agricultural soils (Serrano-Silva et al., 2014). Inorganic fertiliser does not contain a C source to facilitate methanogenesis (Moreno-Garcia et al., 2020), and thus CH_4 production, and the plasma induction process prevents CH_4 production during slurry storage by acidifying the slurry and reducing its pH (Petersen et al., 2012; Overmeyer et al., 2021; Tooth et al., 2012; Ambrose et al., 2023), so no CH_4 was emitted from IF and TPS upon application. There is the potential for CH_4 to be produced in soil, and then emitted, following the application of slurry due to the anoxic conditions created by rapid C mineralisation after the input of C in the organic fertiliser (Le Mer and Roger, 2001; Yuan et al., 2019), this accounts for the elevated CH_4 emissions from UPS. The lower pH of the treated pig slurry, as a result of acidification during plasma treatment, prohibiting methanogenesis during storage also appears to inhibit CH_4 production on application to the field, as the C input via treated pig slurry application does not induce CH_4 emissions. The plasma induction process therefore has clear benefits in terms of reducing CH_4 emissions during the storage and application of pig slurry to agricultural soil.

6.4.3 CO_2-equivalent and GHGI highest from plasma-treated pig slurry

Nitrous oxide has a higher global warming potential (273) than CH_4 (27.9) (Smith et al., 2021), and, as N_2O emissions were considerably higher from TPS compared to the other treatments, CO_2-equivalent emissions were therefore also highest from TPS. The higher CH_4 fluxes from UPS compared to TPS and IF were not large enough to outweigh the high N_2O fluxes from TPS when converted to CO_2-equivalent. Across the literature, cumulative CO_2-equivalent fluxes from WW fertilised with 100-300 kg inorganic N ha^{-1} range from 15 to 102.5 g CO_2-equivalent m^{-2} (Sainju et al., 2022; Huang et al., 2013) (Table A5.6); the CO_2-

equivalent emissions we measured from IF are within this range. There is a lack of data on CO_2-equivalent emissions from pig slurry when used as an organic fertiliser, presenting a significant research gap that must be addressed to enhance the understanding of the impacts of fertiliser type on GHG emissions. As all treatments received a similar amount of plant-available N, the lack of influence of treatment type on WW growth, including DM yield, grain yield and grain protein content is not unexpected. Cai et al. (2013) also observed no significant difference in grain yield between crops amended with a similar N rate of inorganic and organic fertilisers. Our results show that it is possible to replace over half of inorganic N fertiliser with organic N fertiliser and achieve the same yield. As yield was not significantly different between the treatments, this meant that GHGI followed the trend of cumulative CO_2-equivalent emissions, with the highest fluxes from TPS. When considering WW yield, the phosphorus and potassium applied to the crop via the fertiliser treatments should be noted – the pig slurry and treated pig slurry contained phosphorus and potassium whereas the inorganic fertiliser did not. As soil potassium data is not available, it is not possible to assess whether this was a factor limiting crop production, however it is unlikely as the yield of ~12 t ha^{-1} for all treatments is high, and the soil was not P limited (P index of 3). As we consider cumulative emissions, it is also important to note that ~6 weeks of data are not included in this study due to an error with the GHG measurement chambers. Given the uniform and consistent flux pattern in the weeks prior to this, and the fact that there were no N fertiliser applications during this time, we propose that the addition of this missing data would have a minimal impact on the cumulative emissions.

6.4.4 Diurnal N_2O emissions observed outside of N_2O peaks

The diurnal pattern and peak of N_2O emissions during the middle of the day (observed from Extension S10 onwards) for all treatments coincides with maximum CO_2 uptake. This pattern was also reported in a review by Wu et al. (2021) who found that over half of the datasets reviewed observed N_2O fluxes peaking during the day. Chadwick et al. (2000) and Keane et al. (2018) hypothesise that increases in soil temperature, WFPS and PAR increased N_2O fluxes. Furthermore, Keane et al. (2018) propose that, as C availability is a key driver of denitrification, higher PAR and temperature during the middle of the day would increase photosynthate exudation and microbial respiration, reducing oxygen availability, and stimulating denitrification and N_2O emission. Our results support these hypotheses, as we found that, when excluding fluxes measured within 0 to 7 days of the first two fertiliser applications, N_2O fluxes increased with WFPS and PAR. The Tillering S5 and Extension S6 growth stages coincided with the applications of pig slurry and treated pig slurry, which subsequently caused peaks of N_2O emission, and so no diurnal patterns in N_2O emissions were observed from any treatments during these growth stages.

6.4.5 Implications for research and policy

We show that treating pig slurry with plasma induction does not reduce overall non-CO_2 GHG emissions, in fact it increases them in comparison to untreated pig slurry and inorganic fertiliser. Although soil CH_4 emissions were reduced by treating pig slurry with plasma induction, N_2O soil emissions from plasma-treated slurry were considerably greater than non-treated slurry. Furthermore, the CO_2-equivalent emissions from the organic fertiliser treatments (TPS and UPS) were higher than those from the inorganic fertiliser treatment (IF). These trade-offs between N_2O and CH_4 emissions highlight the need to continue the development of innovative technologies to improve agricultural sustainability. Whilst other research has found benefits of the use of plasma-treated slurries, such as lower NH_3 emissions (Gillbard, 2023) and positive effects on soil fauna (Mousavi et al., 2022), the high N_2O emissions found in our study show that more research is required to determine how these emissions can be reduced. This may include de-watering slurries or using nitrification inhibitors to reduce N_2O emissions associated with the application of organic fertilisers to soils to improve on-farm waste management and farm adherence to agricultural policy (Ruser and Schulz, 2015; Willen et al., 2016). Further research exploring the influence of fertiliser type on GHG emissions should also measure fluxes from a control treatment receiving no fertiliser, which would enable the calculation of emission factors, and from a range of environments to assess the influence of climate and soil variables. Whilst we show that, overall, differences in GHG emissions were considerable between treatments, the cumulative N_2O and CH_4 emissions were not significantly different. This is likely to be due to the small number of replicates per treatment (N=3). A replicated study with both an increased sample size per treatment and control treatment would strengthen the results. Furthermore, a replicated study would allow the different quantity of available N applied to the treatments in this study to be addressed. The fertilizer in this study was applied based on analysis of previous pig slurry and treated pig slurry, however these characteristics (such as available N) changed over time and thus were slightly different in the slurries applied. As this experiment only focuses on emissions from fertiliser application until ~6 weeks before harvest, future trials should be longer-term, measuring GHG emissions across a full crop season as well as across years to account for inter-annual variability. It is crucial that this research is conducted prior to the commercialisation of new technologies for organic waste management. It should be noted that the plasma induction process reduced slurry pH from ~7 to below 5 (Table 6.1), and that slurry acidification is known to reduce NH_3 emissions by 70 % (Kupper et al. 2020). Measuring NH_3 emissions alongside GHGs would provide a more comprehensive understanding of the emissions associated with the use of agricultural fertilisers and ensure that all trade-offs are fully accounted for. These measurements should be integrated into dynamic biogeochemical models and life-cycle analyses to account for other significant emissions associated with the use of agricultural fertilisers, such as those generated in fertiliser manufacturing from the Haber-Bosh process, and allow the full environmental and climatic impact of fertiliser production and application to be ascertained.

6.5 Conclusion

The use of plasma-treated pig slurry as an organic soil amendment reduced soil CH_4 emissions relative to non-treated pig slurry after application. Plasma-treated slurry increased N_2O emissions considerably, however, which outweighed the savings from CH_4 reduction and so CO_2-equivalent emissions were greater from treated than non-treated pig slurry. Winter wheat yield was high for all treatments and was not affected by the fertiliser type used. Plasma-treated pig slurry is therefore not currently a suitable soil amendment should farmers wish to reduce GHG emissions from their land. Furthermore, the application of organic fertilisers (i.e., treated and non-treated pig slurries) resulted in higher GHG emissions than when inorganic fertiliser was applied. We therefore recommend that our results be integrated into a life-cycle analysis, to determine whether the use of organic fertilisers still emit more than inorganic fertilisers when the associated downstream GHG emissions are considered. In addition, future research should focus on how N_2O emissions can be reduced from plasma-treated pig slurry, conducting plot trials to assess the effect of fertiliser rate, timing and placement.

CRediT author statement

Isobel L. Lloyd: Conceptualisation, Methodology, Formal analysis, Investigation, Data curation, Writing – Original draft, Writing – Review and editing, Visualisation, Funding acquisition. **Richard P. Grayson:** Conceptualisation, Methodology, Investigation, Resources, Writing – Review and editing, Supervision, Funding acquisition. **Marcelo V. Galdos:** Conceptualisation, Methodology, Resources, Writing – Review and editing, Supervision. **Ross Morrison:** Conceptualisation, Methodology, Resources, Writing – Review and editing, Supervision. **Pippa J. Chapman:** Conceptualisation, Methodology, Resources, Writing – Review and editing, Supervision, Funding acquisition, Project administration.

Data availability

Data supporting the findings of this study are available at: https://doi.org/10.5285/4ed0023e-da9b-45a8-86de-3a371cc7dcc1

Chapter 7 Synthesis

The intensification of agriculture over the last 200 years has depleted soil organic carbon (SOC) stocks and caused an estimated soil carbon (C) debt of 133 Pg C (Sanderman et al., 2017). The depletion of SOC is a global problem that has detrimental impacts on soil health as it reduces soil fertility, impairs soil hydrological functions and degrades soil structure (Lal, 2004b). Furthermore, SOC loss contributes to anthropogenic greenhouse gas (GHG) emissions (Lal, 2004a; Jiang et al., 2023), making it increasingly difficult to achieve the Sustainable Development Goals set by the United Nations (Keesstra et al., 2016). There is an urgent need to reduce GHG emissions from the agricultural sector and enhance SOC storage, not only to meet national net zero targets, such as the UK Government's aim for net zero by 2050 (DESNZ, 2023b), and achieve the goals of the Paris Climate Agreement (United Nations, 2015), but also to improve soil health and resilience (Lal, 2006; Minasny et al., 2017) and support environmental health and wellbeing (Victoria et al., 2012; West et al., 2013; Milne et al., 2015).

A variety of best management practices are promoted as effective mechanisms to reduce soil C loss and GHG emissions from agricultural soils (Table 1.1). These are focussed on reducing soil disturbance (Soussana et al., 2007), avoiding fallow periods (Lal, 2015b; Jian et al., 2020), increasing C inputs (C_I) from external sources (Chew et al., 2019; Wall et al., 2023) and optimising fertiliser application (Misselbrook et al., 2016; Rosolem et al., 2017; Cardenas et al., 2019). The success of each practice at reducing C loss and GHG emissions will depend on the local soil and climate conditions (Davidson and Janssens, 2006; Jager et al., 2011; Smith, 2012; Oertel et al., 2016; Shakoor et al., 2021; Black et al., 2022). To avoid further SOC loss and GHG emissions, prevent trade-offs between emissions reductions, and avoid transferring emissions to other areas of the sector, the choice of best management practice(s) should be chosen carefully, on a site-by-site basis, considering a site's environmental conditions. Despite this, there is extremely limited knowledge on the impacts that various best management practices have on C dynamics and GHG fluxes from a range of agricultural sites in the UK with different soil types and climate conditions. This thesis compared net ecosystem exchange (NEE), net ecosystem productivity (NEP) and GHG emissions from sites in the UK under different agricultural management practices. In doing so, the thesis addresses the main question of how these environmental and management factors affect GHG fluxes from UK agricultural soils. This chapter synthesises the findings from Chapters 2-6 and discusses the implications of these findings in the context of C sequestration, GHG emissions and agricultural policy. The chapter ends with an outline of the limitations of the research conducted and recommendations for future work.

7.1 Key research findings

The main aim of this thesis was to assess how NEP and GHG emissions from agricultural soils in the UK are affected by the management practices used and how these fluxes are related to climate conditions and soil type. The observational, monitoring and experimental studies undertaken to achieve this were driven by five main research questions. This section reports the key findings of the research that answer these questions (summarised in Table 7.1).

TABLE 7.1 SUMMARY OF THE KEY RESEARCH FINDINGS AND CORRESPONDING CHAPTERS.

Research question	Main finding	Chapter	Method
1. How do climate, soil type and agricultural management influence the NEP of global agricultural soils?	Global croplands lose more C than managed grasslands; the NEP of global croplands is influenced by climate type and the NEP of global managed grasslands is influenced by N fertilisation rate.	2	Meta-analysis
2. How does soil type affect the NEE and NEP of maize grown	Growing maize for bioenergy results in C loss; C loss over twice as high when maize is grown on peat compared to mineral soil.	3	Eddy covariance

for bioenergy in the UK?			
3. How does crop type affect the NEE and NEP of agricultural land in the UK?	The cropland behaved as a C source during the maize growing season but was a C sink during the winter wheat and vining pea growing seasons. When considering fallow periods and crop growing seasons, the field was a C source over 2.5-years.	4	Eddy covariance
4. How does agricultural land use affect the NEE and NEP of agricultural land in the UK?	Permanent pasture was a C source and cropland was C neutral over one year.	5	Eddy covariance
5. How does fertiliser type influence GHG fluxes from a winter wheat crop grown on a mineral soil in the UK?	Carbon dioxide (CO_2)-equivalent emissions were higher from winter wheat fertilised with plasma-treated pig slurry compared to untreated pig slurry and inorganic fertiliser; plasma-treated pig slurry had no methane (CH_4) emission but considerably higher nitrous oxide (N_2O) emissions relative to untreated pig slurry and inorganic fertiliser.	6	Automated chambers

7.1.1 The effect of climate, soil type and agricultural management on the NEP of global agricultural soils

The aim of Chapter 2 was to synthesise global NEP data, measured on an annual basis, from a range of cropland and managed grassland sites spanning varying soil types, climates and agricultural management practices to answer the question “How do climate, soil type and agricultural management influence the NEP of global agricultural soils?”. A total of 242 annual measurements were included in the meta-analysis, taken from 40 publications selected on the basis that annual NEE was measured with eddy covariance (EC) and that carbon import (C_I) and C export (C_H) were reported for the calculation of NEP, along with sufficient meta-data to contextualise the site. The results of the meta-analysis showed that, on average, global croplands had a significantly higher NEP (i.e., were losing more C), a significantly more negative NEE and a significantly lower C_I than managed grasslands (Figure 7.1). Carbon export was not significantly different between the two land uses (Figure 7.1) as the greater *in-situ* CO_2 uptake of croplands compared to managed grasslands was counteracted by the significantly greater mean annual C_I to managed grasslands.

The mean annual NEP of global croplands was significantly influenced by a site’s Köppen climate classification, but not by soil type, the amount of nitrogen (N) fertiliser added, the inclusion of cover crops, residue management (i.e., retention or removal), crop type or tillage method. Croplands in Warm-summer Mediterranean climates (temperate) had a significantly lower annual NEP than croplands in Warm-summer humid continental climates and Monsoon-influenced hot-summer humid continental climates (subtropical). The higher NEP of the sites in subtropical climates compared to temperate climates can be attributed to subtropical climates typically being warmer which results in higher microbial activity, soil organic matter (SOM) decomposition and total ecosystem respiration (TER), and thus higher NEE and NEP (Lopez-Garrido et al., 2014; Maia et al., 2019; Badaru et al., 2022). The mean annual NEP of global managed grasslands was significantly influenced by the amount of N fertiliser added to a site, but not by Köppen climate classification, soil type, or the grassland management method (i.e., cut, grazed or cut + grazed). Managed grasslands receiving 301-400 kg N ha^{-1} yr^{-1} had a significantly lower annual NEP than those receiving 1-100 and 101-200 kg N ha^{-1} yr^{-1}. The C sink activity of managed grasslands receiving a high amount of N fertiliser may be attributed to high rates of fertiliser addition stimulating vegetation growth and therefore C uptake (Liu et al., 2019). Many of the environmental and management variables considered in the study had no significant influence on the annual NEP of either croplands or managed grasslands. It was expected that soil type would influence annual NEP due to the relationship between soil clay content and CO_2 emissions (Mangalassery et al., 2015; Prout et al., 2022), and that sites subject to conservation tillage would have lower CO_2 emissions, and thus annual NEP, as a result of less soil disturbance and SOM decomposition (Smith, 2004; Stavi and Lal, 2013). Additionally, residue retention, the inclusion of cover crops and grassland management method were expected to significantly influence annual NEP as a result of these factors heavily controlling C_H and C_I (Rutledge et al., 2015; Carswell et al., 2019; Cates and Jackson, 2019; Nunes et al., 2020), however this was not found. The lack of significant influence of many of these variables on the annual NEP of croplands and managed grasslands can primarily be attributed to the dataset being limited both spatially and temporally.

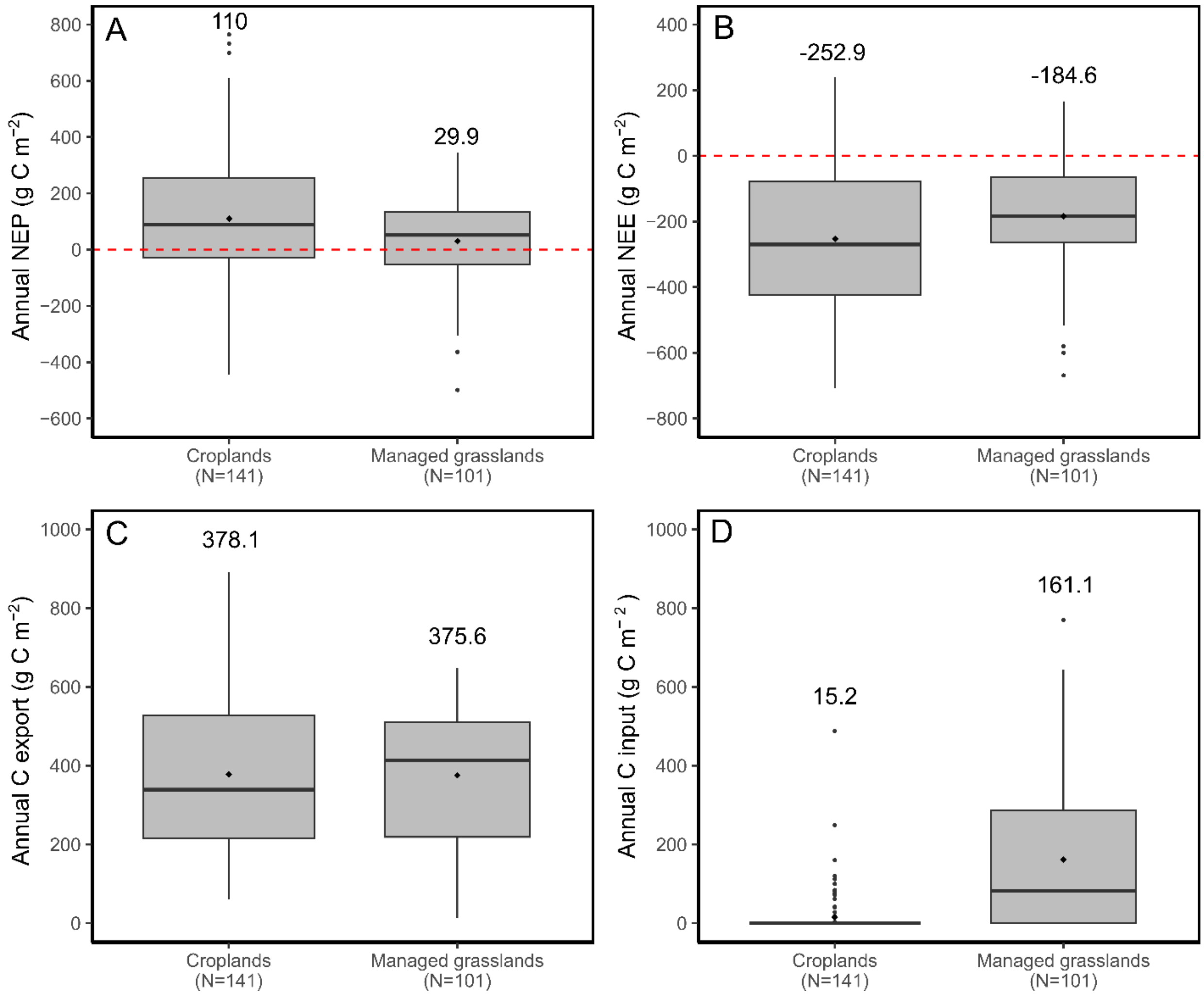

FIGURE 7.1 BOXPLOTS SHOWING THE RANGE OF REPORTED (A) ANNUAL NEP, (B) ANNUAL NEE, (C) ANNUAL C EXPORT AND (D) ANNUAL C INPUT VALUES FOR THE CROPLANDS AND MANAGED GRASSLANDS DATA. THE WIDTH OF EACH BOXPLOT IS PROPORTIONAL TO THE NUMBER OF SAMPLES IN EACH GROUP; THE DIAMOND WITHIN EACH BOX AND THE VALUE ASSOCIATED WITH EACH BOX REPRESENT THE MEAN OF THE GROUP. NOTE THE SCALE FOR EACH PLOT. FOR PLOTS A AND B, POSITIVE VALUES INDICATE C LOSS FROM AND NEGATIVE VALUES INDICATE C ACCUMULATION WITHIN THE AGROECOSYSTEM.

7.1.2 The effect of soil type on NEE and NEP of maize grown for bioenergy in the UK

The aim of Chapter 3 was to measure the NEE and NEP of maize grown for bioenergy in the UK on contrasting soil types to answer the question "How does soil type affect the NEE and NEP of maize grown for bioenergy in the UK?". Net ecosystem exchange was measured with

EC and NEP calculated over the maize growing season at two sites in the UK, one with mineral soil and one on previously drained lowland peat. The results showed that both sites behaved as C sources during the maize growing season and that C loss was over twice as high from the peat site (PS) than the mineral site (MS) (Figure 7.2). The growing season NEP of maize grown at MS and PS is within the range reported in other published studies (Figure 7.2), however it is important to note that all these studies are from sites with mineral soil and that there are no published studies reporting the growing season NEP of maize grown on peat. The NEP of MS is lower than that of most other published studies that, similarly to this study, exported wholecrop maize. This is likely due to the NEE in our study being similar but C_H being comparatively lower than that reported in the published literature (Figure 7.2). The *in-situ* net CO_2 uptake as NEE at MS was over twice that at PS (Figure 7.2). When partitioning NEE into gross primary productivity (GPP) and TER, GPP was similar at both sites but TER was considerably higher at PS meaning that NEE was less negative at this site. Previous studies have shown that peatlands used for agricultural production have high TER due to the lowering of the water table and peat decomposition following drainage (Lohila et al., 2003; Evans et al., 2021). Maize yield was high at both sites, although the C content of the maize at MS was marginally higher than that at PS. The higher C_H at MS was counteracted by the more negative NEE at that site meaning that overall C losses were greater at PS.

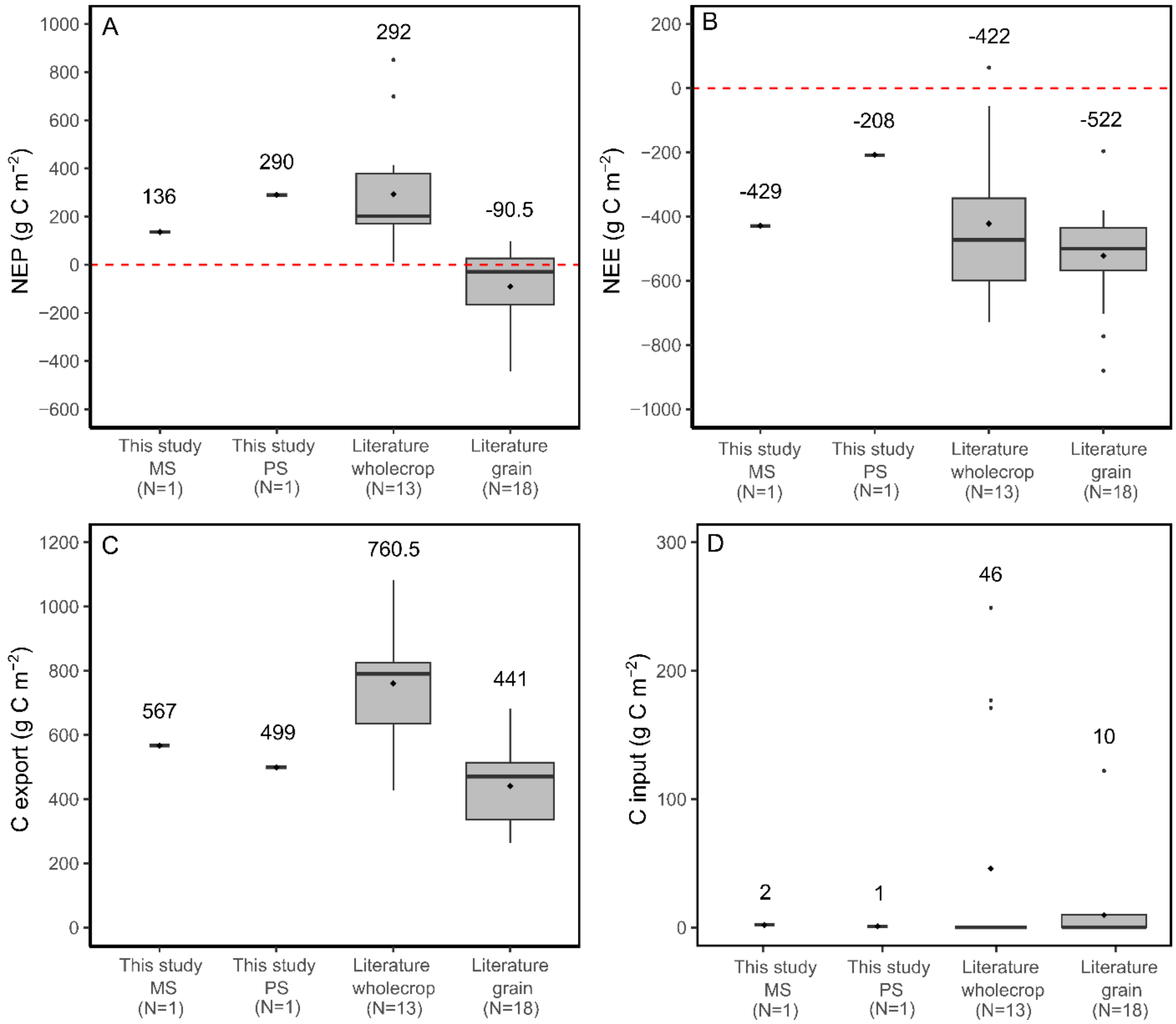

FIGURE 7.2 BOXPLOTS SHOWING THE RANGE OF REPORTED (A) NEP, (B) NEE, (C) C EXPORT AND (D) C INPUT VALUES FOR MAIZE IN THIS STUDY (MS = MINERAL SITE AND PS = PEAT SITE) AND THROUGHOUT THE LITERATURE (SPLIT BY WHETHER WHOLECROP MAIZE OR GRAIN ONLY WAS HARVESTED). THE WIDTH OF EACH BOXPLOT IS PROPORTIONAL TO THE NUMBER OF SAMPLES IN EACH GROUP; THE DIAMOND WITHIN EACH BOX AND THE VALUE ASSOCIATED WITH EACH BOX REPRESENT THE MEAN OF THE GROUP. NOTE THE SCALE FOR EACH PLOT. FOR PLOTS A AND B, POSITIVE VALUES INDICATE C LOSS FROM AND NEGATIVE VALUES INDICATE C ACCUMULATION WITHIN THE AGROECOSYSTEM.

7.1.3 The effect of crop type on NEE and NEP of UK soil

The aim of Chapter 4 was to measure and compare the NEE and NEP of a cropland over 2.5 years to answer the question “How does crop type affect the NEE and NEP of agricultural land in the UK?”. The results showed that C was being lost during the maize growing season whereas the field was behaving as a C sink during the winter wheat and vining pea growing seasons (Figure 7.3). Other studies have also found that croplands with a maize-wheat rotation behave as C sources during the maize growing season and as C sinks during the winter wheat growing season (Buysse et al., 2017; Poyda et al., 2019). The growing season NEP for maize and winter wheat are well within the range reported by other studies measuring the growing season NEP of these crops (Figure 7.3). There are no published studies reporting the growing season NEP of vining pea to compare our results with. The *in-situ* net CO_2 uptake as NEE was the least negative during the vining pea growing season followed by maize and winter wheat (Figure 7.3). Imports of C were low during the maize and vining pea growing seasons and were considerably higher for winter wheat due to two applications of pig slurry that the crop received in the spring (Figure 7.3). At the end of the vining pea growing season only the pea pods were harvested, with the residues left in the field, whereas for maize and winter wheat the whole-crop was harvested (i.e., grain and straw). This meant that vining pea had the lowest C_H of the three crops, which explains the greatest C sink behaviour for this crop when considering NEP during its growing season. Although C_H was greatest from winter wheat, it was smaller than its C_I and NEE combined, so the crop behaved as an overall C sink during its growing season. The C_H of maize was greater than its NEE and C_I combined, hence the field acted as a C source during the maize growing season. When summing the NEP for the three crop growing seasons, CF shows an overall C uptake, however when accounting for NEP during the fallow periods following the crop growing seasons, CF behaved as a large C source over the 2.5 year measurement period. This highlights the importance of considering fallow periods in NEP calculations, as the absence of vegetation and the decomposition of crop residues on the soil surface following a crop harvest can have a considerable impact on annual and growing season NEP values.

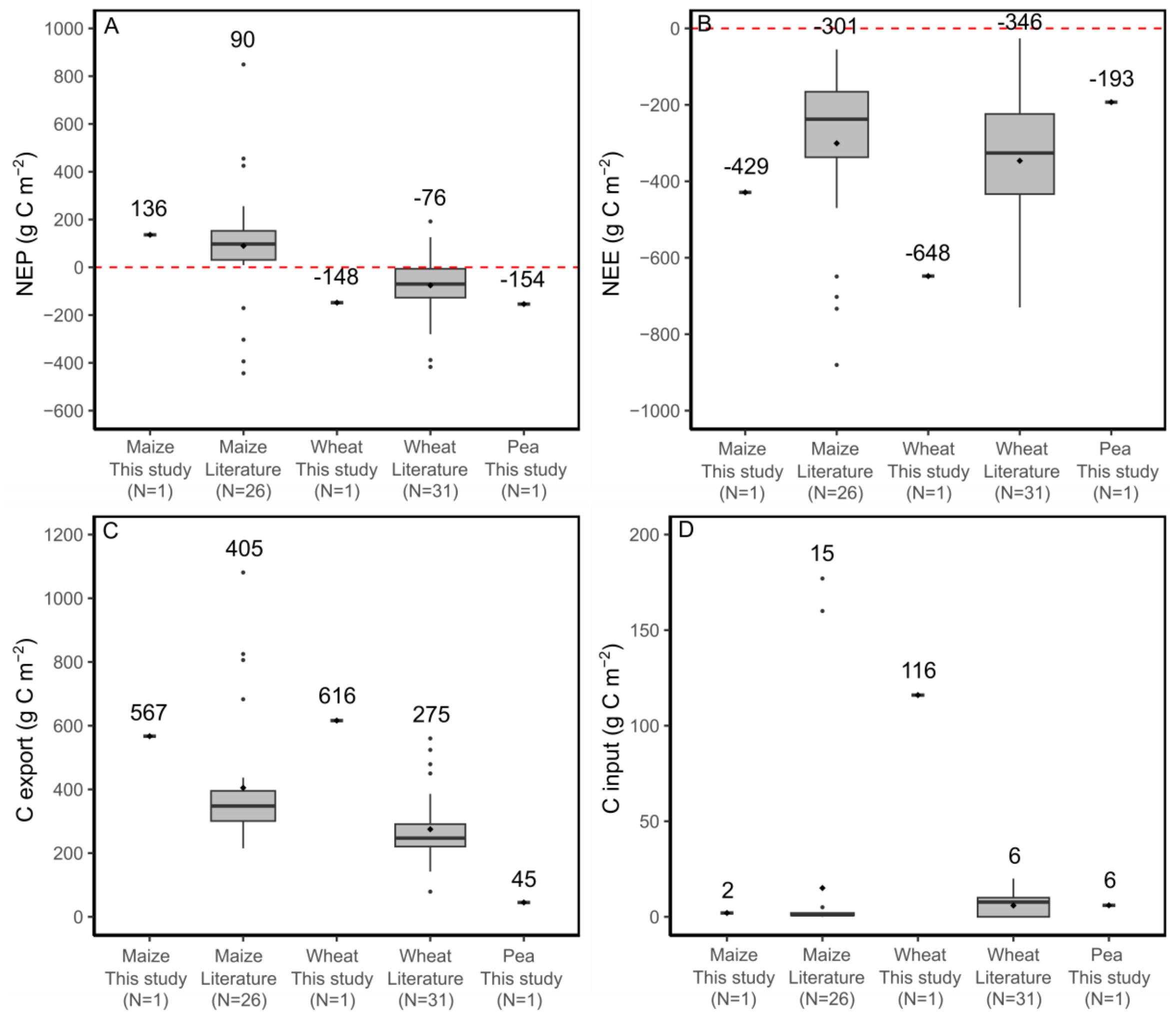

Figure 7.3 Boxplots showing the range of reported (A) NEP, (B) NEE, (C) C export and (D) C input values over maize, winter wheat and pea growing seasons in this study and in the published literature. The width of each boxplot is proportional to the number of samples in each group; the diamond within each box and the value associated with each box represent the mean of the group. Note the scale for each plot. For plots A and B, positive values indicate C loss from and negative values indicate C accumulation within the agroecosystem

7.1.4 The effect of agricultural land use on NEE and NEP of UK soil

The aim of Chapter 5 was to measure and compare the NEE and NEP of a neighbouring cropland and permanent pasture over one year to answer the question “How does agricultural land use affect the NEE and NEP of agricultural land in the UK?”. During the one-year measurement period, the cropland (CF) consisted of a winter wheat growing season and subsequent fallow period and the permanent pasture (PP) was periodically grazed by sheep and was cut once for silage. The results showed that PP was behaving as a C source and that CF was a small C sink (Figure 7.3). There are no published studies comparing the annual NEP of a neighbouring cropland and managed grassland for the results of this study to be compared with, in the UK or globally. When considering the agroecosystems independently, however, the annual NEP of PP is considerably higher than that reported by the literature for most other cut and grazed grasslands in temperate climates (Figure 7.4). Poyda et al. (2019) report the annual NEP of a cropland over multiple years, where the management includes a winter wheat growing season and fallow period; the NEP of CF fits well within the range reported by Poyda et al. (2019) (Figure 7.4). Net ecosystem exchange showed *in-situ* CO_2 uptake in CF whereas PP had a small CO_2 emission (Figure 6.3). The GPP of CF was higher than that of PP and TER was higher in PP than in CF, meaning that less of the GPP was respired as TER in CF (i.e., NEE was most negative). Continuous defoliation and respiration from grazing livestock in PP explain this difference (Soussana et al., 2007; Senapati et al., 2014; Rutledge et al., 2015). Carbon imports were higher to CF than PP, although C_H from CF was also considerably higher than that from PP (Figure 7.4).

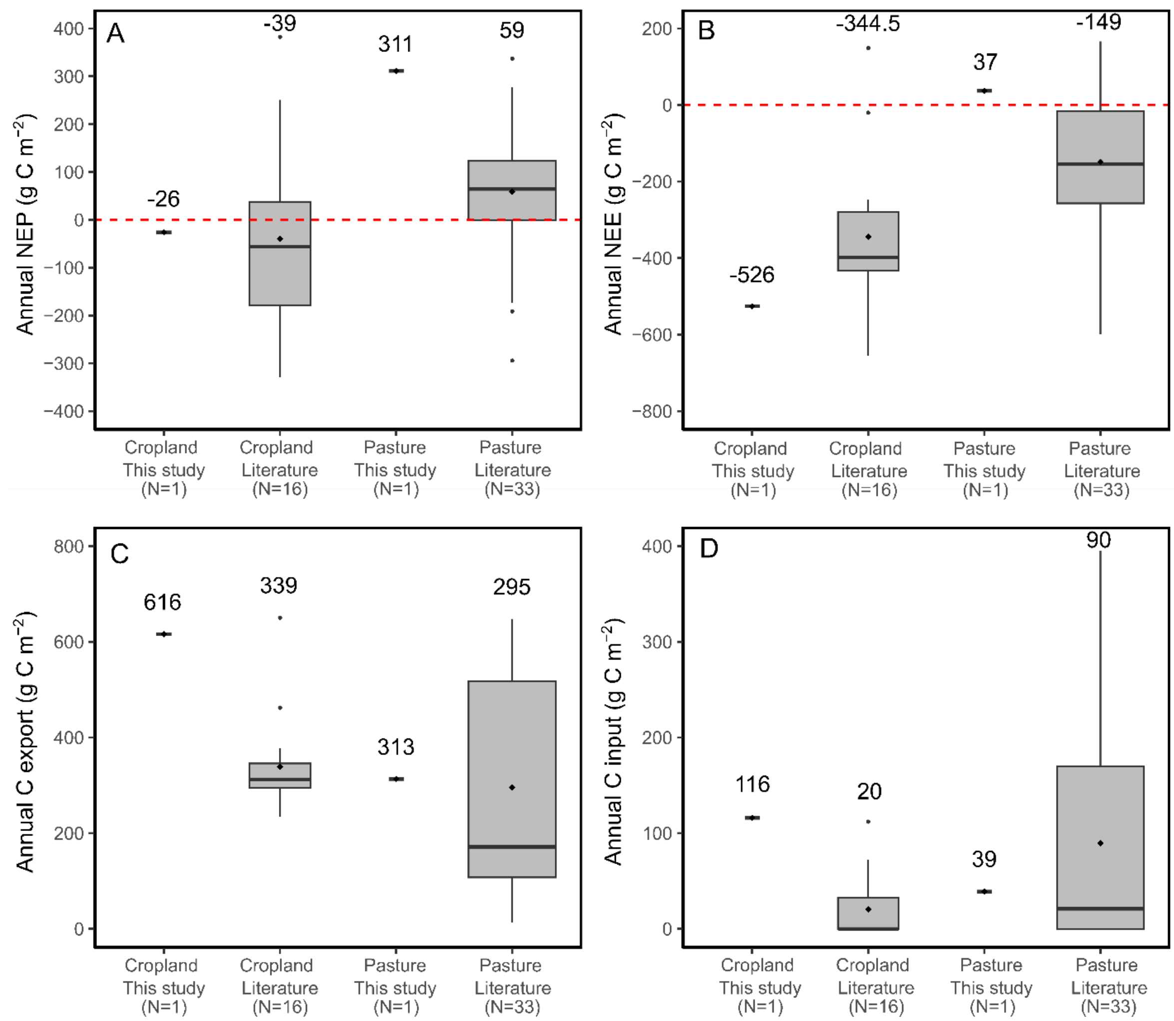

FIGURE 7.4 BOXPLOTS SHOWING THE RANGE OF REPORTED (A) ANNUAL NEP, (B) ANNUAL NEE, (C) ANNUAL C EXPORT AND (D) ANNUAL C INPUT VALUES FOR CROPLANDS (INCLUDING WINTER WHEAT GROWING SEASON AND FALLOW PERIODS) AND FOR CUT AND GRAZED GRASSLANDS IN TEMPERATE CLIMATES IN THIS STUDY (CF = CROPLAND THIS STUDYAND PP = PASTURE THIS STUDY) AND IN THE PUBLISHED LITERATURE. THE WIDTH OF EACH BOXPLOT IS PROPORTIONAL TO THE NUMBER OF SAMPLES IN EACH GROUP; THE DIAMOND WITHIN EACH BOX AND THE VALUE ASSOCIATED WITH EACH BOX REPRESENT THE MEAN OF THE GROUP. NOTE THE SCALE FOR EACH PLOT. FOR PLOTS A AND B, POSITIVE VALUES INDICATE C LOSS FROM AND NEGATIVE VALUES INDICATE C ACCUMULATION WITHIN THE AGROECOSYSTEM.

7.1.5 The effect of fertiliser type on GHG fluxes from mineral soil in the UK

The aim of Chapter 6 was to measure N_2O and CH_4 fluxes from winter wheat grown on mineral soil in the UK amended with different fertilisers to answer the question “How does fertiliser type influence GHG fluxes from a winter wheat crop grown on a mineral soil in the UK?”. Automated flux chambers were used to measure GHG emissions from winter wheat fertilised with three treatments: (1) inorganic fertiliser (IF), (2) untreated pig slurry and inorganic fertiliser (UPS), and (3) plasma-treated pig slurry and inorganic fertiliser (TPS). The application of a combination of both untreated pig slurry and inorganic fertiliser to agricultural croplands is common practice for mixed arable and pasture farms, as the application of livestock waste adds organic matter (OM) to the soil and improves on-farm waste management (Ruser and Schulz, 2015; Willen et al., 2016). The use of livestock waste as an organic fertiliser releases significant emissions of CH_4 and N_2O and so innovative management practices such as plasma induction are being developed to reduce these GHG emissions. The results showed that overall CO_2-equivalent emissions from winter wheat were higher when fertilised with TPS compared to UPS and IF (Figure 7.5). There was no difference in winter wheat yield between the treatments, so yield-scaled CO_2-equivalent emissions also followed this pattern. There is a lack of data reporting CO_2-equivalent emissions from winter wheat fertilised with treated or untreated pig slurry, however the CO_2-equivalent emissions from the winter wheat fertilised with IF in the study are within the range reported by the published literature (Figure 7.5). Total N_2O emissions were higher from winter wheat fertilised with TPS than UPS and IF. Total CH_4 emissions were higher from winter wheat fertilised with UPS than TPS and IF. The considerably higher global warming potential of N_2O compared to CH_4 therefore resulted in the winter wheat fertilised with TPS having the highest CO_2-equivalent emissions (Figure 7.5). Diurnal emissions of N_2O were also observed from all treatments towards the end of the winter wheat growing season, which has also been reported by other studies (Wu et al., 2021). These diurnal emissions were driven by variations in water-filled pore space (WFPS) and photosynthetically active radiation (PAR) which controlled denitrification, and thus N_2O emission.

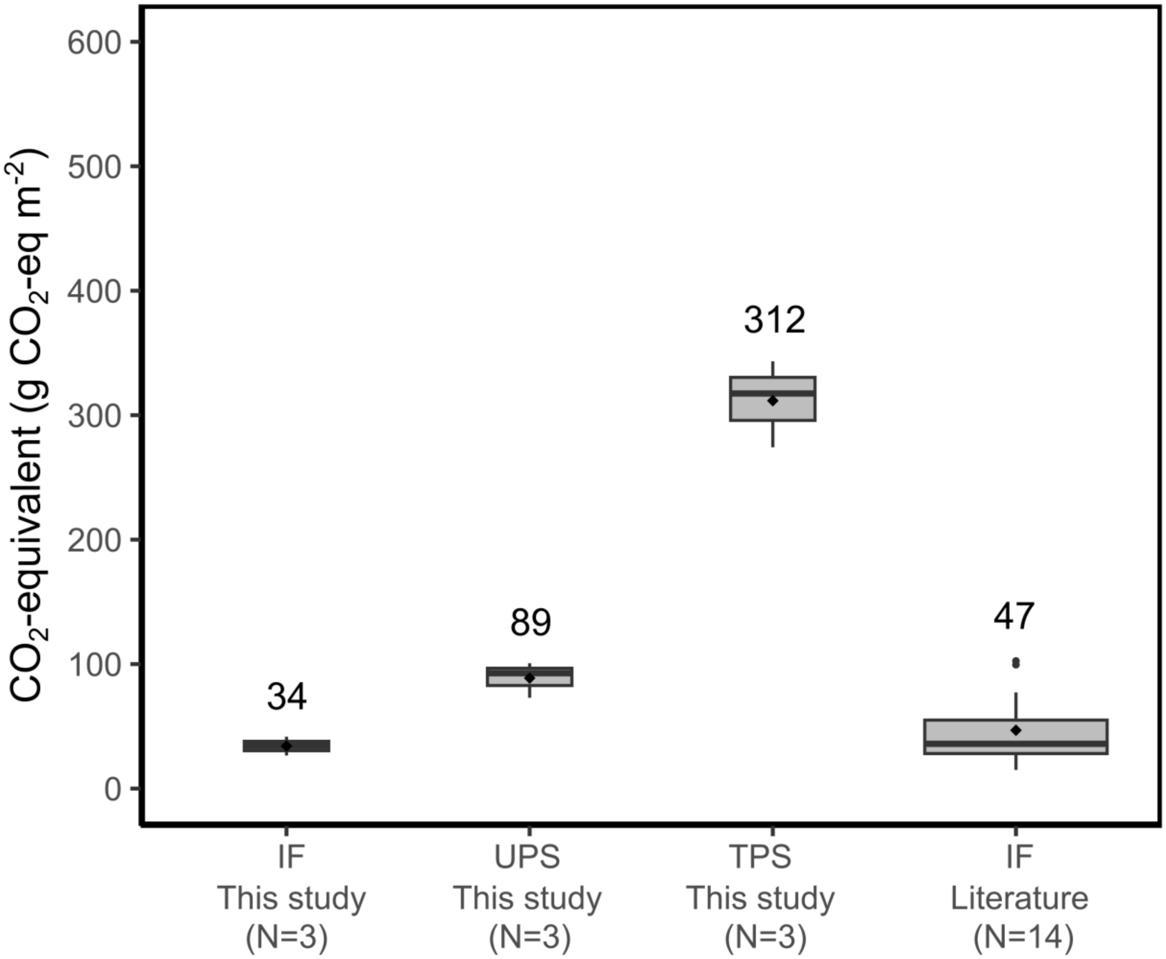

FIGURE 7.5 BOXPLOTS SHOWING THE RANGE OF REPORTED CO_2-EQUIVALENT EMISSIONS (CH_4 AND N_2O ONLY) FROM WINTER WHEAT FERTILISED WITH INORGANIC FERTILISER (IF), UNTREATED PIG SLURRY (UPS) AND PLASMA-TREATED PIG SLURRY (TPS) IN THIS STUDY AND FERTILISED WITH 150-300 KG N HA^{-1} IN THE FORM OF INORGANIC FERTILISER MEASURED IN THE PUBLISHED LITERATURE.

7.2 Implications of research findings

The following sections provide a synthesis of the results in context of the wider policy and future research, in response to Question 6.

This thesis presents some of the first measurements of NEP from agriculturally managed soils in the UK, and of GHG emissions following the application of plasma-treated pig slurry to land. These results have considerable implications for UK policy and the management of agricultural soils for reduced C loss and GHG emissions. The actions required to achieve the environmental goals outlined in the UK's Environment Improvement Plan (EIP): improving nature, improving environmental quality, improving resource use, improving climate change mitigation, improving biosecurity and enhancing the natural environment (Figure 7.6) (UK Government, 2023b) are incorporated into a range of policies implemented by the UK Government. The results of this research have particular implications on the Net Zero Government Initiative (DESNZ, 2023a) and Circular Economy Package (CEP) (UK Government, 2020), the Environmental Land Management (ELM) schemes – specifically the Sustainable Farming Incentive (SFI) – which are part of the Agricultural Transition Plan (DEFRA, 2023a; DEFRA, 2023b; DEFRA, 2024d), the Biomass Strategy (DESNZ, 2023b) and the England Peat Action Plan (UK Government, 2021a). Furthermore, the data will be applicable to future climate change modelling, to predict the impacts of climate change on the major aspects of the C balance, which will aid farmers in adopting appropriate management practices to adapt to the impacts of climate change.

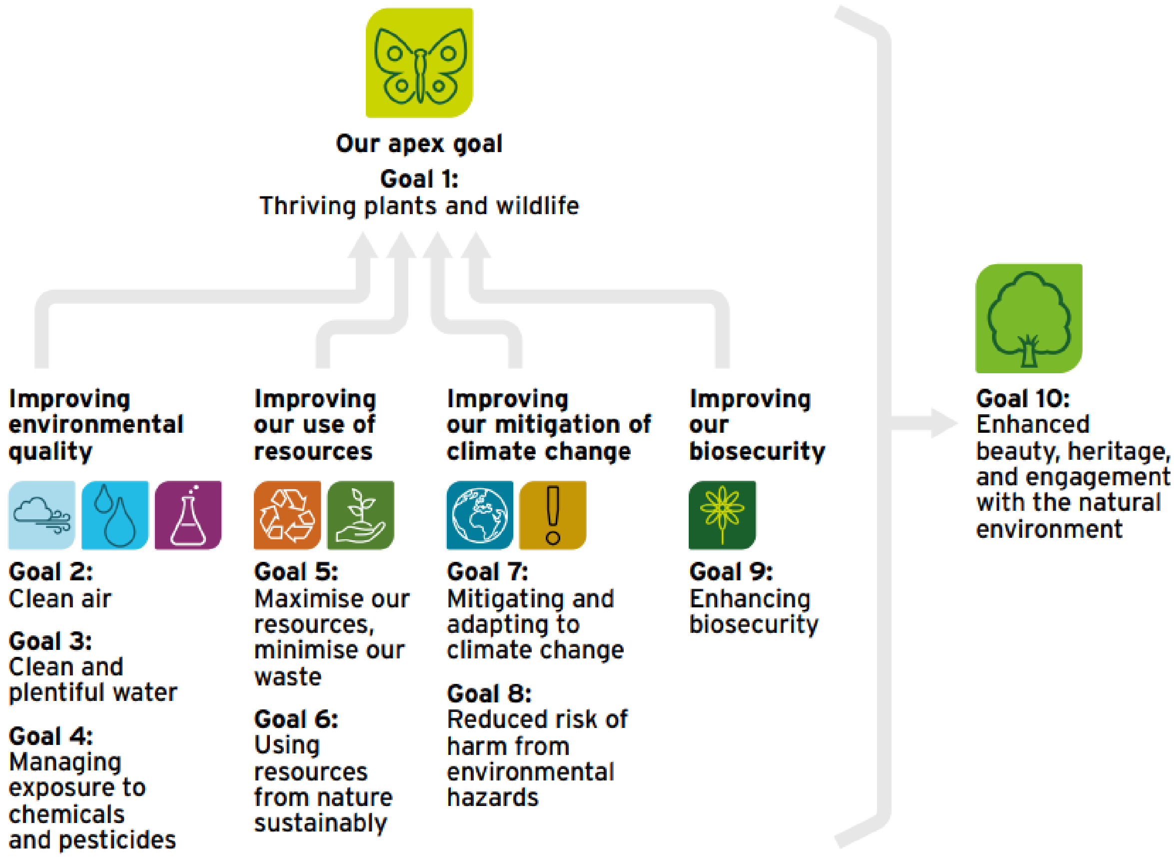

Figure 7.6 Environmental goals outlined in the Environmental Improvement Plan (UK Government, 2023a).

7.2.1 Implications for achieving net zero

The UK's Net Zero Government Initiative focuses on the country's target to reduce its GHG emissions by 100 % from 1990 levels by 2050 (DESNZ, 2023a). The Net Zero Growth Plan (DESNZ, 2023c) and the corresponding Carbon Budget Delivery Plan (UK Government, 2023b) outline the changes that sectors of the UK economy can make to achieve this. The Carbon Budget Delivery Plan provides a comprehensive list of actions that can be implemented throughout the agricultural sector to reduce GHG emissions, many of which are related to and incentivised as part of the SFI and link to other UK policies. The findings of this research provide evidence to support the implementation of many of the measures promoted to achieve net zero (Table 7.2), however it is also important to consider the potential trade-offs that may arise from their adoption. The growth of cover crops during fallow periods, for example, is included in the SFI Arable and Horticultural Soils Standard as a method of reducing soil erosion and CO_2 emission, both of which are more likely when the soil is left bare (DEFRA, no date c), as evidenced by the positive NEE and NEP measured during the fallow periods in CF in Chapter 4. Cover cropping during fallow periods may not reduce NEP at every site, however; the meta-analysis (Chapter 2) showed that 33 % of global croplands with cover crops were accumulating C and that 67 % were losing C. Therefore, the fate of cover crop residues should be carefully considered so as not to displace the CO_2 emissions saved by growing cover crops during fallow periods. In line with the CEP, the EIP (UK Government, 2023a) recognizes the importance of utilizing livestock manures and slurries as organic fertilisers to both provide benefits to the soil and to promote an on-farm circular economy by reducing waste. The EIP has a particular focus on the management of livestock wastes for ammonia (NH_3) reduction (UK Government, 2023a); a reduction in NH_3 emissions is required, although non-CO_2 GHGs such as CH_4 and N_2O cannot be ignored. Plasma induction is being explored as a method to reduce GHG emissions from livestock wastes, and has been shown to reduce NH_3 emissions relative to untreated livestock waste (Kupper et al., 2020; Gillbard, 2023). The results of Chapter 6, however, show that, relative to an untreated pig slurry, the application of plasma-treated pig slurry to winter wheat increased soil N_2O emissions to such an extent that TPS was more polluting on a CO_2-equivalent basis (N_2O + CH_4) than UPS and IF. The experimental study (Chapter 6) did not consider NH_3 emissions, however, which highlights the need for studies to consider N_2O, CH_4 and NH_3 together when measuring the influence of innovative fertiliser technologies on soil emissions. These results also highlight the need for life-cycle analyses to be conducted, which would consider the impacts of treatments on upstream emissions as well as those originating on-field, to fully evaluate the extent to which actions can contribute to net zero.

TABLE 7.2 RELATIONSHIPS BETWEEN MEASURES RECOMMENDED BY THE CARBON BUDGET DELIVERY PLAN (UK GOVERNMENT, 2023B) AND THE FINDINGS OF THIS RESEARCH. INFORMATION IN BRACKETS REFERS TO THE POLICY NUMBER (#) AND THE DATE OF IMPLEMENTATION.

Government policy	Project findings
Grow cover crops in rotation during fallow periods (#173, implemented 2022)	Chapters 4 and 5 show substantial C loss from a cropland during fallow periods; this policy therefore looks useful however evidence is needed on whether C losses are reduced/avoided by growing cover crops. Chapter 2 shows that cover crops need to be carefully managed for successful climate change mitigation and to avoid displacing CO_2 emissions.
Integrate leys into arable rotations (#160, implemented 2024)	Chapters 4 and 5 show substantial C loss from a cropland during fallow periods; this therefore looks useful however evidence is needed on whether growing leys in this period instead may reduce or avoid these C losses. Chapters 4 and 5 also show that managed grasslands do not necessarily have lower SOC loss/higher CO_2 uptake than croplands and so grass leys should be managed to avoid further C loss.
Analyse manure prior to application (#159, implemented 2022); avoid excessive N use by developing a nutrient management plan (#161, implemented 2022)	Chapter 6 highlights the importance of ensuring that the nutrient content of organic fertilisers is analysed prior to application to avoid excessive N supply which can cause N_2O emission, although more research is required.
Use of nitrification inhibitors (#167, implemented 2022)	In response to the results of Chapter 6, further research into how nitrification inhibitors could be paired with innovative

	technologies such as plasma induction to reduce both ammonia (NH_3) and non-CO_2 GHG emissions is recommended.
Plant perennial bioenergy crops (#177, to be implemented 2026)	The results of Chapter 2 show that annual bioenergy crops were C sources; this government policy may offer a solution to reducing these C emissions, although more evidence is required to establish this.
Responsible peatland management by raising water tables and promoting wetter farming (#179, to be implemented 2025)	Chapter 2 provides further evidence for the argument that peatland should not be used for intensive agricultural production and instead should be managed in ways that facilitate restoration for future C gain.

7.2.2 Implications for Environmental Land Management schemes

As part of the SFI, one of the main ELM schemes, farmers are paid for the implementation of a number of 'environmentally-friendly' farming practices, such as growing a multi-species cover crop in winter, maintaining a legume fallow that is not grazed or fertilised and restricting the amount of organic and inorganic fertilisers that are applied to grassland (DEFRA, 2023b; 2024c; 2024d). These practices aim to protect the soil surface, increase root density, maintain soil structure, and minimise nutrient pollution, with the underlying goals of increasing soil C sequestration and reducing GHG emissions (DEFRA, 2023c; 2024c; 2024d; no date d), and have been shown by multiple studies to be successful at doing so relative to alternative intensive methods (Malhi et al., 2011; Laird and Chang, 2013; Zhang et al., 2015; Chen et al., 2019). The results of Chapters 2-6 show that the effectiveness of these best management practices can be highly variable, however, due to local climate conditions and soil type, with these factors affecting soil microbial activity and SOM decomposition rate (Beziat et al., 2009; Lopez-Garrido et al., 2014; Maia et al., 2019; Bandaru, 2022; Prout et al., 2022). The results of the meta-analysis (Chapter 2), for example, showed that 69 % of monitored global croplands and 65 % of global agricultural grasslands are losing C on an annual basis, despite many of these sites being managed with best practices such as residue retention and cover cropping. Globally, 31 % of croplands with residue retention were gaining C and 69 % were losing C, and 33 % of croplands with cover crops were gaining C and 67 % were losing C (Chapter 2). The results presented in Chapter 5 show that over one year a cut and grazed pasture was losing C, whereas a cropland growing winter wheat followed by a fallow period had a small C uptake. Chapter 4 shows that over a 2.5-year period the cropland was losing C, however, which was mainly attributed to the C loss over multiple fallow periods outweighing the C uptake by vegetation during crop growing seasons.

The success of practices incentivised as part of the SFI will therefore vary by site due to variations in the climate and soil type across the UK. In response to this, policy recommendations and agri-environment schemes (AES) should take a targeted approach based on the regional soil and climate conditions, and any other management practices being used. The research conducted by this project has highlighted the significant lack of data on C fluxes from agriculturally managed soils in the UK. Furthermore, the conclusions made as part of this project are primarily based on results which have been measured on an annual basis, and therefore internanual variability is not accounted for, although it is acknowledged that this will be a crucial element of future research. Long-term evidence from multiple sites across the country would allow more evidence-based decisions and targeted policy recommendations. In addition, the actions that farmers are incentivised for should be based on the impacts of a practice on both C fluxes and GHG emissions; these should not be considered separately as there is the potential for trade-offs or pollution swapping to occur. The addition of (untreated) pig slurry as an organic fertiliser to winter wheat in the cropland in Chapters 4 and 5 contributed to the negative NEP (i.e., C uptake) of the field. The results of Chapter 6 show that the application of untreated pig slurry to winter wheat, however, causes an emission of N_2O and CH_4, which is likely to have offset some or all of the C sequestered by the field.

7.2.3 Implications for the Biomass Strategy

The results of Chapters 3 and 4 show that growing maize for bioenergy may not be a reliable climate change mitigation strategy or a viable path to achieving net zero as maize cultivation results in SOC loss, particularly on peat soils. Maize is one of the most popular bioenergy crops grown in the UK (DEFRA, 2020c) and is used to generate energy via anaerobic digestion (AD) (DESNZ, 2023b). Bioenergy crops are viewed as a sustainable alternative to fossil fuels based on the principle that the crops take up CO_2 during growth which can then be used to displace fossil fuels when harvested and transformed into energy (DESNZ, 2023b). Furthermore, bioenergy crops can also facilitate C sink behaviour by storing C in the soil during the growing season (DESNZ, 2023b). The results of the monitoring studies in Chapters 3 and 4, however, show that growing maize did not result in C accumulation by a cropland during its growing season or the fallow period following whole-crop harvest. Although maize takes up CO_2 while it is growing, this C is exported from the field at harvest and released back to the atmosphere following AD, so is not sequestered into the soil. Some of this C loss would be compensated for providing the residues from AD were returned to the soil as a form of organic fertiliser, a practice encouraged within the Biomass Strategy (DESNZ, 2023b), but this is not done at all sites and did not occur at either of the sites monitored in Chapter 3. Furthermore, growing maize on a drained peatland had over twice the C loss of maize grown on a mineral soil, and so there is a clear need to move away from the use of peat soils in intensive agricultural production. The results of Chapters 3 and 4 align with the UK Government's recommendation to move away from the use of food and feed crops, such as maize, for bioenergy and to instead utilise waste feedstocks to reduce the pressure on food prices and further promote a circular economy (DESNZ, 2023b). It is important to note that the conclusions presented here are related to the growth of maize for bioenergy only, and do not consider C emissions from outside the field boundary (i.e., the AD process itself and transport emissions). These emissions can be accounted for by life-cycle analysis, which is identified as a future research priority in Section 7.4.

7.2.4 Implications for the England Peat Action Plan

The England Peat Action Plan highlights the extent to which peatlands in England have been degraded as a result of drainage for intensive agriculture, and details how this degradation can be reversed to reduce C emissions, improve water quality and flood mitigation and benefit nature (UK Government, 2021a). In response to this, the Lowland Agricultural Peat Task Force was established in 2021 by DEFRA to develop sustainable management regimes which would facilitate peat C preservation whilst ensuring profitable agriculture (DEFRA, 2023d). The Lowland Agricultural Peat Task Force Chair's Report, published in 2023, outlines 14 recommendations for the more sustainable management of lowland peat, including subsidising farmers for raising the water table level, moving to farming practices which compliment wetter production and investing in the research and development of water tolerant food crops with a low C footprint (DEFRA, 2023d). These recommendations align with the consensus throughout the literature that peatlands should not be used for intensive agricultural production and that instead drained peatlands should be sustainably managed to replenish previously lost C (Nursyamsi et al., 2016; Freeman et al., 2022; Lloyd et al., 2023e). The results of Chapter 3 show that C loss from maize grown on peat was double that from maize grown on mineral soil; the higher C loss from the peat site was driven by its higher TER and thus less negative NEE. These results highlight that the continued usage of peatlands for intensive agricultural production is unsustainable, and support the recommendations of the Lowland Agricultural Peat Task Force to manage peat less intensively and raise the water table level to reduce CO_2 emissions. The development of incentives for farmers who farm on lowland peat will be critical to achieving this.

7.2.5 Recommendations for policy

The findings of this research have substantial implications for existing and developing UK policy, particularly with relation to how soils should be managed for reduced GHG emissions and increased soil C storage. There is a clear need for more research to be conducted to strengthen the evidence base to inform policy decision making on which management practices should be utilised in certain areas of the UK based on the environmental conditions, and for a requirement to simultaneously measure C fluxes and GHG emissions. Based on the results of this research, policymakers should (continue to) consider research into the success and feasibility of the following incentivised policy actions:

- Moving away from supporting annual crops for bioenergy production and instead grow alternative bioenergy crops on marginal land.
- Encouraging the application of organic amendments to croplands; a reduction in NEP requires an increase of C_I by the addition of organic amendments. It will be important to consider the potential risks associated with livestock waste application regarding increased N_2O and CH_4 emissions, however, and the potential content of pharmaceuticals and emerging contaminants (Gworek et al., 2021).
- Encouraging the sustainable management of lowland peat, including raising the water table level; the results of this thesis show that growing maize on drained lowland peat is undesirable with regards to its environmental impacts.
- Limiting the length of fallow periods between crops, which could be combined with existing SFI actions to encourage the growth of cover crops, although more research is needed into the termination of cover crops and how their decomposition influences TER, NEE and NEP.

7.3 Limitations of the research

This thesis provided new results on how soil type, climate and agricultural management practices influence the NEP of global and UK croplands and agriculturally managed grasslands, and on the influence of fertiliser type on GHG emissions from winter wheat grown in the UK. There are limitations associated with the work that must be considered, however.

This thesis was primarily focussed on evaluating the C sink or source strength of agricultural soils at the field scale as a result of the local environmental conditions and management practices. To measure this, NEP was calculated. Net ecosystem productivity considers NEE, C_I as seed, organic fertiliser or livestock excreta, and C_H as harvested or grazed aboveground biomass (Evans et al., 2021). Net ecosystem C balance (NECB) builds on this by also considering smaller C fluxes such as dissolved organic C, C in precipitation and C in CH_4 (Ciais et al., 2010b; Smith et al., 2010), and thus provides a more accurate estimation of the C dynamics of an agroecosystem. When conducting the literature search for the global meta-analysis (Chapter 2) it was clear that, amongst the studies measuring C fluxes in agriculture, NEP is reported much less than NEE, and NECB even less so. Of the 719 total publications that resulted from the key word search, only 52 were associated with the term 'net ecosystem carbon balance' (Chapter 2, Table 2.1). The additional C fluxes required for the calculation of NECB relative to NEP therefore present a considerable barrier for researchers to obtain more accurate estimates of the C balance of agroecosystems. This is likely to be due to the challenges associated with calculating such small fluxes. As NEP is used more frequently than NECB across the literature, and is easier to measure, NEP was calculated rather than NECB throughout this project to estimate whether the sites in the monitoring studies were losing or accumulating C. It is acknowledged that whilst the results may therefore not be as accurate as they could theoretically be, as the smaller fluxes considered in NECB are not included, they represent a good estimate of whether a field is accumulating or losing C and provide data to fill the knowledge gap surrounding C fluxes from UK agricultural soils. Furthermore, the estimations of C_I and C_H at the study sites throughout this thesis involve some assumptions, for example the percentage of C returned to a field as dung via grazing livestock. Whilst these considerations may increase the uncertainty surrounding the results, the calculation of NEP of the croplands and managed grasslands in this study still offers a valuable, urgently needed and novel insight into the C dynamics of these agricultural sites in the UK.

It is important to highlight the discrepancies in the terminology used throughout the literature to describe C fluxes from agri-ecosystems. Aside from NEE, which is well established as the the difference between the CO_2 flux assimilated by photosynthesis (GPP) and respired from plant and soil processes (TER) (Eugster and Merbold, 2015), there is considerable variation between the definitions of other terminology used throughout the field, primarily based on the scale of flux measurements and the consideration of lateral fluxes in addition to NEE. Figure 7.7 illustrates an example of the discrepancies between the definitions of the same terminology from three sources, and Table 7.3 provides an example of the range of definitions used for the same terminology throughout the literature. Efforts have been made to establish standardized terminology for use in this field (Chapin III et al., 2010; Smith et al., 2010), however there is still ambiguity, with researchers disagreeing on the use of these terms. Where we use NEP, for example, using the definition also used by IPCC (2000) and Evans et al. (2021), Niu et al. (2021) and Peng et al. (2022) use NBP and Abraha et al. (2018) use NEE_{adj}.

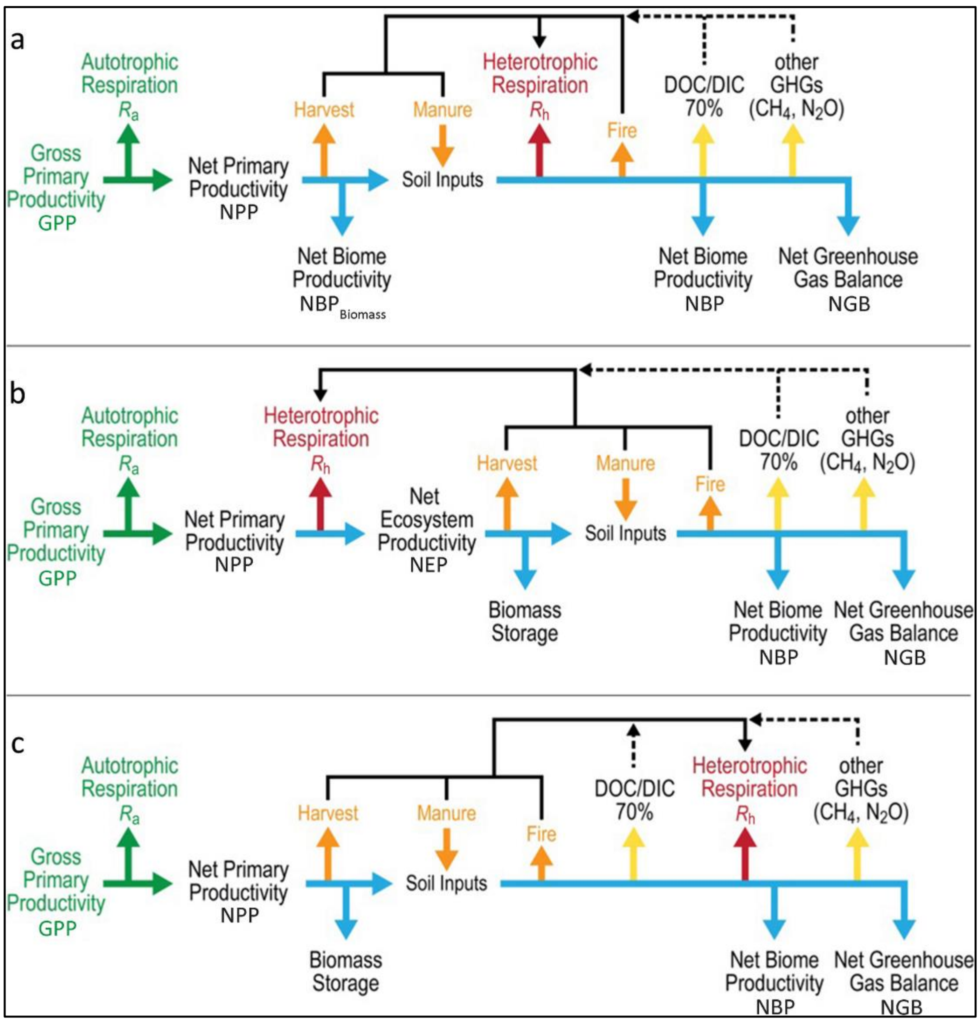

FIGURE 7.7 EXAMPLES OF VARIATIONS IN THE DEFINITIONS OF TERMINOLOGY THROUGHOUT THE LITERATURE. SOURCE: SCHULZE ET AL. (2021).

TABLE 7.3 AN EXAMPLE OF THE VARYING DEFINITIONS USED THROUGHOUT THE LITERATURE TO DESCRIBE CARBON FLUXES IN AGRI-ECOSYSTEMS INCLUDING NET

ECOSYSTEM PRODUCTIVITY (NEP), NET BIOME PRODUCTIVITY (NBP) AND NET ECOSYSTEM CARBON BALANCE (NECB).

Term	Definition	Reference
Net ecosystem production or net ecosystem productivity (NEP)	Accounts for NEE, C_I as seed, organic fertiliser or livestock excreta, and C_H as harvested or grazed aboveground biomas	Evans et al. (2021)
	Net production of carbon by an ecosystem, the difference between the rate of production of living organic matter and the decomposition rate of dead organic matter	IPCC (2000)
	NEP = GPP – autotrophic respiration – heterotrophic respiration	Schulze et al. (2021)
Net biome production or net biome productivity (NBP)	Net production of carbon by a region containing multiple ecosystems, the difference between the rate of production of living organic matter and the decomposition rate of dead organic matter	IPCC (2000)
	Extrapolation of NECB to larger spatial scales	Chapin III et al. (2006)
	$NBP = C_I – C_H – NEE$	Peng et al. (2022)
	$NBP = NEE + C_H – C_I$	Prescher et al. (2010)
	$NBP = -NEE – C_H - C_I$	Niu et al. (2021)
Adjusted net ecosystem exchange (NEE_{adj})	$NEE_{adj} = NEE + C_H$	Abraha et al. (2018)
Net ecosystem productivity with harvest (NEPH)	NEPH = GPP – autotrophic respiration – heterotrophic respiration – C removed as harvested biomass	Schulze et al. (2021)

Net ecosystem carbon balance (NECB)	Accounts for NEE, C_I as seed, organic fertiliser or livestock excreta, and C_H as harvested or grazed abovegrounds biomass, in addition to smaller fluxes of C_I and C_H including: dissolved organic C, volatile organic C, C in CH_4, C in carbon monoxide, particulate C (e.g., C in soot, C in precipitation, C in wind)	Ciais et al. (2010b), Smith et al. (2010), Chapin III et al. (2006)

The sample size of the croplands and managed grasslands datasets in the meta-analysis (Chapter 2) were relatively small (N=141 and N=101 respectively) due to the limited number of publications that met the inclusion criteria. The criteria used in the meta-analysis were selected to ensure that the results from each site would be comparable, and that there were sufficient contextual information about each site to evaluate the influence of climate, soil type and management on NEP. The criteria that publications most frequently failed to meet were: (i) reporting C_I and C_H in addition to NEE, meaning that NEP could not be calculated, and (ii) measuring C fluxes on an annual basis (i.e., many publications reported fluxes over the crop growing season only), meaning that data would likely be biased to show overwhelming C sink activity due to the omission of fallow period fluxes (as highlighted in Chapter 4). The croplands and managed grasslands datasets both lacked global representation, with the majority of measurements from sites in the USA and Germany. This spatial limitation also resulted in there being few measurements from sites with medium and heavy soils and in tropical and arid climates. The lack of significant differences detected between the annual NEP of sites due to climate classification, soil type or agricultural management practice can likely be attributed to the inconsistency in sample size of several of the variables. Across the literature, there were also large inconsistencies in the meta-data that were reported alongside C fluxes. Very few publications, for example, provided information on irrigation management at the study site, so this could not be included as a potential control on NEP, however it is acknowledged that irrigation management may influence NEE (Lee et al., 2009; Laubach et al., 2019; Franco-Luesma et al., 2020a), and therefore NEP. Lastly, the results of the meta-analysis are partly limited by the lack of specificity in how some of the climate, soil, and management variables have been reported and subsequently grouped. The soil data, for example, were grouped into very broad categories (i.e., ‘light’, ‘medium’ and ‘heavy’) because not all publications reported soil data in sufficient detail to include a more specific textural classification. There is a clear need to conduct more studies that measure NEP on an annual basis, reporting C_I and C_H with NEE, and including sufficient meta-data to allow for more robust analysis.

Gap-filling is an established method to complete datasets with missing data (Reichstein et al., 2005; 2016; Dorich et al., 2020; Lucas-Moffat et al., 2022), however the smaller percentage of data that is gap-filled, the closer to reality the dataset is. The method was applied in the monitoring studies using EC (Chapter 3, Chapter 4, Chapter 5) and the experimental study using flux chambers (Chapter 6). Across these chapters, the proportion of the final datasets that were gap-filled ranged from 10 % (Chapter 3) to 37 % (Chapter 5). The use of marginal distribution sampling to gap-fill short periods of data – up to 2 weeks – as utilised in Chapters 3-5, is established with standardised methods (Reichstein et al., 2005; 2016). There is considerable difficulty associated with gap-filling longer periods of missing data, however, such as the missing 128-day period during F2 in Chapter 4, which is why this period was not gap-filled. When small periods of data are missing, fluxes can be inferred based on data recorded on days prior to and after the gap using marginal distribution sampling, as the climate conditions, mainly air temperature and PAR, will be similar. Over longer periods of time, the climate conditions can change considerably and so it is not feasible to fill in long gaps using data recorded several months ago, for example. As discussed in Section 4.3.3, some publications have attempted to fill longer periods of missing data using various methods, however this has the potential to result in potentially unrealistic values which cannot be reported with confidence as there is considerable potential for the fluxes to be under- or over-estimated.

In addition to gap-filling, the minimum detectable flux (MDF) approach for quality controlling automated chamber data should be discussed. This method aims to remove data that cannot be reported with confidence (Nickerson, 2016), and is calculated using a range of data related to the instrument, including analytical accuracy of the instrument for the gas of interest, chamber closure time, chamber volume, and atmospheric pressure (Equation 7.1). It is suggested that values below the MDF are removed from the dataset and then gap-filled. When applying this approach to the GHG flux data in Chapter 6, the MDF of N_2O was calculated to be 0.175 nmol m^{-2} hr^{-1} without collar extensions and 0.183 nmol m^{-2} $2hr^{-1}$ with collar extensions, and for CH_4 was 0.07 nmol m^{-2} $2hr^{-1}$ without collar extensions and 0.073 $nmol^{-2}$ hr^{-1} with collar extensions. Removing values from the dataset that were below these MDFs would have removed a significant proportion of the dataset: 57 % of N_2O values and 88 % of CH_4 values. In addition, any negative fluxes of N_2O and CH_4 were removed, although negative fluxes of these GHGs are entirely possible (Biernat et al., 2020; Liu et al., 2022). Table 7.4 shows average daily and average cumulative values of N_2O and CH_4 per fertiliser treatment both with and without values below the MDFs removed and gap-filled. It was decided that the GHG flux dataset in Chapter 6 would include values below the MDFs for completion of the dataset.

$$MDF\ (nmol\ GHG\ m^{-2}\ hour^{-1}) = \left(\frac{A_A}{t_c}\right)\left(\frac{V\ P}{S\ R\ T}\right) \qquad \text{(Equation 7.1)}$$

where A_A is the analytical accuracy of the instrument (ppm), t_c is the closure time (hours), V is the chamber volume (m^3), P is atmospheric pressure (Pa), S is the chamber surface area (m^{-2}), R is the ideal gas constant (m^{-3} Pa K^{-1} mol^{-1}) and T is ambient temperature (K) (Nickerson, 2016).

TABLE 7.4 MEAN DAILY AND MEAN CUMULATIVE N_2O AND CH_4 FLUXES OVER THE 83-DAY MEASUREMENT PERIOD ± STANDARD DEVIATION (SD) FOR EACH FERTILISER TREATMENT (IF = INORGANIC FERTILISER, UPS = UNTREATED PIG SLURRY, TPS = TREATED PIG SLURRY). 'WITHOUT MDF' REFERS TO THE DATA UTILISED IN CHAPTER 6, WHERE VALUES BELOW THE MDF WERE INCLUDED IN THE DATASET. 'WITH MDF' REFERS TO THE DATA NOT UTILISED IN CHAPTER 6, WHERE VALUES BELOW THE MDF WERE EXCLUDED AND GAP-FILLED.

	Without MDF			**With MDF**		
	IF	**UPS**	**TPS**	**IF**	**UPS**	**TPS**

Mean daily N_2O flux ± SD (g N m^{-2} day^{-1})	0.002 ± 0	0.004 ± 0	0.013 ± 0	0.002 ± 0	0.004 ± 0	0.014 ± 0
Mean cumulative N_2O flux ± SD (g N m^{-2} day^{-1})	0.13 ± 0	0.32 ± 0.1	1.14 ± 0.1	0.15 ± 0	0.34 ± 0.1	1.18 ± 0.1
Mean daily CH_4 flux ± SD (g C m^{-2} day^{-1})	-0.0003 ± 5.8e-05	0.0004 ± 0.0006	-0.0003 ± 0.0001	0.001 ± 0	0.004 ± 0.002	0.001 ± 0.0003
Mean cumulative CH_4 flux ± SD (g C m^{-2} day^{-1})	-1.4 ± 0.3	3.2 ± 1.4	-1.4 ± 0.6	0.08 ± 0.02	0.32 ± 0.2	0.1 ± 0.02

Excluding the measurements taken in CF over 2.5-years (Chapter 4), most of the monitoring and experimental studies in this thesis were conducted over a short time frame, ranging from three months (Chapter 6) to twelve months (Chapter 5), due to limitations with sampling equipment. As a result, the GHG budget of winter wheat for each fertiliser treatment could not be calculated as chamber measurements were not taken over the entire crop growing season (Chapter 6), and the inter-annual variability of NEP was not accounted for when comparing the cropland and permanent pasture as only one year of data was measured (Chapter 5). Assessing the internanual variability of NEE and NEP is important for capturing responses to climate variations. Using only one year of flux data to assess the C sink or source capacity of an agroecosystem can therefore over- or under-estimate the amount of C a system is assimilating. The annual NEP of the cropland and permanent pasture measured in this thesis (Chapter 5) cannot be considered representative of either site, as extremely dry conditions were experienced in 2022 and 2023. It is likely that fluxes would be different in 'normal' years and so longer-term monitoring would provide a more reliable insight to the annual fluxes at the sites, particularly over entire crop rotations (i.e., 3-5 years).

The experiment measuring GHG fluxes using automated chambers (Chapter 6) was partly limited by the experimental design. Many of these limitations were due to financial and operational constraints, which could be addressed in future studies. Although automated chambers are capable of sampling GHG fluxes at a much higher temporal resolution than manual chambers (Gorres et al., 2016), they are expensive (*c.* £16,000 per chamber), which meant measurements in the study were limited to three replicates per treatment. Increasing the number of replicates per treatment in future studies, where possible, would increase the reliability of the results. Furthermore, there was no scope for a control treatment to be included in the experiment (i.e., winter wheat with no fertiliser applied). Including a control treatment would allow the effects of the fertiliser treatments to be more clearly identified and would enable the calculation of emission factors. In the absence of these measurements, a complete replication of the experiment would strengthen the results but was not feasible within the PhD timeframe. The amount of total available N was slightly higher for the plasma-treated pig slurry treatment (253 kg available N ha^{-1}) than the other two treatments (220 kg available N ha^{-1} each); whilst this is unlikely to be the only reason for the higher N_2O emissions from the winter wheat fertilised with plasma-treated pig slurry, it will be a contributing factor. Applying the same amount of available N to each treatment would increase the confidence in the results and make GHG emissions more comparable between treatments, although it is acknowledged that this may be difficult due to the variable N content of pig slurry and thus the UPS and TPS treatments. In addition, a complete replication of the experiment would provide a supplementary benefit of allowing interannual climate variability to be accounted for.

7.4 Directions for further research

The research conducted as part of this project highlights the urgent need for more measurements of NEP and GHG fluxes associated with the use of different agricultural management practices from sites with different combinations of climate conditions and soil types. The monitoring and experimental studies in this project have primarily concentrated on measurements of C loss and GHG emissions associated with business-as-usual practices. It will be necessary to continue these measurements of current farming systems to establish a baseline for UK agricultural emissions. This can be used as a reference point, to assess the effectiveness of best management practices and agricultural interventions at reducing emissions. These robust measurements will be critical for generating evidence-based policies and making site-specific recommendations to farmers, and are particularly vital in countries where data is lacking, including the UK. To provide policymakers with long-term estimates of NEP and GHG fluxes associated with multiple agricultural management practices, it will be imperative that the measurements taken as part of this project are continued and expanded at a greater number of sites across the UK; this could be achieved by establishing a network of long-term nationwide EC flux towers.

To enhance the understanding of the impacts of agricultural management practices on NEP and GHG fluxes, and further advance the quality of research outputs, the following recommendations are made for future research:

- Measure C_I and C_H alongside NEE so NEP can be calculated; measuring NEE only will overestimate the amount of C accumulated by an agroecosystem. This would benefit from the development of a standardised method to calculate C_H via grazing livestock and the proportion of biomass returned as excreta.
- Measure NEP together with N_2O and CH_4 fluxes where possible to ensure all trade-offs are considered.
- Design before-after control-impact (BACI) type studies to directly assess the impacts of management practices on NEP and GHG fluxes.
- Measure the impact of increased additions of C on NEP and if this varies depending on the form of C_I.
- Measure the impact of growing cover crops during fallow periods on NEP.
- Conduct further trials on emerging products and technologies (i.e., plasma-treated pig slurry) to measure the impact of application rate and timing on NEP/GHG fluxes and explore whether emissions can be reduced by applying additional products such as nitrification inhibitors. Measuring the impact of these products on SOC on a long-term basis would also allow insights as to whether any non-CO_2 emissions can be mitigated by increased SOC
- When comparing NEP/GHG fluxes between crops, measure data during the fallow period following harvest in addition to the crop growing season to ensure that all fluxes for a field are accounted for (as TER is often greater than GPP during fallow periods).

- When comparing NEP/GHG fluxes between sites, measure data on an annual basis.
- When comparing NEP/GHG fluxes between croplands or between croplands and grasslands, measure data over the entire crop rotation.
- When measuring the impact of a treatment on NEP/GHG fluxes, include an unfertilised control to clearly identify the effects of fertiliser treatments and to enable the calculation of emission factors.
- Report sufficient meta-data including; average daily air temperature and total precipitation during the measurement period, soil texture, SOC content and stock, grassland management (i.e., cutting, grazing), crop/vegetation type, biomass yield, N fertiliser rate, amount and type of C_I, amount and type of C_H, tillage method, tillage depth, tillage frequency, grazing species, grazing duration, grazing intensity (i.e., as livestock units), cover crops grown in fallow periods, number of harvests, and any management during fallow periods.
- Incorporate results into life-cycle analyses where possible to account for all upstream and downstream emissions associated with the use of a management practice.

7.5 Conclusion

The majority of croplands and managed grasslands investigated in this thesis were behaving as net C sources, with considerably fewer sites acting as net C sinks. On a global scale, croplands had a greater C loss than managed grasslands. Over its growing season, the NEP of maize grown in the UK was heavily influenced by soil type, with C losses over double when grown on peat compared to mineral soil. On an annual basis, the NEP of a cut and grazed pasture in the UK showed C loss, whereas a neighbouring cropland was C neutral. Of the crops grown in the cropland, winter wheat and vining pea behaved as C sinks during their growing seasons, but maize behaved as a C source. Over the entire 2.5-year measurement period (i.e., considering crop growing seasons and fallow periods), the cropland was a C source. The fertilisation of a winter wheat crop with plasma-treated pig slurry increased non-CO_2 GHG emissions relative to winter wheat fertilised with untreated pig slurry and inorganic fertiliser.

The results show that, despite the implementation of best management practices globally, agricultural soils are still losing C and emitting GHGs to the atmosphere. If net zero and C sequestration targets are to be met, GHG emissions must be reduced and C sequestration must be increased in agricultural soils. Farmers must be provided with evidence-based guidance on which management practices are best suited to achieve this, based on the local climate and soil conditions. This cannot be achieved without sufficient measurements of the impacts of different agricultural management practices on NEP and GHG fluxes, in the UK and globally. A considerable increase in the number of these measurements is recommended, with particular focus on evaluating the impacts of increased C_I, the growth of cover crops in fallow periods and the use of novel fertilisers. The studies conducted to record these measurements should be designed as BACI type studies, should report both NEP and GHG fluxes together, and should incorporate results into life-cycle analyses. This data will allow policymakers to make more targeted recommendations for how they can improve their practices and develop the existing agricultural policies in the UK to ensure a reduction in GHG emissions and an increase in soil C storage for climate change mitigation.

7.6 References

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Appendices

A1 Supporting information for Chapter 2

TABLE A1.1 OVERVIEW OF THE DATA CLASSIFIED AS ARABLE CROPLAND. AW = WET TROPICAL SAVANNA CLIMATE, BSK = COLD SEMI-ARID CLIMATE, BWK = COLD DESERT CLIMATE, CFA = HUMID SUBTROPICAL CLIMATE, CFB = TEMPERATE OCEANIC CLIMATE, CSB = WARM-SUMMER MEDITERRANEAN CLIMATE, CWA = MONSOON-INFLUENCED HUMID SUBTROPICAL CLIMATE, DFA = HOT-SUMMER HUMID CONTINENTAL CLIMATE, DFB = WARM-SUMMER HUMID CONTINENTAL CLIMATE, DWA = MONSOON-INFLUENCED HOT-SUMMER HUMID CONTINENTAL CLIMATE. ANNUAL NEE AND ANNUAL NEP FOLLOW THE MICROMETEOROLOGICAL SIGN CONVENTION (AS IN EVANS ET AL., 2021), WHERE A NEGATIVE VALUE INDICATES THE AGROECOSYSTEM IS ACCUMULATING C AND A POSITIVE VALUE INDICATES C LOSS FROM THE AGROECOSYSTEM.

Crop type	Crop species	Country	Köppen climate classification	Soil type	Amount of N fertiliser added (kg ha^{-1} yr^{-1})	Tillage	Residues retained	Cover crops	Carbon import (g C m^{-2})	Carbon export (g C m^{-2})	Annual NEE (g C m^{-2})	Annual NEP (g C m^{-2})	Reference
Annual	Cover crop, silage maize, winter wheat	Germany	Cfb	Light	201-300	Conventional	Yes	Yes	0	797.4	-32.6	764.8	Poyda et al. (2019)
Annual	Cover crop, silage maize	Germany	Cfb	Medium	201-300	Conventional	Yes	Yes	40	891.5	-119.1	732.4	Poyda et al. (2019)
Annual	Maize	Canada	Dfb	Light	101-200	Unknown	Unknown	Unknown	0	635	64	699	Eichelmann et al. (2016)
Annual	Cover crop, silage maize, winter wheat	Germany	Cfb	Medium	101-200	Conventional	Yes	Yes	61.6	778.3	-105.6	611.1	Poyda et al. (2019)
Annual	Rice	Korea	Dwa	Medium	101-200	Conventional	Yes	No	0	587.9	4.5	592.4	Hwang et al. (2020)
Annual	Cover crop, silage maize, winter wheat	Germany	Cfb	Medium	201-300	Conventional	Yes	Yes	100	827.5	-166.6	560.9	Poyda et al. (2019)
Perennial	Alfalfa	USA	Dfb	Light	1-100	No till	Yes	Unknown	0	853	-326	527	Saliendra et al. (2018)
Annual	Rice	Korea	Dwa	Medium	101-200	Conventional	Yes	No	0	590.2	-74.9	515.3	Hwang et al. (2020)
Annual	Rice	Korea	Dwa	Medium	101-200	Conventional	Yes	No	0	574.8	-70.5	504.3	Hwang et al. (2020)
Annual	Maize	China	BSk	Light	101-200	Conventional	Yes	Unknown	0	637.8	-135.3	502.5	Niu et al. (2021)
Annual	Maize	China	BSk	Light	101-200	No till	Yes	Unknown	0	474.8	12.7	487.5	Niu et al. (2021)
Annual	Winter rapeseed, winter wheat	Germany	Cfb	Light	101-200	Conventional	Yes	No	0	244.4	239.8	484.2	Poyda et al. (2019)
Annual	Cover crop, silage maize, winter wheat	Germany	Cfb	Light	201-300	Conventional	Yes	Yes	0	652.1	-204.8	447.3	Poyda et al. (2019)
Annual	Winter wheat, summer maize	China	Cwa	Light	>401	Unknown	Yes	No	0	649	-205	444	Yue et al. (2023)
Annual	Maize	China	BSk	Light	101-200	Conventional	Yes	Unknown	0	613.2	-204.6	408.6	Niu et al. (2021)

Annual	Winter wheat, summer maize	China	Cwa	Light	>401	Unknown	Yes	No	0	581	-173	408	Yue et al. (2023)
Annual	Potato	Germany	Cfb	Medium	Unknown	Reduced tillage	Yes	Unknown	0	420	-34	386	Anthoni et al. (2004)
Annual	Winter wheat	Germany	Cfb	Light	101-200	Conventional	Yes	No	0	233.5	148.7	382.2	Poyda et al. (2019)
Annual	Winter wheat, summer maize	China	Cwa	Light	>401	Unknown	Yes	No	0	659	-280	379	Yue et al. (2023)
Perennial	Alfalfa	USA	Dfb	Light	1-100	No till	Yes	Unknown	0	698	-340	358	Saliendra et al. (2018)
Annual	Cover crop, silage maize, winter wheat	Germany	Cfb	Light	201-300	Conventional	Yes	Yes	0	689.2	-332.3	356.9	Poyda et al. (2019)
Annual	Winter wheat, summer maize	China	Cwa	Light	>401	Conventional	No	Unknown	0	538.2	-197.6	340.6	Li et al. (2006)
Annual	Cover crop, silage maize, winter wheat	Germany	Cfb	Light	201-300	Conventional	Yes	Yes	0	723.2	-394	329.2	Poyda et al. (2019)
Annual	Maize	France	Cfb	Heavy	201-300	Conventional	Yes	No	249	806	-240	317	Beziat et al. (2009)
Annual	Maize	USA	Dfa	Light	0	No till	Yes	No	0	335	-29	306	Abraha et al. (2018)
Annual	Cover crop, silage maize, cover crop, silage maize	Germany	Cfb	Medium	101-200	Conventional	Yes	Yes	0	700.7	-398.1	302.6	Poyda et al. (2019)
Annual	Maize	USA	Dfa	Light	101-200	No till	No	No	0	161	135	296	Zenone et al. (2013)
Annual	Cover crop, summer barley, cover crop, silage maize	Germany	Cfb	Medium	101-200	Conventional	Yes	Yes	28	212.1	108.1	292.2	Poyda et al. (2019)
Annual	Maize	USA	Dfa	Light	0	No till	Yes	No	0	216	66	282	Abraha et al. (2018)
Annual	Soybean	USA	Dfa	Medium	0	Unknown	No	Unknown	0	125.1	155.7	280.8	Zenone et al. (2011)
Annual	Silage maize, winter barley	Germany	Cfb	Medium	201-300	Conventional	Yes	No	120	738.1	-339.5	278.6	Poyda et al. (2019)
Annual	Maize	USA	Dfa	Light	101-200	No till	Yes	No	0	352	-77	275	Abraha et al. (2018)
Annual	Maize	China	BSk	Light	101-200	Conventional	Yes	Unknown	0	438.8	-170.3	268.5	Niu et al. (2021)
Annual	Winter rapeseed, winter wheat	Germany	Cfb	Light	201-300	Conventional	Yes	No	0	204.2	52.3	256.5	Poyda et al. (2019)
Annual	Soybean	USA	Dfa	Medium	0	No till	No	Unknown	0	116.1	139.6	255.7	Zenone et al. (2011)
Annual	Winter wheat	Germany	Cfb	Light	101-200	Conventional	Yes	No	0	524	-270	254	Schmidt et al. (2012)
Annual	Maize	USA	Dfa	Light	101-200	No till	Yes	No	0	347	-96	251	Abraha et al. (2018)
Annual	Winter wheat, winter barley	Germany	Cfb	Medium	101-200	Conventional	Yes	No	0	650.2	-399.6	250.6	Poyda et al. (2019)
Annual	Soybean	USA	Dfa	Medium	0	No till	No	Unknown	0	122	128.1	250.1	Zenone et al. (2011)
Annual	Cover crop, silage maize, winter wheat	Germany	Cfb	Medium	>401	Conventional	Yes	Yes	84	606.3	-272.3	250	Poyda et al. (2019)
Annual	Maize	USA	Dfa	Medium	0	No till	Yes	Unknown	0	183	48	231	Grant et al. (2007)
Annual	Soybean	USA	Dfa	Medium	0	No till	Yes	No	0	183	48	231	Verma et al. (2005)
Annual	Maize	USA	Dfa	Light	101-200	No till	No	Yes	0	303	-75	228	Zenone et al. (2013)
Annual	Soybean	USA	Dfa	Heavy	0	No till	Yes	Unknown	0	135.7	90.8	226.5	Zenone et al. (2011)
Annual	Soybean	USA	Dfa	Light	Unknown	No till	Unknown	Unknown	0	120.6	103.7	224.3	Hollinger et al. (2005)
Annual	Silage maize, winter wheat	Germany	Cfb	Light	>401	Conventional	Yes	No	75.6	571.6	-283.5	212.5	Poyda et al. (2019)
Annual	Winter wheat	Germany	Cfb	Light	101-200	Conventional	Yes	No	0	479	-270	209	Schmidt et al. (2012)
Annual	Maize	USA	Dfa	Light	101-200	No till	No	Yes	0	462	-261	201	Zenone et al. (2013)
Annual	Winter wheat, summer maize	China	Cwa	Light	>401	Unknown	Yes	No	0	578	-395	183	Yue et al. (2023)
Annual	Winter wheat, summer maize	China	Cwa	Light	0	Unknown	Yes	No	0	441	-260	181	Yue et al. (2023)

Annual	Barley	USA	Csb	Light	Unknown	Conventional	Yes	No	0	267	-87	180	Chi et al. (2017)
Annual	Maize	USA	Dfa	Medium	0	No till	Yes	Unknown	0	153	18	171	Grant et al. (2007)
Annual	Soybean	USA	Dfa	Medium	0	No till	Yes	No	0	153	18	171	Verma et al. (2005)
Annual	Cotton	China	Cwa	Medium	1-100	Conventional	Yes	No	3.2	195.9	-36.3	156.4	Liu et al. (2019)
Annual	Soybean	USA	Dfa	Light	Unknown	No till	Unknown	Unknown	0	129.8	9.2	139	Hollinger et al. (2005)
Annual	Maize	USA	Dfa	Light	101-200	No till	Yes	No	0	361	-222	139	Abraha et al. (2018)
Perennial	Alfalfa	USA	Dfb	Light	1-100	No till	Yes	Unknown	0	451	-315	136	Saliendra et al. (2018)
Annual	Maize	USA	Cfa	Light	201-300	No till	Yes	Unknown	0	683	-547	136	Maleski et al. (2019)
Annual	Cover crop, summer barley, winter rapeseed	Germany	Cfb	Medium	101-200	Conventional	Yes	Yes	0	320.8	-185.6	135.2	Poyda et al. (2019)
Annual	Cover crop, soybean,maize, CC	Brazil	Aw	Heavy	Unknown	No till	Yes	Yes	0	449	-321	128	Dalmagro et al. (2002)
Annual	Cover crop, grain maize, winter wheat	Germany	Cfb	Light	101-200	Conventional	Yes	Yes	0	330.9	-208.7	122.2	Poyda et al. (2019)
Annual	Winter wheat, cover crop, silage maize	Germany	Cfb	Light	201-300	Conventional	Yes	Yes	0	347	-230.3	116.7	Poyda et al. (2019)
Annual	Soybean	USA	Dfa	Medium	0	No till	Yes	Unknown	0	81.9	31	112.9	Zenone et al. (2011)
Annual	Winter wheat, summer maize	China	Cwa	Light	>401	Conventional	No	Unknown	0	425.4	-317.9	107.5	Li et al. (2006)
Annual	Winter wheat, summer maize	China	Cwa	Light	>401	Unknown	Yes	No	0	663	-559	104	Yue et al. (2023)
Annual	Winter wheat	Germany	Cfb	Medium	Unknown	Reduced tillage	Yes	Unknown	0	290	-193	97	Anthoni et al. (2004)
Annual	Cotton	China	BWk	Light	301-400	Conventional	Yes	Unknown	0	179	-84	95	Ming et al. (2021)
Annual	Winter wheat, summer maize	China	Cwa	Light	>401	Unknown	Yes	No	0	435	-340	95	Yue et al. (2023)
Annual	Cotton	China	BWk	Light	301-400	Conventional	Yes	Unknown	0	168	-77	91	Ming et al. (2021)
Annual	Sunflower	France	Cfb	Medium	0	Conventional	Yes	No	0	97.6	-8.5	89.1	Beziat et al. (2009)
Annual	Maize	USA	Dfa	Medium	201-300	No till	Yes	No	0	470	-381	89	Verma et al. (2005)
Annual	Winter wheat, summer maize	China	Cwa	Light	>401	Unknown	Yes	No	0	527	-444	83	Yue et al. (2023)
Annual	Maize	USA	Dfa	Medium	201-300	No till	Yes	No	0	503	-424	79	Verma et al. (2005)
Annual	Winter wheat, summer maize	China	Cwa	Light	201-300	Unknown	Yes	No	0	577	-499	78	Yue et al. (2023)
Annual	Winter wheat, cover crop, summer barley	Germany	Cfb	Medium	1-100	Conventional	Yes	Yes	0	322.8	-247.7	75.1	Poyda et al. (2019)
Annual	Winter rapeseed, winter wheat	Germany	Cfb	Medium	>401	Conventional	Yes	No	0	152.4	-78.4	74	Poyda et al. (2019)
Annual	Soybean	USA	Dfa	Medium	0	No till	Yes	Unknown	0	127.3	-56	71.3	Zenone et al. (2011)
Annual	Cotton	China	BWk	Light	301-400	Conventional	Yes	Unknown	0	180	-110	70	Ming et al. (2021)
Annual	Cotton	China	BWk	Light	301-400	Conventional	Yes	Unknown	0	157	-108	49	Ming et al. (2021)
Annual	Maize	USA	Dfa	Light	101-200	No till	Yes	No	0	230	-181	49	Abraha et al. (2018)
Annual	Winter rapeseed, winter wheat	Germany	Cfb	Light	101-200	Conventional	Yes	No	0	171.6	-124.4	47.2	Poyda et al. (2019)
Annual	Winter wheat, summer maize	China	Cwa	Light	>401	Unknown	Yes	No	0	645	-602	43	Yue et al. (2023)
Annual	Garbanzo	USA	Csb	Light	Unknown	No till	Yes	No	0	60	-20	40	Waldo et al. (2016)
Annual	Winter wheat, summer maize	China	Cwa	Light	>401	Unknown	Yes	No	0	488	-450	38	Yue et al. (2023)
Annual	Cotton	China	BWk	Light	301-400	Conventional	Yes	Unknown	0	155	-125	30	Ming et al. (2021)

Annual	Maize	USA	Dfa	Light	101-200	No till	No	No	0	307	-280	27	Zenone et al. (2013)
Annual	Winter wheat	Germany	Cfb	Medium	201-300	Conventional	Yes	No	0	316.3	-291	25.3	Poyda et al. (2019)
Annual	Pea	USA	Csb	Light	Unknown	Conventional	Yes	No	0	63	-39	24	Chi et al. (2017)
Annual	Maize	USA	Dfa	Light	0	No till	Yes	No	0	102	-80	22	Abraha et al. (2018)
Annual	Winter wheat, cover crop, summer barley	Germany	Cfb	Medium	1-100	Conventional	Yes	Yes	0	289.8	-269.7	20.1	Poyda et al. (2019)
Annual	Soybean, rice, french bean	Brazil	Aw	Heavy	Unknown	No till	Yes	No	0	334	-316	18	Dalmagro et al. (2002)
Annual	Winter barley, spelt	Germany	Cfb	Medium	101-200	Conventional	Yes	No	72	404.6	-322.6	10	Poyda et al. (2019)
Annual	Winter wheat, summer maize	China	Cwa	Light	>401	Unknown	Yes	No	0	590	-581	9	Yue et al. (2023)
Annual	Winter rapeseed, winter wheat	Germany	Cfb	Light	101-200	Conventional	Yes	No	0	221.8	-213.2	8.6	Poyda et al. (2019)
Annual	Maize	USA	Dfa	Light	0	No till	Yes	No	0	302	-296	6	Abraha et al. (2018)
Annual	Maize	USA	Dfa	Medium	101-200	No till	Yes	No	0	521	-517	4	Verma et al. (2005)
Annual	Maize	USA	Dfa	Light	101-200	No till	Yes	No	0	266	-265	1	Abraha et al. (2018)
Annual	Winter wheat, cover crop, silage maize	Germany	Cfb	Light	>401	Conventional	Yes	Yes	0	276.8	-284	-7.2	Poyda et al. (2019)
Annual	Maize	USA	Dfa	Medium	101-200	No till	Yes	No	0	518	-529	-11	Verma et al. (2005)
Annual	Maize	China	Cwa	Light	201-300	Conventional	Yes	Unknown	0	470	-484	-14	Gao et al. (2017)
Annual	Maize	China	Cwa	Light	201-300	Conventional	Yes	Unknown	0	476	-491	-15	Gao et al. (2017)
Annual	Winter rapeseed, winter wheat	Germany	Cfb	Medium	101-200	Conventional	Yes	No	0	62.4	-77.6	-15.2	Poyda et al. (2019)
Annual	Winter barley, cover crop, silage maize	Germany	Cfb	Medium	201-300	Conventional	Yes	Yes	160	257.1	-114	-16.9	Poyda et al. (2019)
Annual	Winter wheat, summer maize	China	Cwa	Light	>401	Unknown	Yes	No	0	528	-545	-17	Yue et al. (2023)
Annual	Winter wheat, cover crop, silage maize	Germany	Cfb	Light	201-300	Conventional	Yes	Yes	112	462.2	-373	-22.8	Poyda et al. (2019)
Annual	Winter wheat	France	Cfb	Medium	101-200	Conventional	Yes	No	0	277	-305	-28	Beziat et al. (2009)
Annual	Soybean	USA	Dfa	Medium	101-200	No till	Yes	Unknown	0	538	-572	-34	Grant et al. (2007)
Annual	Maize	USA	Dfa	Medium	101-200	No till	Yes	No	0	538	-572	-34	Verma et al. (2005)
Annual	Winter wheat, summer maize	China	Cwa	Light	>401	Unknown	Yes	No	0	535	-582	-47	Yue et al. (2023)
Annual	Wheat, soybean	Brazil	Cfa	Heavy	201-300	No till	Yes	No	0	383	-433	-50	Veeck et al. (2022)
Annual	Maize	China	Cwa	Light	201-300	Conventional	Yes	Unknown	0	502	-553	-51	Gao et al. (2017)
Annual	Soybean	USA	Dfa	Light	Unknown	No till	Unknown	Unknown	0	127.2	-210.4	-83.2	Hollinger et al. (2005)
Annual	Winter wheat	France	Cfb	Heavy	301-400	Conventional	Yes	No	80.9	383	-387	-84.9	Beziat et al. (2009)
Annual	Canola	USA	Csb	Light	Unknown	Conventional	Yes	No	0	77	-162	-85	Chi et al. (2017)
Perennial	Alfalfa	USA	Dfb	Light	1-100	No till	No	Unknown	0	314	-404	-90	Saliendra et al. (2018)
Annual	Winter wheat, winter rapeseed	Germany	Cfb	Light	101-200	Conventional	Yes	No	72	378	-396.3	-90.3	Poyda et al. (2019)
Annual	Maize	USA	Dfa	Light	101-200	No till	Yes	No	0	321	-412	-91	Abraha et al. (2018)
Annual	Soybean	USA	Dfa	Medium	1-100	No till	Yes	Unknown	0	297	-397	-100	Grant et al. (2007)
Annual	Maize	USA	Dfa	Medium	1-100	No till	Yes	No	0	297	-397	-100	Verma et al. (2005)
Annual	Winter wheat, winter rapeseed	Germany	Cfb	Light	201-300	Conventional	Yes	No	0	302.4	-432.4	-130	Poyda et al. (2019)

Annual	Winter wheat, cover crop, silage maize	Germany	Cfb	Light	201-300	Conventional	Yes	Yes	0	295.8	-429.7	-133.9	Poyda et al. (2019)
Annual	Wheat, maize	China	Cwa	Medium	>401	Conventional	Yes	No	18.4	589.3	-708	-137.1	Liu et al. (2019)
Annual	Canola	USA	Csb	Light	Unknown	No till	Yes	No	0	67	-210	-143	Chi et al. (2017)
Annual	Maize	USA	Dfa	Light	Unknown	No till	Unknown	Unknown	0	339.4	-505.1	-165.7	Hollinger et al. (2005)
Annual	Spelt, cover crop, silage maize	Germany	Cfb	Medium	101-200	Conventional	Yes	Yes	120	383.1	-432.3	-169.2	Poyda et al. (2019)
Annual	Winter wheat, cover crop, silage maize	Germany	Cfb	Light	201-300	Conventional	Yes	Yes	0	262.2	-436	-173.8	Poyda et al. (2019)
Annual	Maize	USA	Dfa	Medium	101-200	No till	Yes	No	0	335	-510	-175	Verma et al. (2005)
Annual	Winter wheat, cover crop, grain maize	Germany	Cfb	Light	201-300	Conventional	Yes	Yes	72	301	-421.6	-192.6	Poyda et al. (2019)
Annual	Maize	USA	Dfa	Light	Unknown	No till	Unknown	Unknown	0	339.5	-532.2	-192.7	Hollinger et al. (2005)
Annual	Winter wheat, cover crop, silage maize	Germany	Cfb	Medium	>401	Conventional	Yes	Yes	28	308.9	-490.6	-209.7	Poyda et al. (2019)
Annual	Winter wheat, winter barley	Germany	Cfb	Medium	201-300	Conventional	Yes	No	42	345.4	-515.6	-212.2	Poyda et al. (2019)
Annual	Winter wheat	USA	Csb	Light	Unknown	No till	Yes	No	0	232	-450	-218	Chi et al. (2017)
Annual	Winter wheat	USA	Csb	Light	Unknown	No till	Yes	No	0	287	-517	-230	Waldo et al. (2016)
Perennial	Blueberry	Canada	Cfb	Light	101-200	No till	Unknown	Unknown	488	84	171	-233	Pow et al. (2020)
Annual	Winter wheat	USA	BSk	Light	Unknown	No till	Yes	No	0	67	-310	-243	Chi et al. (2017)
Annual	Winter wheat	USA	Csb	Light	Unknown	Conventional	Yes	No	0	269	-521	-252	Chi et al. (2017)
Annual	Maize	USA	Dfa	Light	Unknown	No till	Unknown	Unknown	0	436.9	-691.8	-254.9	Hollinger et al. (2005)
Annual	Winter wheat, winter rapeseed	Germany	Cfb	Light	201-300	Conventional	Yes	No	0	325.4	-654.3	-328.9	Poyda et al. (2019)
Annual	Winter wheat	USA	Csb	Light	Unknown	Conventional	Yes	No	0	176	-569	-393	Chi et al. (2017)
Annual	Winter barley, cover crop, silage maize	Germany	Cfb	Medium	201-300	Conventional	Yes	Yes	112	343.8	-629	-397.2	Poyda et al. (2019)
Annual	Winter wheat	USA	BSk	Light	Unknown	Conventional	Yes	No	0	79	-524	-445	Waldo et al. (2016)

TABLE A1.2 OVERVIEW OF THE DATA CLASSIFIED AS MANAGED GRASSLAND. CFA = HUMID SUBTROPICAL CLIMATE, CFB = TEMPERATE OCEANIC CLIMATE, DFB = WARM-SUMMER HUMID CONTINENTAL CLIMATE, DFC = SUBARCTIC CLIMATE. ANNUAL NEE AND ANNUAL NEP FOLLOW THE MICROMETEOROLOGICAL SIGN CONVENTION (AS IN EVANS ET AL., 2021), WHERE A NEGATIVE VALUE INDICATES THE AGROECOSYSTEM IS ACCUMULATING C AND A POSITIVE VALUE INDICATES C LOSS FROM THE AGROECOSYSTEM.

Management	**Country**	**Köppen climate classification**	**Soil type**	**Amount of N fertiliser added (kg ha^{-1} yr^{-1})**	**C import (g C m^{-2})**	**C export (g C m^{-2})**	**Annual NEE (g C m^{-2})**	**Annual NEP (g C m^{-2})**	**Reference**

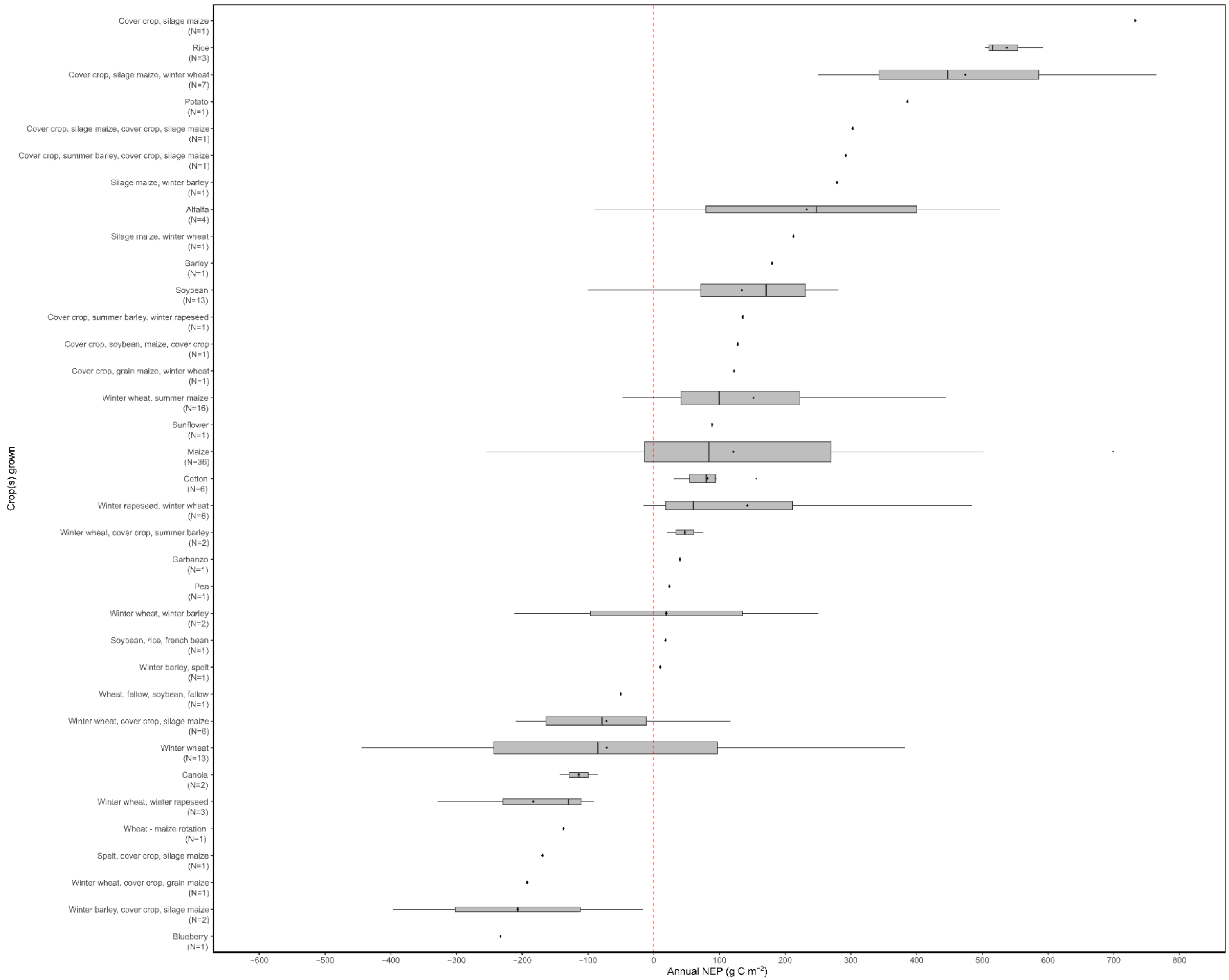
Cut	Japan	Cfa	Medium	201-300	0	553	-207	346	Hirata et al. (2013)
Cut + grazed	USA	Cfa	Light	101-200	0	171	166	337	Skinner (2013)
Grazed	New Zealand	Cfb	Light	1-100	142	417	14	289	Wall et al. (2023b)
Cut	Japan	Cfa	Medium	101-200	0	392	-111	281	Hirata et al. (2013)
Cut + grazed	New Zealand	Cfb	Light	0	100	564	-187	277	Laubach et al. (2019)
Cut	USA	Dfb	Light	0	0	311	-35	276	Saliendra et al. (2018)
Cut + grazed	USA	Cfa	Light	201-300	0	148	126	274	Skinner (2013)
Grazed	New Zealand	Cfb	Light	101-200	246	527	-25	256	Wall et al. (2023b)
Cut	Japan	Cfa	Medium	201-300	0	291	-47	244	Hirata et al. (2013)
Cut + grazed	New Zealand	Cfb	Light	1-100	157.8	638	-260	220.2	Laubach et al. (2023)
Cut	Japan	Cfa	Medium	101-200	0	413	-205	208	Hirata et al. (2013)
Cut	Japan	Dfc	Light	1-100	0	410	-206	204	Hirata et al. (2013)
Cut	Japan	Dfc	Light	1-100	0	370	-166	204	Hirata et al. (2013)
Grazed	New Zealand	Cfb	Light	1-100	195	460	-66	199	Rutledge et al. (2017)
Cut	Japan	Dfb	Light	101-200	0	540	-354	186	Hirata et al. (2013)
Cut	Japan	Dfb	Light	101-200	0	510	-340	170	Hirata et al. (2013)
Cut + grazed	USA	Cfa	Light	101-200	0	145	23	168	Skinner (2013)
Cut + grazed	USA	Cfa	Light	1-100	32	162	29	159	Skinner (2008)
Cut	USA	Dfb	Light	0	0	106	50	156	Saliendra et al. (2018)
Cut + grazed	USA	Cfa	Light	1-100	0	76	77	153	Skinner (2013)
Grazed	New Zealand	Cfb	Light	1-100	205	465	-108	152	Wall et al. (2020a)
Grazed	New Zealand	Cfb	Light	1-100	323	537	-63	151	Rutledge et al. (2017)
Grazed	New Zealand	Cfb	Light	1-100	325	640	-167	148	Rutledge et al. (2017)
Cut	Switzerland	Cfb	Heavy	0	0	219	-71	148	Ammann et al. (2007)
Cut	Japan	Cfa	Medium	201-300	0	489	-347	142	Hirata et al. (2013)
Grazed	USA	Cfa	Light	1-100	31	83	81	133	Skinner (2008)
Cut	Japan	Dfc	Light	1-100	0	360	-229	131	Hirata et al. (2013)
Grazed	New Zealand	Cfb	Light	1-100	288	483	-64	131	Rutledge et al. (2017)
Grazed	USA	Cfa	Light	1-100	53	142	41	130	Skinner (2008)
Cut + grazed	USA	Cfa	Light	201-300	0	164	-41	123	Skinner (2013)
Cut + grazed	New Zealand	Cfb	Light	0	23	397	-252	122	Laubach et al. (2019)
Grazed	New Zealand	Cfb	Light	1-100	226	462	-115	121	Rutledge et al. (2017)
Cut	Japan	Dfb	Light	1-100	0	400	-284	116	Hirata et al. (2013)
Cut + grazed	Belgium	Cfb	Light	101-200	213	372	-52	107	de la Motte et al. (2016)
Grazed	New Zealand	Cfb	Light	1-100	214	541	-221	106	Wall et al. (2023b)

Grazed	New Zealand	Cfb	Light	1-100	178	439	-156	105	Wall et al. (2020a)
Cut + grazed	USA	Cfa	Light	1-100	0	92	10	102	Skinner (2013)
Cut + grazed	USA	Cfa	Light	201-300	0	109	-15	94	Skinner (2013)
Cut	Japan	Cfa	Medium	201-300	190	516	-235	91	Hirata et al. (2013)
Cut + grazed	USA	Cfa	Light	201-300	0	128	-40	88	Skinner (2013)
Cut + grazed	USA	Cfa	Light	1-100	0	97	-16	81	Skinner (2013)
Cut	USA	Dfb	Light	0	0	166	-87	79	Saliendra et al. (2018)
Grazed	Belgium	Cfb	Light	101-200	86	323	-159	78	de la Motte et al. (2016)
Grazed	New Zealand	Cfb	Light	1-100	252	563	-233	78	Wall et al. (2020a)
Grazed	New Zealand	Cfb	Light	1-100	191	464	-201	72	Rutledge et al. (2017)
Cut + grazed	New Zealand	Cfb	Light	1-100	157.8	563	-338	67.2	Laubach et al. (2023)
Grazed	Belgium	Cfb	Light	1-100	61	230	-102	67	de la Motte et al. (2016)
Cut + grazed	USA	Cfa	Light	1-100	10	83	-9	64	Skinner (2008)
Grazed	New Zealand	Cfb	Light	1-100	286	548	-204	58	Wall et al. (2020a)
Grazed	New Zealand	Cfb	Light	1-100	256	479	-170	53	Wall et al. (2020a)
Grazed	New Zealand	Cfb	Light	1-100	305	583	-226	52	Rutledge et al. (2017)
Cut + grazed	USA	Cfa	Light	0	13	127	-62	52	Skinner (2008)
Cut + grazed	USA	Cfa	Light	1-100	21	103	-33	49	Skinner (2008)
Cut	USA	Dfb	Light	0	0	116	-79	37	Saliendra et al. (2018)
Grazed	New Zealand	Cfb	Light	1-100	179	546	-330	37	Wall et al. (2020a)
Cut + grazed	New Zealand	Cfb	Light	1-100	191	508	-281	36	Wall et al. (2020a)
Grazed	USA	Cfa	Light	1-100	44	118	-44	30	Skinner (2008)
Grazed	New Zealand	Cfb	Light	1-100	342	643	-272	29	Rutledge et al. (2017)
Cut + grazed	USA	Cfa	Light	0	8	191	-155	28	Skinner (2008)
Cut	Switzerland	Cfb	Heavy	0	0	380	-352	28	Ammann et al. (2007)
Grazed	New Zealand	Cfb	Light	1-100	298	625	-302	25	Wall et al. (2020a)
Cut	Japan	Cfa	Light	101-200	0	473	-450	23	Matsuura et al. (2023)
Cut	Japan	Cfa	Medium	201-300	360	393	-12	21	Hirata et al. (2013)
Grazed	Belgium	Cfb	Light	1-100	72	286	-193	21	de la Motte et al. (2016)
Cut + grazed	New Zealand	Cfb	Light	1-100	196	473	-257	20	Wall et al. (2020a)
Cut	Japan	Cfa	Medium	101-200	0	450	-442	8	Hirata et al. (2013)
Cut + grazed	USA	Cfa	Light	101-200	0	103	-104	-1	Skinner (2013)
Cut	Switzerland	Cfb	Heavy	0	0	335	-339	-4	Ammann et al. (2007)
Cut	USA	Dfb	Medium	0	34	485	-458	-7	Wiesner et al. (2022)
Grazed	New Zealand	Cfb	Light	201-300	346	578	-245	-13	Wall et al. (2019)

Cut	Japan	Cfa	Medium	201-300	187	391	-221	-17	Hirata et al. (2013)
Grazed	New Zealand	Cfb	Light	1-100	198	505	-325	-18	Wall et al. (2020a)
Cut + grazed	New Zealand	Cfb	Light	1-100	125.4	584	-485	-26.4	Laubach et al. (2023)
Cut + grazed	New Zealand	Cfb	Light	201-300	384	518	-164	-30	Wall et al. (2019)
Cut	Switzerland	Cfb	Heavy	201-300	59	241	-215	-33	Ammann et al. (2007)
Cut	Japan	Cfa	Medium	>401	341	450	-161	-52	Hirata et al. (2013)
Grazed	New Zealand	Cfb	Light	1-100	255	473	-274	-56	Wall et al. (2020a)
Cut	Japan	Cfa	Medium	101-200	328	480	-248	-96	Matsuura et al. (2023)
Cut	Japan	Cfa	Medium	301-400	380	430	-155	-105	Hirata et al. (2013)
Grazed	New Zealand	Cfb	Light	101-200	316	616	-408	-108	Wall et al. (2020a)
Grazed	New Zealand	Cfb	Light	1-100	643	611	-78	-110	Rutledge et al. (2017)
Cut + grazed	New Zealand	Cfb	Light	201-300	395	648	-364	-111	Wall et al. (2019)
Cut	Germany	Cfb	Light	0	0	147	-260	-113	Hussain et al. (2011)
Cut + grazed	New Zealand	Cfb	Light	Unknown	13.7	107.5	-227.1	-133.3	Rutledge et al. (2015)
Grazed	New Zealand	Cfb	Light	1-100	595	502	-45	-138	Rutledge et al. (2017)
Cut	Switzerland	Cfb	Heavy	101-200	22	401	-517	-138	Ammann et al. (2007)
Grazed	New Zealand	Cfb	Light	1-100	342	527	-345	-160	Wall et al. (2020a)
Cut + grazed	New Zealand	Cfb	Light	Unknown	19.8	29.6	-183.7	-173.9	Rutledge et al. (2015)
Cut	Japan	Dfc	Light	301-400	472	450	-157	-179	Hirata et al. (2013)
Cut + grazed	New Zealand	Cfb	Light	Unknown	14.6	12.8	-189.5	-191.3	Rutledge et al. (2015)
Grazed	New Zealand	Cfb	Light	1-100	276	520	-436	-192	Wall et al. (2020a)
Cut	Japan	Cfa	Medium	>401	346	334	-186	-198	Hirata et al. (2013)
Cut	Japan	Dfc	Light	201-300	467	320	-94	-241	Hirata et al. (2013)
Grazed	New Zealand	Cfb	Light	1-100	588	521	-178	-245	Rutledge et al. (2017)
Cut + grazed	New Zealand	Cfb	Light	1-100	82.4	433	-600	-249.4	Laubach et al. (2023)
Cut	Switzerland	Cfb	Heavy	201-300	59	462	-669	-266	Ammann et al. (2007)
Cut	Japan	Dfc	Light	201-300	530	333	-83	-280	Hirata et al. (2013)
Cut	USA	Dfb	Medium	0	34	308	-580	-306	Wiesner et al. (2022)
Cut	Japan	Dfb	Light	>401	600	500	-263	-363	Hirata et al. (2013)
Cut	Japan	Dfb	Light	301-400	580	400	-184	-364	Hirata et al. (2013)
Cut	Japan	Dfb	Light	301-400	770	410	-139	-499	Hirata et al. (2013)

Table A1.3 Number of observations (N) per country.

Country	**N=**
Belgium	4
Brazil	3
Canada	2
China	30
France	4
Germany	45
Japan	26
Korea	3
New Zealand	40
Switzerland	6
USA	79



Cover crop, silage maize (N=1)
Rice (N=3)
Cover crop, silage maize, winter wheat (N=7)
Potato (N=1)
Cover crop, silage maize, cover crop, silage maize (N=1)
Cover crop, summer barley, cover crop, silage maize (N=1)
Silage maize, winter barley (N=1)
Alfalfa (N=4)
Silage maize, winter wheat (N=1)
Barley (N=1)
Soybean (N=13)
Cover crop, summer barley, winter rapeseed (N=1)
Cover crop, soybean, maize, cover crop (N=1)
Cover crop, grain maize, winter wheat (N=1)
Winter wheat, summer maize (N=16)
Sunflower (N=1)
Maize (N=36)
Cotton (N=6)
Winter rapeseed, winter wheat (N=6)
Winter wheat, cover crop, summer barley (N=2)
Garbanzo (N=1)
Pea (N=1)
Winter wheat, winter barley (N=2)
Soybean, rice, french bean (N=1)
Winter barley, spelt (N=1)
Wheat, fallow, soybean, fallow (N=1)
Winter wheat, cover crop, silage maize (N=6)
Winter wheat (N=13)
Canola (N=2)
Winter wheat, winter rapeseed (N=3)
Wheat - maize rotation (N=1)
Spelt, cover crop, silage maize (N=1)
Winter wheat, cover crop, grain maize (N=1)
Winter barley, cover crop, silage maize (N=2)
Blueberry (N=1)
Crop(s) grown
−600
−500
−400
−300
−200
−100
0
100
200
300
400
500
600
700
800
Annual NEP (g C m^{-2})

Figure A1.1 Boxplots summarising the annual NEP database for croplands, displaying the range of annual NEP measurements grouped by the crop(s) grown during the measurement period. N= indicates the number of observations within each group and the width of each boxplot is proportional to the number of samples in each bin. Negative values indicate C accumulation and positive values indicate C loss.

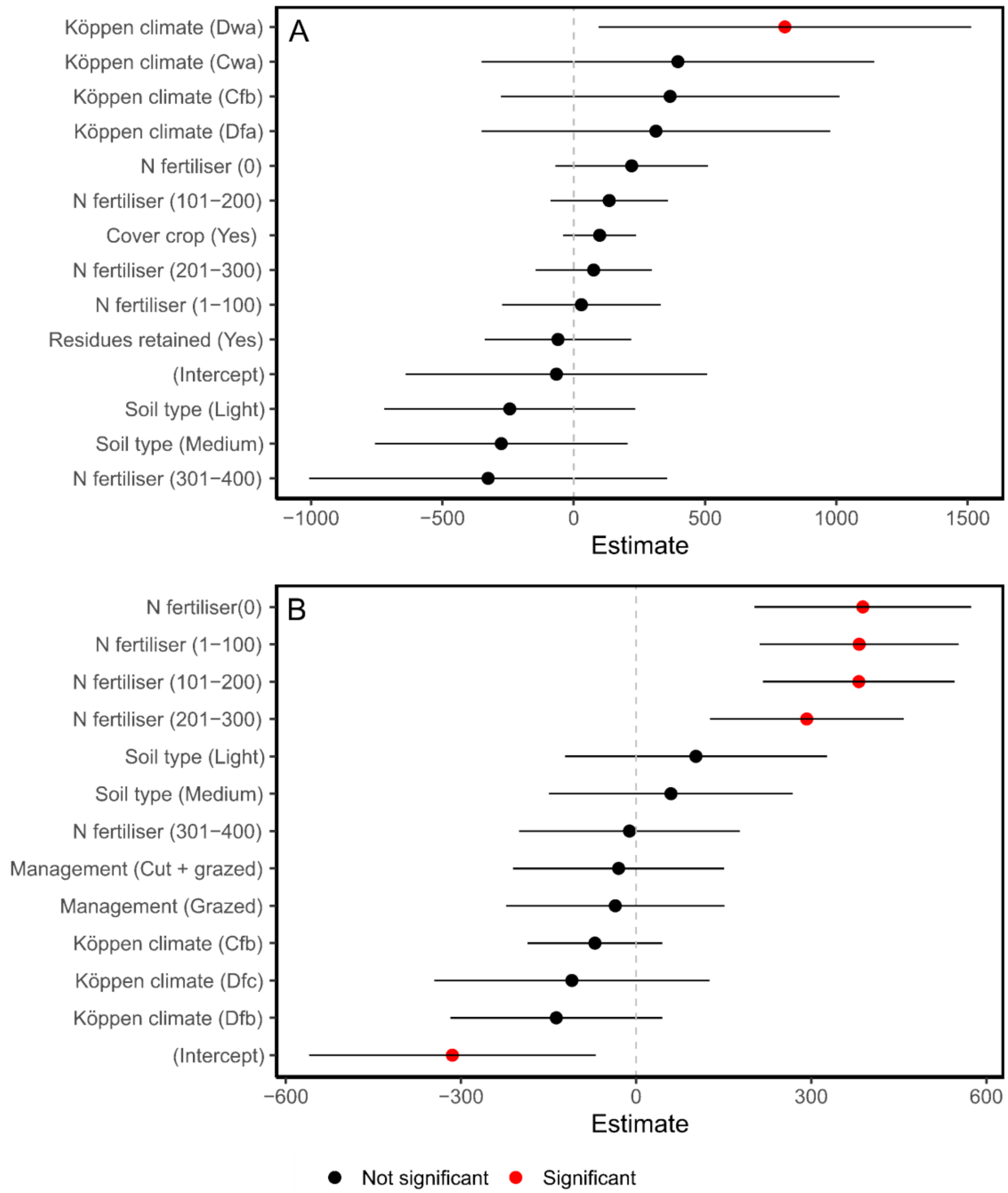


FIGURE A1.2: FOREST PLOT SHOWING RESULTS OF MIXED EFFECTS MODEL FOR (A) CROPLANDS AND (B) MANAGED GRASSLANDS. NOTE THAT AS COMPLETE CASES WERE REQUIRED FOR THIS ANALYSIS, N=75 FOR A AND N=98 FOR B.

A2 Supporting information for Chapter 3

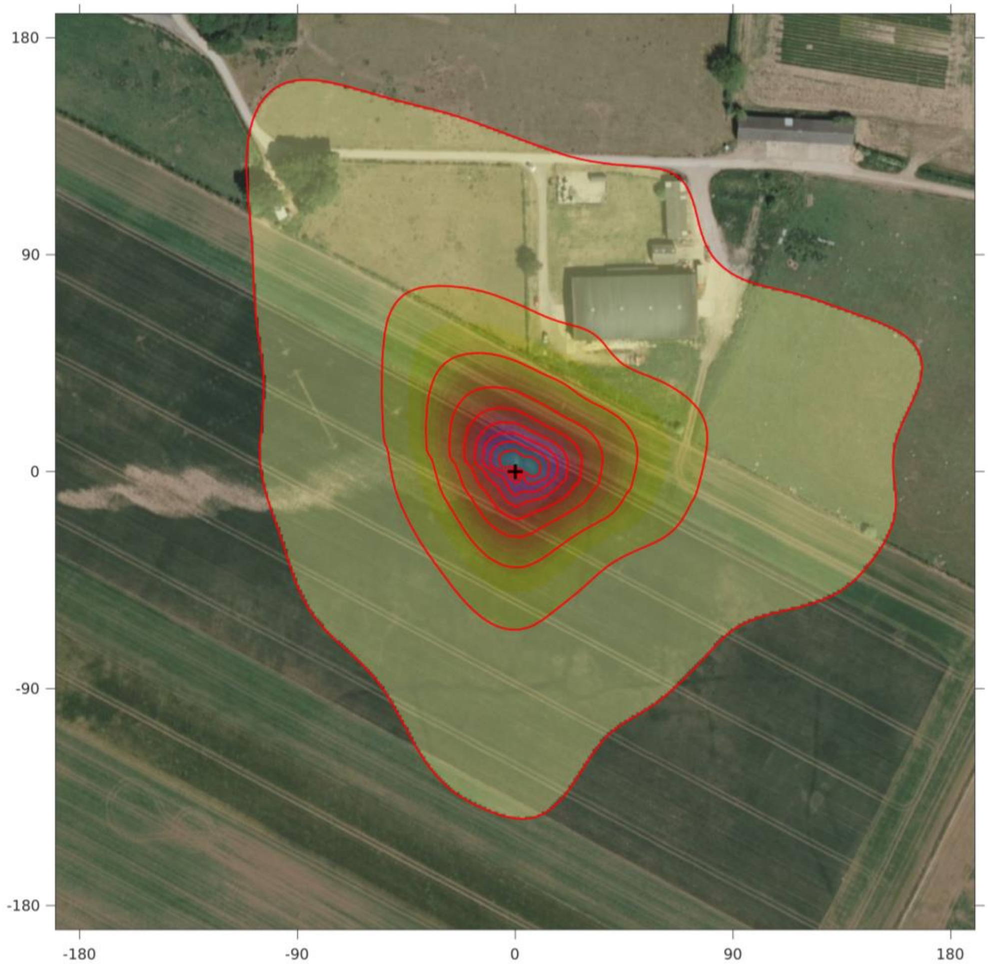


FIGURE A2.1 FLUX FOOTPRINT RADIUS PREDICTION AT MS (KLJUN ET AL., 2015).

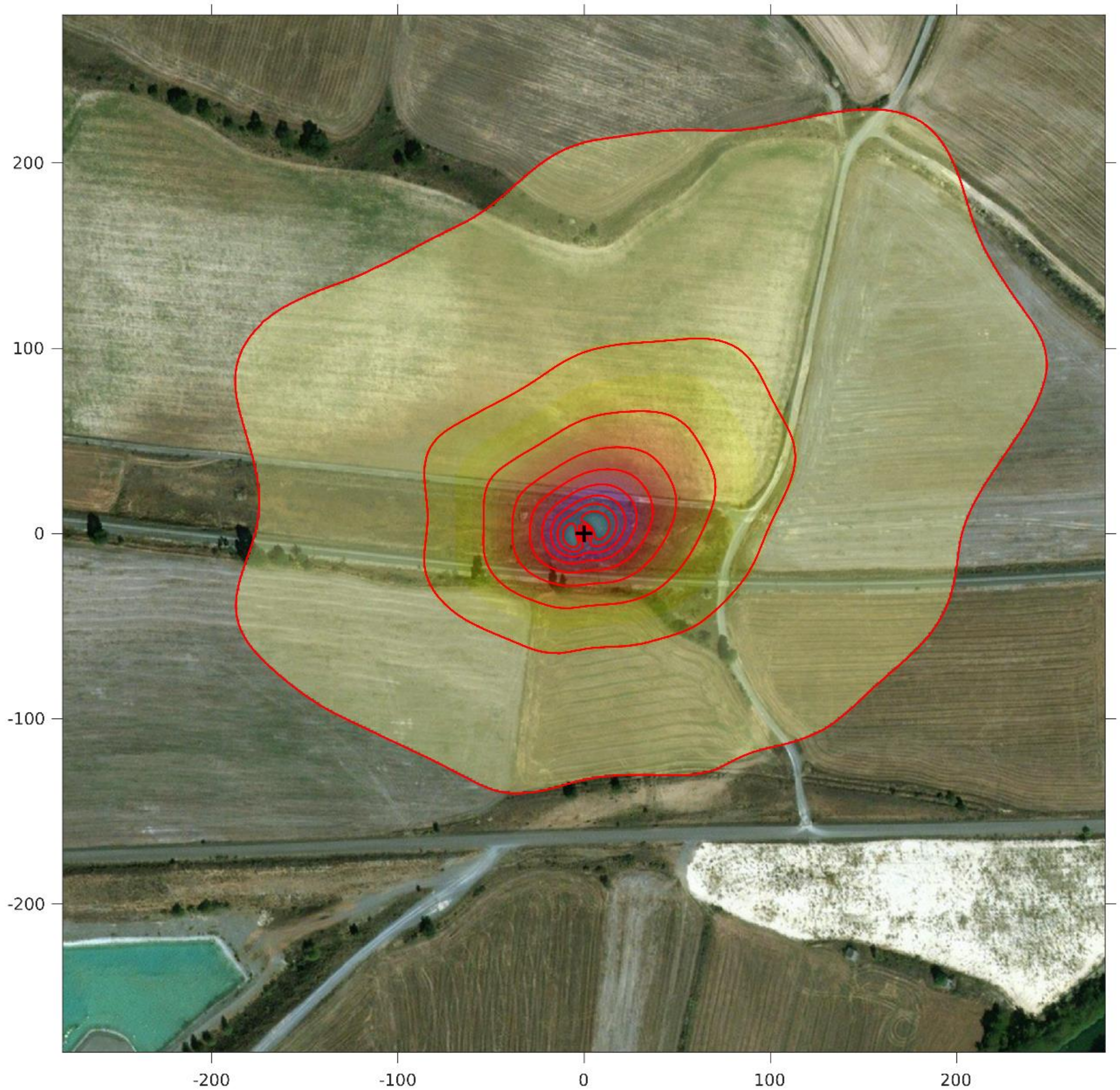


FIGURE A2.2 FLUX FOOTPRINT RADIUS PREDICTION AT PS (KLJUN ET AL., 2015).

TABLE A2.1 OVERVIEW OF NEE AND NEP MEASURED WITH EDDY COVARIANCE OVER THE MAIZE GROWING SEASON AS REPORTED IN THE LITERATURE. THE MICROMETEOROLOGICAL SIGN CONVENTION IS USED FOR NEE AND NEP, WHERE NEGATIVE VALUES INDICATE A GAIN OF C AND POSITIVE VALUES INDICATE A LOSS OF C.

Soil type	**Soil texture**	**Soil classification**	**Export**	**Yield ($g\ m^{-2}$)**	**CUE_h ($g\ C\ g\ C^{-1}$)**	**NEE ($g\ C\ m^{-2}$)**	**C_H ($g\ C\ m^{-2}$)**	**C_I ($g\ C\ m^{-2}$)**	**NEP ($g\ C\ m^{-2}$)**	**Location**	**Climate**	**Mean annual temperature (°C)**	**Mean annual precipitation (mm)**	**Reference**
Mineral	Silty	-	Grain	1157	-	-880	437	0	-443	USA	Temperate	-	-	Hollinger et al. (2005)
Mineral	Silty	-	Grain	899	-	-773	334	0	-439	USA	Temperate	-	-	Hollinger et al. (2005)
Mineral	Silty	-	Grain	1058	-	-702	339	0	-363	USA	Temperate	-	-	Hollinger et al. (2005)
Mineral	Silty clay loam	-	Grain	872	-	-510	335	0	-175	USA	Continental	-	-	Verma et al. (2005)
Mineral	Clay loam	-	Grain	690	-	-470	311	12	-171	China	Continental	-	-	Liu et al. (2019)
Mineral	Silt loam	-	Grain	750	0.3	-490	340	0	-150	France	Temperate	-	-	Loubet et al. (2010)
Mineral	Silty clay loam	-	Grain	772	-	-397	297	0	-100	USA	Continental	-	-	Verma et al. (2005)
Mineral	Sandy loam	-	Grain	980	0.35	-553	502	10	-61	China	Continental	8.2	475	Gao et al. (2017)

Mineral	Silty clay loam	-	Grain	1400	-	-572	538	0	-34	USA	Continental	-	-	Verma et al. (2005)
Mineral	Sandy loam	-	Grain	930	0.37	-491	476	10	-25	China	Continental	8.2	475	Gao et al. (2017)
Mineral	Sandy loam	-	Grain	918	0.36	-484	470	10	-24	China	Continental	8.2	475	Gao et al. (2017)
Mineral	Silty clay loam	-	Grain	1314	-	-529	518	0	-11	USA	Continental	-	-	Verma et al. (2005)
Mineral	Silty clay loam	-	Grain	1351	-	-517	521	0	4	USA	Continental	-	-	Verma et al. (2005)
Mineral	-	Cambisol	Wholecrop	1100	0.34	-473	484	0	11	Italy	Temperate	-	-	Alberti et al (2010)
Mineral	Loam	-	Grain	1666	0.33	-649	683	0	34	USA	Temperate	17.8	1200	Maleski et al. (2019)
Mineral	Clay loam	-	Grain	614	-	-196	263	10	57	China	Continental	-	-	Liu et al. (2019)
Mineral	Silty clay loam	-	Grain	1297	-	-424	503	0	79	USA	Continental	-	-	Verma et al. (2005)
Mineral	-	Cambisol	Wholecrop	920	0.25	-343	428	0	85	Italy	Temperate	-	-	Alberti et al (2010)
Mineral	Silty clay loam	-	Grain	1212	-	-381	470	0	89	USA	Continental	-	-	Verma et al. (2005)

Mineral	Silt loam	-	Grain	1300	0.58	-380	600	122	98	France	Temperate	-	-	Loubet et al. (2010)
Mineral	Clayey loam	Cambisol	Wholecrop	1230	0.51	-429	567	2	136	UK	Temperate	9.5	885	This study
Mineral	Sandy loam	-	Wholecrop	1848	0.71	-630	796	0	166	China	Continental	7.8	160	Guo et al. (2021)
Mineral	-	Luvisol	Wholecrop	2015	0.5	-600	770	0	170	Belgium	Temperate	10	800	Buysse et al. (2017)
Mineral	Sandy loam	-	Wholecrop	1410	0.56	-407	599	0	192	China	Continental	7.8	160	Guo et al. (2021)
Mineral	-	Gleysol	Wholecrop	1750	0.44	-597	790	0	193	Netherlands	Temperate	10.5	803	Jans et al. (2010)
Mineral	Sandy loam	-	Wholecrop	2032	0.55	-730	932	0	202	China	Continental	7.8	160	Guo et al. (2021)
Mineral	Clay loam	-	Wholecrop	1745	0.8	-351	806	249	206	France	Temperate	-	617	Tallec et al. (2013)
Mineral	Sandy loam	-	Wholecrop	1767	0.6	-527	758	0	231	China	Continental	7.8	160	Guo et al. (2021)
Peat		Histosol	Wholecrop	1130	0.35	-208	499	1	290	UK	Temperate	10.8	558	This study
Mineral	Sandy loam	-	Wholecrop	2277	0.67	-601	980	0	379	China	Continental	7.8	160	Guo et al. (2021)
Mineral	Silt loam	-	Wholecrop	2100	0.49	-240	826	171	415	New Zealand	Temperate	13.3	1249	Wall et al. (2020b)

Mineral	-	Luvisol	Wholecrop	1439	0.46	64	635	0	699	Canada	Continental	-	-	Eichelmann et al. (2016)
Mineral	Silt loam	-	Wholecrop	2670	0.5	-55	1083	177	851	New Zealand	Temperate	13.3	1249	Wall et al. (2020b)
Mineral (average)	-	-	-	1359	0.48	-479	669	24	72	-		-	-	-
Peat (average)	-	-	-	1130	0.35	-208	499	1	290	-		-	-	-

A3 Supporting information for Chapter 4

TABLE A3.1 AVERAGE DAILY AIR TEMPERATURE AND TOTAL PRECIPITATION OVER THE MAIZE, WINTER WHEAT AND VINING PEA GROWING SEASONS.

	Average daily air temperature (°C)	**Total precipitation (mm)**
Maize (02/06/2021-10/10/2021)	15.5	231
Winter wheat (21/10/2021-20/08/2022)	10.2	427
Vining pea (14/05/2023-20/07/2023)	13.9	73

TABLE A3.2 OVERVIEW OF NEE AND NEP MEASURED WITH EDDY COVARIANCE OVER MAIZE, WINTER WHEAT AND VINING PEA GROWING SEASONS AS REPORTED IN THE LITERATURE FOR SITES WITH TEMPERATE CLIMATES ONLY. THE MICROMETEOROLOGICAL SIGN CONVENTION IS USED FOR NEE AND NEP, WHERE NEGATIVE VALUES INDICATE C ACCUMULATION AND POSITIVE VALUES INDICATE C LOSS.

Crop	**Harvested**	**Yield (t DM ha^{-1})**	**CUE_h (g C g C^{-1})**	**NEE (g C m^{-2})**	**NEE (t CO_2-eq ha^{-1})**	**C_H (g C m^{-2})**	**C_I (g C m^{-2})**	**NEP (g C m^{-2})**	**NEP (t CO_2-eq ha^{-1})**	**Location**	**Mean annual temperature (° C)**	**Mean annual precipitation (mm)**	**Reference**
Maize	Grain	11.656.8	-	-880.4	-32	436.9	0	-443.5	-16	USA	-	-	Hollinger et al. (2005)
	Grain	9	-	-733.4	-27	339.5	0	-393.9	-14	USA	-	-	Hollinger et al. (2005)
	Grain	10.6	-	-702.4	-26	399.4	0	-303	-11	USA	-	-	Hollinger et al. (2005)
	Grain	6.9	-	-469.9	-17	311.2	1.3	-170.7	-6	China	-	-	Liu et al. (2019)
	Grain	-	-	-373	-14	383	1	9	0	China	13.1	528	Yue et al. (2023)
	Grain	-	-	-279	-10	297	2	16	1	China	13.1	528	Yue et al. (2023)
	Grain	-	-	-280	-10	312	2	30	1	China	13.1	528	Yue et al. (2023)
	Wholecrop	30.2	0.33	-649	-24	683	-	34	1	USA	17.8	1200	Maleski et al. (2019)
	Grain	-	-	-278	-10	327	1	48	2	China	13.1	528	Yue et al. (2023)
	Grain	-	-	-165.6	-6	215.3	0	49.7	2	China	13.1	528	Li et al. (2006)
	Grain	-	-	-295	-11	351	1	55	2	China	13.1	528	Yue et al. (2023)
	Grain	6.1	-	-196.4	-7	263.2	1.3	56.6	2	China	-	-	Liu et al. (2019)
	Grain	-	-	-207	-8	299	2	90	3	China	13.1	528	Yue et al. (2023)
	Grain	-	-	-108	-4	215	2	105	4	China	13.1	528	Yue et al. (2023)
	Grain	-	-	-102	-4	220	5	113	4	China	13.1	528	Yue et al. (2023)
	Grain	-	-	-235	-9	354	1	118	4	China	13.1	528	Yue et al. (2023)
	Grain	-	-	-160	-6	280	1	119	4	China	13.1	528	Yue et al. (2023)

	Grain	-	-	-183	-7	306	1	122	4	China	13.1	528	Yue et al. (2023)
	Grain	9.8	0.42	-244	-9	368	0	124	5	China	13.3	532	Lei and Yang (2010)
	Wholecrop	12.3	0.51	-429	-16	567	2	136	5	UK	9.5	639	This study
	Grain	-	-	-190	-7	353	1	162	6	China	13.1	528	Yue et al. (2023)
	Grain	-	-	-166	-7	353	1	186	7	China	13.1	528	Yue et al. (2023)
	Grain	-	-	-120.1	-4	344.6	0	224.5	8	China	13.1	528	Li et al. (2006)
	Grain	-	-	-151	-6	408	1	256	9	China	13.1	528	Yue et al. (2023)
	Wholecrop	-	0.38	-240	-9	825	160	425	16	New Zealand	12.5	480	Wall et al. (2020b)
	Wholecrop	17.5	-	-351	-13	806	0	455	17	France	-	-	Tallec et al. (2013)
	Wholecrop	-	0.64	-55	-2	1081	177	849	31	New Zealand	12.5	480	Wall et al. (2020b)
Winter wheat	Grain	-	0.08	-494	-18	79	2	-417	-15	USA	9	550	Waldo et al. (2016)
	Grain	-	0.23	-671	-25	287	4	-388	-14	USA	10	247	Waldo et al. (2016)
	Grain	-	0.27	-730	-27	310	0	-280	-10	Belgium	9.8	800	Aubinet et al. (2009)
	Straw					140							
	Grain	3.5	0.15	-347	-13	142	0	-205	-8	Brazil	17.7	1907	Veeck et al. (2022)
	Grain	11.2	-	-471	-17	297	0	-174	-6	France	-	-	Tallec et al. (2013)
	Grain	-	-	-396	-15	229	7	-174	-6	China	13.1	528	Yue et al. (2023)
	Grain + straw	13.1	-	-538	-20	386	0	-152	-6	France	-	-	Tallec et al. (2013)
	Grain	6.8	0.23	-394	-14	261	0	-133	-5	China	13.3	532	Lei and Yang (2010)
	Grain	10.3	0.37	-648	-24	403	116	-148	-5	UK	9.5	639	This study
	Straw	5.2				213							
	Grain	-	-	-317	-12	208	12	-121	-4	China	13.1	528	Yue et al. (2023)
	Grain	8.3	0.4	-627	-23	342	8	-111	-4	Germany	9.9	698	Schmidt et al. (2012)
	Straw	1.5				182			-4				
	Grain	-	-	-336	-12	238	6	-104	-4	China	13.1	528	Yue et al. (2023)
	Grain	6.6	0.31	-395	-15	295	0	-100	-3	China	13.3	532	Lei and Yang (2010)
	Grain	-	-	-354	-13	271	8	-91	-3	China	13.1	528	Yue et al. (2023)
	Grain	-	-	-329	-12	263	11	-77	-3	China	13.1	528	Yue et al. (2023)

	Grain	6.5	0.26	-326	-12	250	0	-76	-3	China	13.3	532	Lei and Yang (2010)
	Grain	-	0.35	-630	-23	370	0	-70	-3	Belgium	9.8	800	Aubinet et al. (2009)
	Straw					190							
	Grain	-	-	-278	-10	220	8	-66	-2	China	13.1	528	Yue et al. (2023)
	Grain	7.3	0.45	-537	-20	283	8	-66	-2	Germany	9.9	698	Schmidt et al. (2012)
	Straw	1.3				196							
	Grain	6.4	0.32	-303	-11	247	0	-56	-2	China	13.3	532	Lei and Yang (2010)
	Grain	6.1	-	-225.7	-8	235.3	8.2	-36.3	-1	China	-	-	Liu et al. (2019)
	Grain	-	-	-222	-8	221	20	-21	-1	China	13.1	528	Yue et al. (2023)
	Grain	-	-	-272	-10	262	11	-21	-1	China	13.1	528	Yue et al. (2023)
	Grain	-	-	-217	-8	215	9	-11	0	China	13.1	528	Yue et al. (2023)
	Grain	-	-	-215	-8	224	10	-1	0	China	13.1	528	Yue et al. (2023)
	Grain	-	-	-297	-11	312	11	4	0	China	13.1	528	Yue et al. (2023)
	Grain	6.4	-	-238.1	-9	278.1	7.7	24.6	1	China	-	-	Liu et al. (2019)
	Grain	-	-	-152.2	-6	210.1	0	57.9	2	China	13.1	528	Li et al. (2006)
	Grain	-	-	-143	-5	241	11	87	3	China	13.1	528	Yue et al. (2023)
	Grain	-	-	-178	-7	306	11	117	4	China	13.1	528	Yue et al. (2023)
	Grain	-	-	-77.6	-3	203.6	0	126	5	China	13.1	528	Li et al. (2006)
	Grain	-	-	-26	-1	228	10	192	7	China	13.1	528	Yue et al. (2023)
Vining pea	Pods	1.1	0.09	-193	-7	45	6	-154	-9	UK	9.5	639	This study

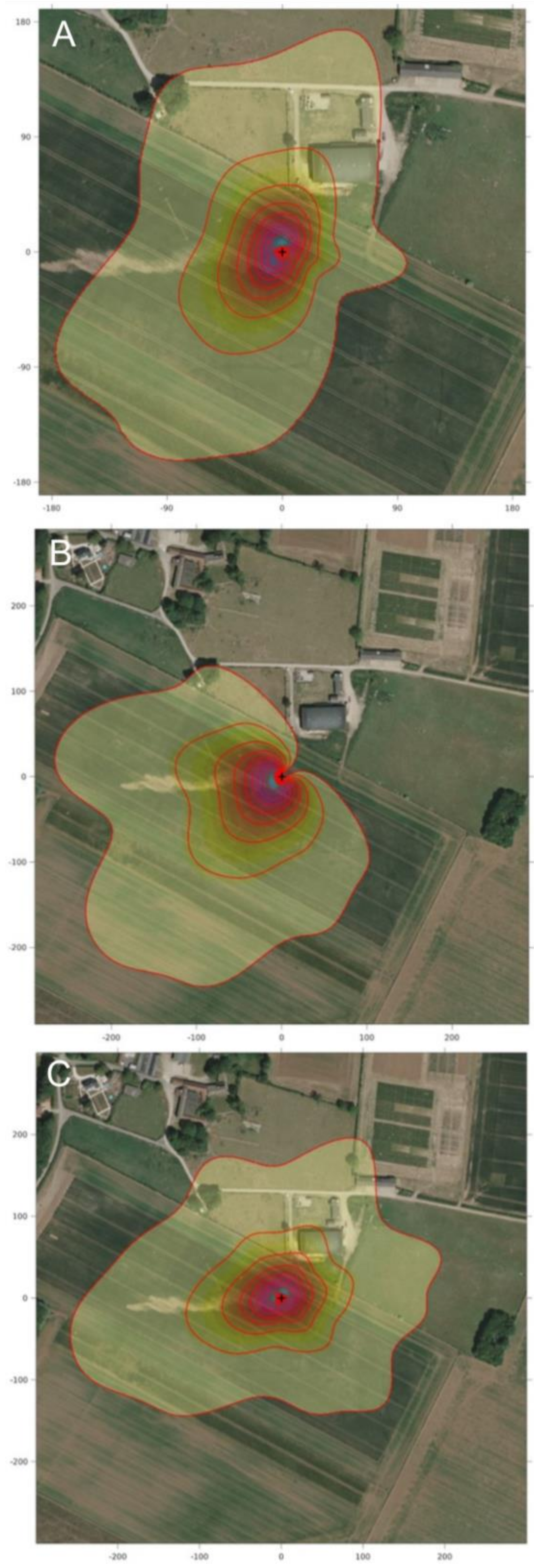

A
180
90
0
-90
-180
-180
-90
0
90
180
B
200
100
0
-100
-200
-200
-100
0
100
200
C
200
100
0
-100
-200
-200
-100
0
100
200

FIGURE A3.1 FLUX FOOTPRINT RADIUS PREDICTION IN CF DURING (A) 2021, (B) 2022 AND (C) 2023 (KLJUN ET AL., 2015).

A4 Supporting information for Chapter 5

TABLE A4.1 MANAGEMENT IN CF OVER THE ONE-YEAR MEASUREMENT PERIOD.

Date	Management
20/10/2021	Non-inversion tillage: 25 cm
21/10/2021	Non-inversion tillage: 25 cm Planted winter wheat (Extase variety) using precision drill: 440 seeds m^{-2}
10/11/2021	Herbicide (Flufenacet + pendimethalin): 4 L ha^{-1}
01/02/2022 21/03/2022	Fertiliser (Pig slurry): 30 m^{-3} ha^{-1} (of which 87 kg N ha^{-1}, 54.9 kg P ha^{-1}, 61.8 kg K ha^{-1} and 450 kg C ha^{-1})
16/04/2022	Fertiliser (N26+5SO3): 120 kg N ha^{-1}, 23 kg S ha^{-1}
26/04/2022	Fungicide (Bixafen, fluopyram + prothioconazole): 0.9 L ha^{-1} Plant growth regulator (Chlormequat chloride): 2.2 L ha^{-1}
14/05/2022	Herbicide (Pyroxsulam + floraulam): 265 g ha^{-1}
20/05/2022	Fungicide (Fenpicoxamid + prothioconazole): 1.5 L ha^{-1} Plant growth regulator (Mepiquat chloride + 2-chloroethylphosphoric acid): 1 L ha^{-1}
20/08/2022	Winter wheat harvest: 15.5 t ha^{-1} dry matter (10.3 t ha^{-1} grain, 5.2 t ha^{-1} straw)
October 2022	Non-inversion tillage: 25 cm

TABLE A4.2 OVERVIEW OF ANNUAL NEE AND NEP MEASURED WITH EDDY COVARIANCE FOR CROPLANDS GROWING WINTER WHEAT AND AGRICULTURAL GRASSLANDS MANAGED WITH CUTTING AND GRAZING OVER ONE YEAR AS REPORTED IN THE LITERATURE FOR SITES WITH TEMPERATE CLIMATES ONLY. THE MICROMETEOROLOGICAL SIGN CONVENTION IS USED FOR NEE AND NEP, WHERE NEGATIVE VALUES INDICATE A GAIN OF C AND POSITIVE VALUES INDICATE A LOSS OF C.

System	Harvested	Yield (t DM ha^{-1})	CUE_h (g C g C g^{-1})	NEE (g C m^{-2})	NEE (t CO_2-eq ha^{-1})	C_H (g C m^{-2})	C_I (g C m^{-2})	NEP (g C m^{-2})	NEP (t CO_2-eq ha^{-1})	Location	Mean annual temperature (° C)	Mean annual precipitation (mm)	Reference
Cropland (winter wheat)	Grain	10.2	-	-654.3	-24	325.4	0	-328.9	-12	Germany	9.4	889	Poyda et al. (2019)
	Grain	9.5	-	-515.6	-19	345.4	42	-212.2	-8	Germany	9.4	889	Poyda et al. (2019)
	Grain	9	-	-490.6	-18	308.9	29	-209.7	-8	Germany	9.4	889	Poyda et al. (2019)
	Grain	9.4	-	-421.6	-15	301	72	-192.6	-7	Germany	9.4	889	Poyda et al. (2019)
	Grain	7.9	-	-436	-16	262.2	0	-173.8	-6	Germany	9.4	889	Poyda et al. (2019)
	Grain	10.5	-	-429.7	-16	295.8	0	-133.9	-5	Germany	9.4	889	Poyda et al. (2019)
	Grain	9.9	-	-432.4	-16	302.4	0	-130	-5	Germany	9.4	889	Poyda et al. (2019)
	Grain	10.1	-	-396.3	-15	378	72	-90.3	-3	Germany	9.4	889	Poyda et al. (2019)
	Grain	8.9	-	-373	-14	462.2	112	-22.8	-1	Germany	9.4	889	Poyda et al. (2019)
	Grain	10.3	0.37	-526	-19	403	116	-26	-1	UK	9.5	639	This study
	Straw	5.2				213							
	Grain	9.9	-	-269.7	-10	289.8	0	20.1	1	Germany	9.4	889	Poyda et al. (2019)
	Grain	10.4	-	-291	-11	316.3	0	25.3	1	Germany	9.4	889	Poyda et al. (2019)
	Grain	7.9	-	-284	-10	276.8	0	-7.2	0	Germany	9.4	889	Poyda et al. (2019)
	Grain	8.3	-	-247.7	-9	322.8	0	75.1	3	Germany	9.4	889	Poyda et al. (2019)
	Grain	9.4	-	-20.3	-8	347	0	116.7	4	Germany	9.4	889	Poyda et al. (2019)
	Grain	9.3	-	-399.6	-15	650.2	0	250.6	9	Germany	9.4	889	Poyda et al. (2019)
	Grain	6.9	-	148.7	5	233.5	0	382.2	14	Germany	9.4	889	Poyda et al. (2019)
Cut and grazed grassland	-	-	-	-600	-22	433	82.4	-249.4	-9	New Zealand	12.1	640	Laubach et al. (2023)
	-	-	0.007	-189.5	-7	12.8	14.6	-191.3	-7	New Zealand	13.8	1126	Rutledge et al. (2015)
	-	-	0.01	-183.7	-7	29.6	19.8	-173.9	-6	New Zealand	13.8	1126	Rutledge et al. (2015)
	-	-	0.04	-227.1	-8	107.5	13.7	-133.3	-5	New Zealand	13.8	1126	Rutledge et al. (2015)
	-	-	0.2	-364	-13	648	395	-111	-4	New Zealand	13.3	1250	Wall et al. (2019)
	-	-	0.2	-164	-6	518	384	-30	-1	New Zealand	13.3	1250	Wall et al. (2019)
	-	-	-	-485	-18	584	125.4	-26.4	-1	New Zealand	12.1	640	Laubach et al. (2023)
	-	-	0.1	-253	-9	441	192	-4	-1	New Zealand	13.3	1249	Wall et al. (2023a)
	-	-	0.1	-104	-4	103	0	-1	0	USA	-	-	Skinner (2013)
	-	-	0.2	-341	-13	523	170	12	0	New Zealand	13.3	1249	Wall et al. (2023a)
	-	-	-	-257	-9	473	196	20	1	New Zealand	13.3	1249	Wall et al. (2020b)
	-	-	0.2	-361	-13	592	208	23	1	New Zealand	13.3	1249	Wall et al. (2023a)
	-	-	-	-155	-6	191	8	28	1	USA	-	-	Skinner (2008)
	-	-	-	-281	-10	508	191	36	1	New Zealand	13.3	1249	Wall et al. (2020b)
	-	-	-	-33	-1	103	21	49	2	USA	-	-	Skinner (2008)
	-	-	-	-62	-2	127	13	52	2	USA	-	-	Skinner (2008)
	-	-	-	-9	0	83	10	64	2	USA	-	-	Skinner (2008)
	-	-	-	-338	-12	563	157.8	67.2	2	New Zealand	12,1	640	Laubach et al. (2023)
	-	-	0.1	-16	-1	97	0	81	3	USA	-	-	Skinner (2013)
	-	-	0.1	-40	-1	128	0	88	3	USA	-	-	Skinner (2013)
	-	-	0.1	-15	-1	109	0	94	3	USA	-	-	Skinner (2013)
	-	-	0.1	10	0	92	0	102	4	USA	-	-	Skinner (2013)

	-	-	0.2	-52	-2	372	213	107	4	Belgium	10	847	de la Motte et al. (2016)
	-	-	0.2	-252	-9	397	23	122	4	New Zealand	12.1	640	Laubach et al. (2019)
	-	-	0.1	-41	-2	164	0	123	5	USA	-	-	Skinner (2013)
	-	-	0.1	77	3	76	0	153	6	USA	-	-	Skinner (2013)
	-	-	-	29	1	162	32	159	6	USA	-	-	Skinner (2008)
	-	-	0.1	23	1	145	0	168	6	USA	-	-	Skinner (2013)
	-	-	-	-260	-10	638	157.8	220.2	8	New Zealand	12.1	640	Laubach et al. (2023)
	-	-	0.2	-66	-2	544	226	252	9	New Zealand	13.3	1249	Wall et al. (2023a)
	-	-	0.1	126	5	148	0	274	10	USA	-	-	Skinner (2013)
	-	-	0.5	-187	-7	564	100	277	10	New Zealand	12.1	640	Laubach et al. (2019)
	-	-	0.2	37	1	313	39	311	11	UK	9.5	639	This study
	-	-	0.1	166	6	171	0	337	12	USA	-	-	Skinner (2013)

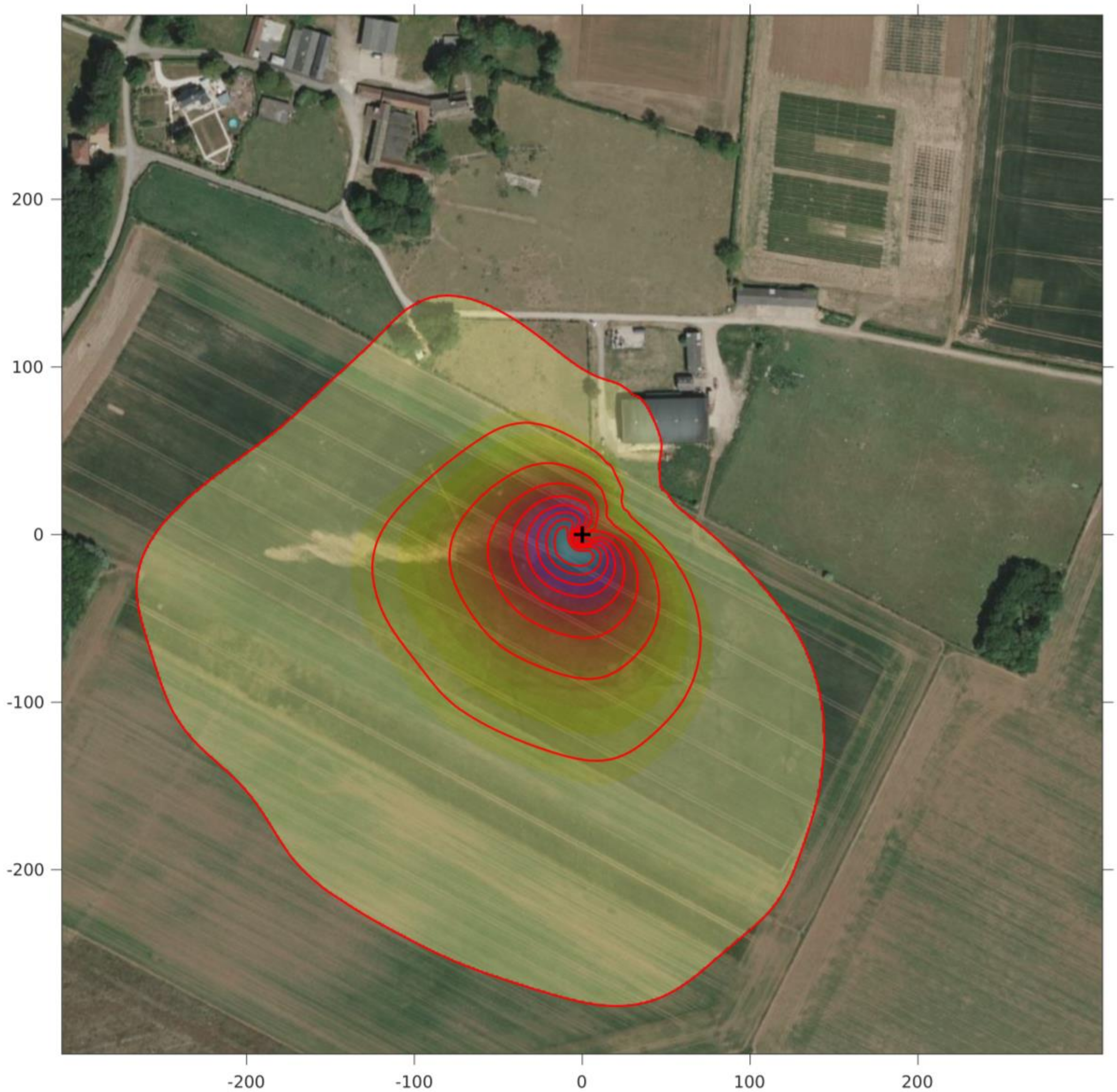


FIGURE A4.1 FLUX FOOTPRINT RADIUS PREDICTION IN CF OVER THE ONE-YEAR MEASUREMENT PERIOD (KLJUN ET AL., 2015).

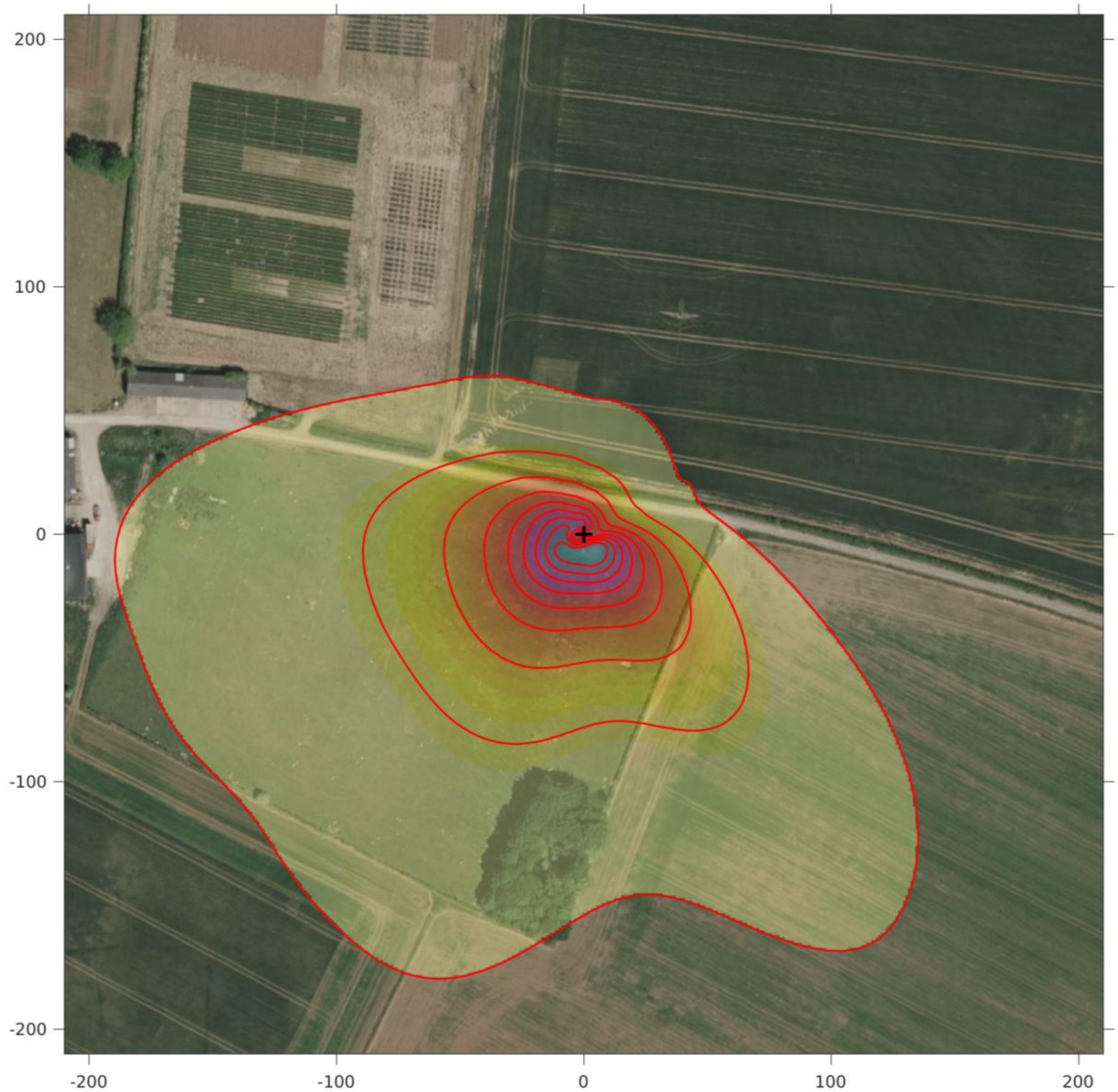


FIGURE A4.2 FLUX FOOTPRINT RADIUS PREDICTION IN PP OVER THE ONE-YEAR MEASUREMENT PERIOD (KLJUN ET AL., 2015).

A5 Supporting information for Chapter 6

TABLE A5.1 SOIL PROPERTIES OF EXPERIMENTAL FIELD, OBTAINED FROM [A] UK SOIL OBSERVATORY (UKRI, 2021); [B] SOIL SAMPLING AT 0-30 CM DEPTH, MEAN ± STANDARD DEVIATION (MEASURED JUNE 2021, N=9); [C] BASED ON SOIL SAMPLING AT 0-10 CM DEPTH; * BASED ON OLSEN'S PHOSPHORUS.

Soil type [a]	Cambisol
Soil texture [a]	Loam
Organic matter content (%) [b]	6.7 ± 0.6
pH ($CaCl_2$) [b]	6.9 ± 0.2
Bulk density (g cm^{-3}) [b]	1.3 ± 0.1
Total carbon (g kg^{-1}) [b]	39.5 ± 9
Total organic carbon (g kg^{-1}) [b]	22.9 ± 4.9
Total nitrogen (g kg^{-1}) [b]	2.3 ± 0.6
Plant available nitrogen (g kg^{-1}) [b]	0.013 ± 0
Olsen's phosphorus (mg kg^{-1}) [b]	36 ± 8
Phosphorus index [c] *	3

TABLE A5.2 APPLICATION RATES FOR EACH TREATMENT; EACH APPLICATION WAS SCALED RELATIVE TO THE SIZE OF EACH CHAMBER AND PLOT (IF = INORGANIC FERTILISER, UPS = UNTREATED PIG SLURRY AND TPS = TREATED PIG SLURRY).

		Application 1 (23/03/2022)	**Application 2 (11/04/2022)**	**Application 3 (25/04/2022)**	**Application 4 (10/05/2022)**	**Total**
IF	Fertiliser	Inorganic fertiliser (granular ammonium nitrate and sulphur compound (30N, 17.5SO3))	Inorganic fertiliser (liquid ammonium nitrate and sulphur compound (26 N, 5SO3))	Inorganic fertiliser (liquid ammonium nitrate and sulphur compound (26 N, 5SO3))		

	Total N (kg ha^{-1})	60	120	40		220
	Available N (kg ha^{-1})	60	120	40		220
UPS	Fertiliser	Untreated pig slurry	Untreated pig slurry	Inorganic fertiliser (liquid ammonium nitrate and sulphur compound (26 N, 5SO3))	Inorganic fertiliser (liquid ammonium nitrate and sulphur compound (26 N, 5SO3))	
	Total N (kg ha^{-1})	102	117	100	10	349
	Available N (kg ha^{-1})	53	57	100	10	220
TPS	Fertiliser	Treated pig slurry	Treated pig slurry	Inorganic fertiliser (liquid ammonium nitrate and sulphur compound (26 N, 5SO3))	Inorganic fertiliser (liquid ammonium nitrate and sulphur compound (26 N, 5SO3))	
	Total N (kg ha^{-1})	70	72	100	42	284
	Available N (kg ha^{-1})	52	59	100	42	253

TABLE A5.3 MEAN DAILY AND MEAN CUMULATIVE CO_2 FLUXES OVER THE 83-DAY MEASUREMENT PERIOD ± STANDARD DEVIATION (SD) FOR EACH FERTILISER TREATMENT (IF = INORGANIC FERTILISER, UPS = UNTREATED PIG SLURRY, TPS = TREATED PIG SLURRY). ACROSS EACH ROW, DIFFERENT LETTERS INDICATE SIGNIFICANT DIFFERENCES IN THE VARIABLE OF INTEREST BETWEEN FERTILISER TREATMENTS.

	IF	**UPS**	**TPS**
Mean daily ± SD (g C m^{-2} day^{-1})	-18.2 ± 1.4 a	-15.4 ± 0.1 b	-16.6 ± 0.6 b

Mean cumulative ± SD (g C m^{-2})	-1561.2 ± 116.9 a	-1325.4 ± 7.8 b	-1428.1 ± 4.7 ab

TABLE A5.4 SIGNIFICANT INTERACTIONS IDENTIFIED BETWEEN ENVIRONMENTAL AND TREATMENT VARIABLES ON N_2O FLUXES (MMOL M^{-2} $2HR^{-1}$) EXCLUDING FLUXES RECORDED WITHIN 0-7 DAYS OF THE FIRST TWO FERTILISER APPLICATIONS (WFPS = WATER-FILLED PORE SPACE, PAR = PHOTOSYNTHETICALLY ACTIVE RADIATION).

Variables	Estimate	P value
Pig slurry + WFPS	2.9e+02	0.00
Soil temperature + WFPS	-2.4e+01	9.8e-05
Soil temperature + PAR	-6.1	0.004
WFPS + PAR	-1.4	0.003
Pig slurry + precipitation	-3e+04	0.02
Soil temperature + WFPS + PAR	1.4e-01	0.001
Pig slurry + soil temperature + precipitation	2.9e+03	0.007
Pig slurry + WFPS + precipitation	6.1e+02	0.007
Pig slurry + soil temperature + WFPS + precipitation	-6.2e+01	0.002

TABLE A5.5 AVERAGE DAILY AIR TEMPERATURE AND TOTAL RAINFALL PER WINTER WHEAT GROWTH STAGE OVER THE MEASUREMENT PERIOD.

Growth stage	Start date	End date	Number of days	Average daily air temperature (°C)	Total rainfall (mm)
Tillering S5	20/03/2022	10/04/2022	21	7.11	31.07
Extension S6	11/04/2022	24/04/2022	14	10.53	7.44
Extension S7	25/04/2022	02/05/2022	8	8.74	0.55
Extension S8	03/05/2022	08/05/2022	5	11.35	0
Extension S9	09/05/2022	16/05/2022	7	13.26	19.35
Extension S10	17/05/2022	23/05/2022	7	14.02	8.24
Heading	24/05/2022	06/06/2022	14	14.5	15.46
Flowering	07/06/2022	13/06/2022	7	11.63	24.71

Table A5.6 Comparison of cumulative growing season CO_2-equivalent emissions of N_2O and CH_4 (g CO_2-equivalent m^{-2}) from winter wheat fertilised with inorganic fertiliser measured using chamber methodologies amongst the literature. Note that all GHG flux measurements were taken using static manual chambers.

Nitrogen fertiliser rate (kg N ha^{-1})	**CO_2-equivalent emissions (g CO_2-equivalent m^{-2})**	**Reference**
220	34.2	This study
150	55.2	Huang et al. 2018
300	99.5	Huang et al. 2018
150	38.85	Huang et al. 2013
300	54.5	Huang et al. 2013
150	77.25	Huang et al. 2013
300	102.55	Huang et al. 2013
300	15	Sainju et al. 2022
300	21.6	Sainju et al. 2022
300	25.8	Sainju et al. 2022
300	27.2	Sainju et al. 2022
300	30.9	Sainju et al. 2022
300	35.8	Sainju et al. 2022
300	35.9	Sainju et al. 2022
300	36	Sainju et al. 2022

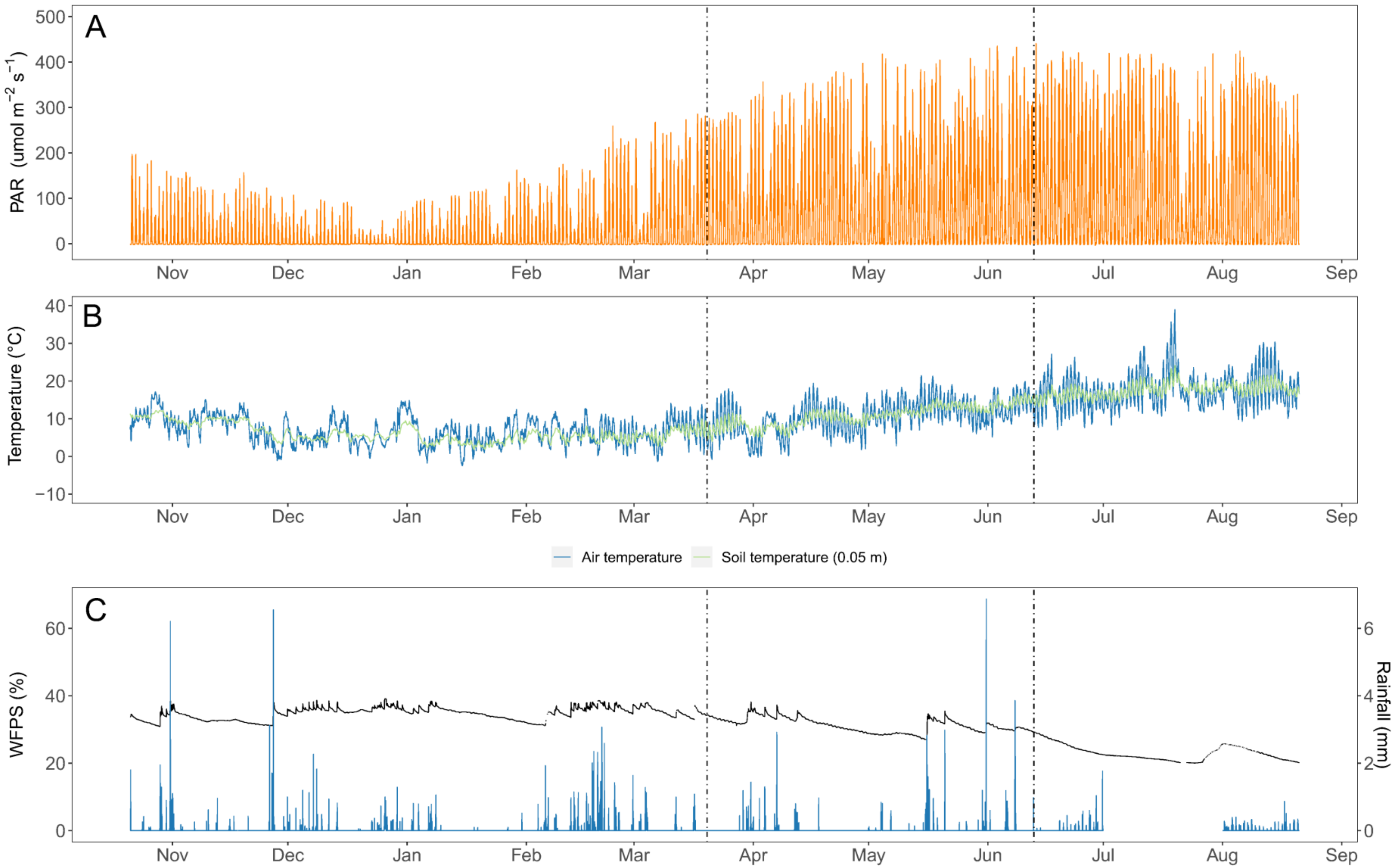
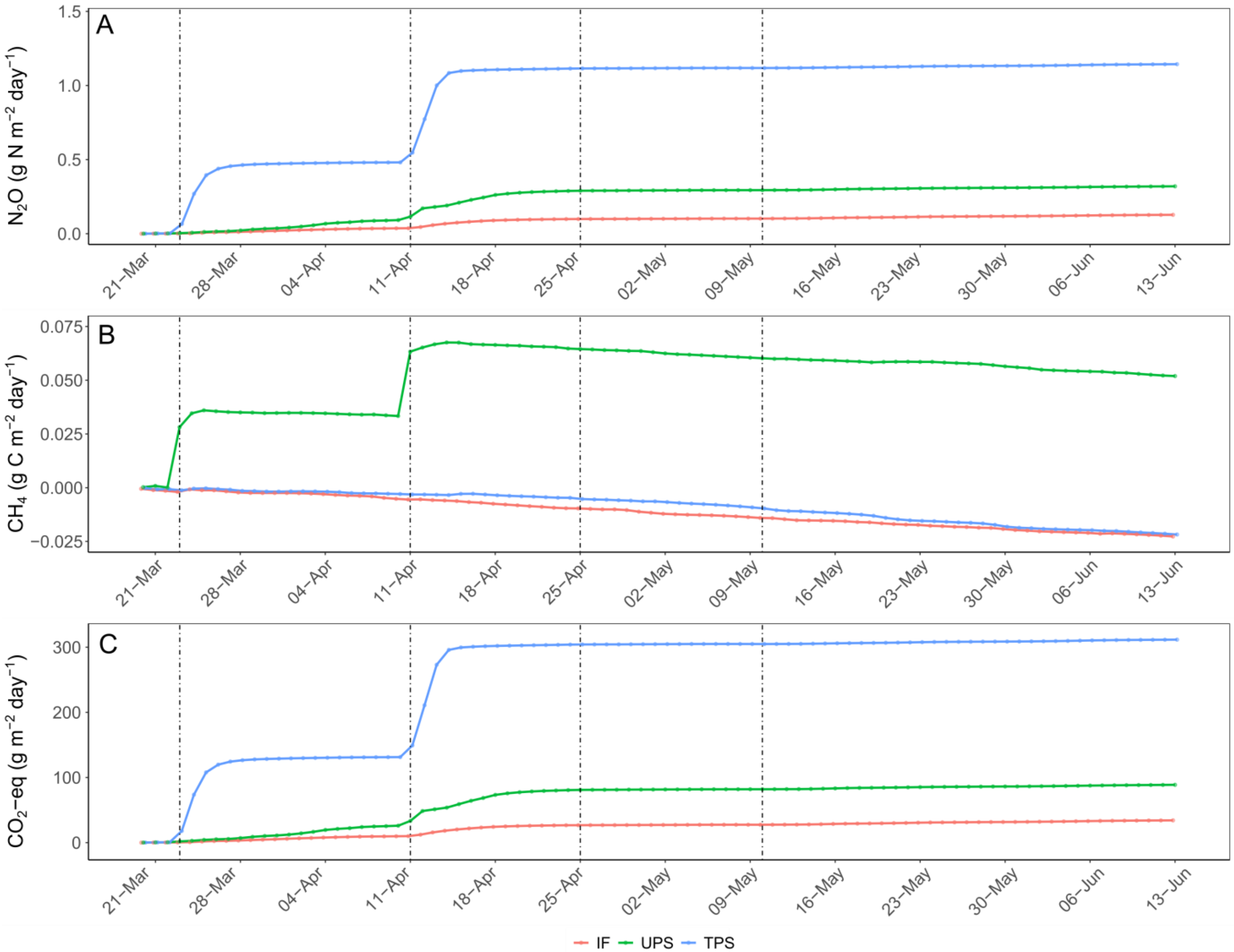

A
PAR (umol m^{-2} s^{-1})
500
400
300
200
100
0
Nov
Dec
Jan
Feb
Mar
Apr
May
Jun
Jul
Aug
Sep
B
Temperature (°C)
40
30
20
10
0
−10
Air temperature
Soil temperature (0.05 m)
C
WFPS (%)
60
40
20
0
Rainfall (mm)
6
4
2
0

Figure A5.1 (A) Photosynthetically active radiation (PAR), (B) air and soil temperature and (C) water-filled pore space (WFPS) and rainfall measured over the WW growing season (21/10/2021-20/08/2022). Dashed lines represent the start and end of the GHG measurement period (20/03/2022-13/06/2022). Gaps in precipitation are a result of errors with instrument data collection. Photosynthetically active radiation, air temperature and precipitation data from COSMOS measurement station at Spen Farm (UKCEH, 2023); soil temperature and soil moisture data measured in-field at a depth of 0.05 m (TEROS 11, METER Group Inc., USA).



A
B
C
N_2O (g N m^{-2} day^{-1})
CH_4 (g C m^{-2} day^{-1})
CO_2–eq (g m^{-2} day^{-1})
1.5
1.0
0.5
0.0
0.075
0.050
0.025
0.000
−0.025
300
200
100
0
21–Mar
28–Mar
04–Apr
11–Apr
18–Apr
25–Apr
02–May
09–May
16–May
23–May
30–May
06–Jun
13–Jun
IF
UPS
TPS

Figure A5.2 Cumulative fluxes of (A) N_2O, (B) CH_4 and (C) CO_2-equivalent fluxes of N_2O and CH_4 for each fertiliser treatment (IF = inorganic fertiliser, UPS = pig slurry, TPS = treated pig slurry). Each data point represents the mean of three chambers used per treatment. Vertical dashed lines represent the four fertiliser applications. Error bars have been removed to aid visualisation.

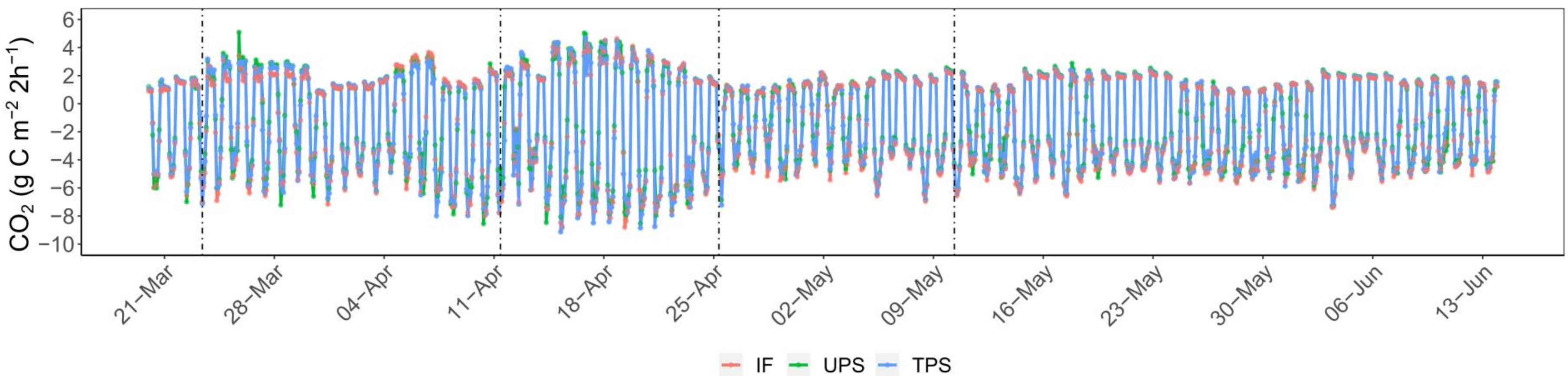


Figure A5.3 2-hour fluxes of CO_2 for each fertiliser treatment (IF = inorganic fertiliser, UPS = untreated pig slurry, TPS = treated pig slurry). Each data point represents the mean of three chambers used per treatment and vertical dashed lines represent the split applications of fertilisers. Error bars have been removed to aid visualisation.

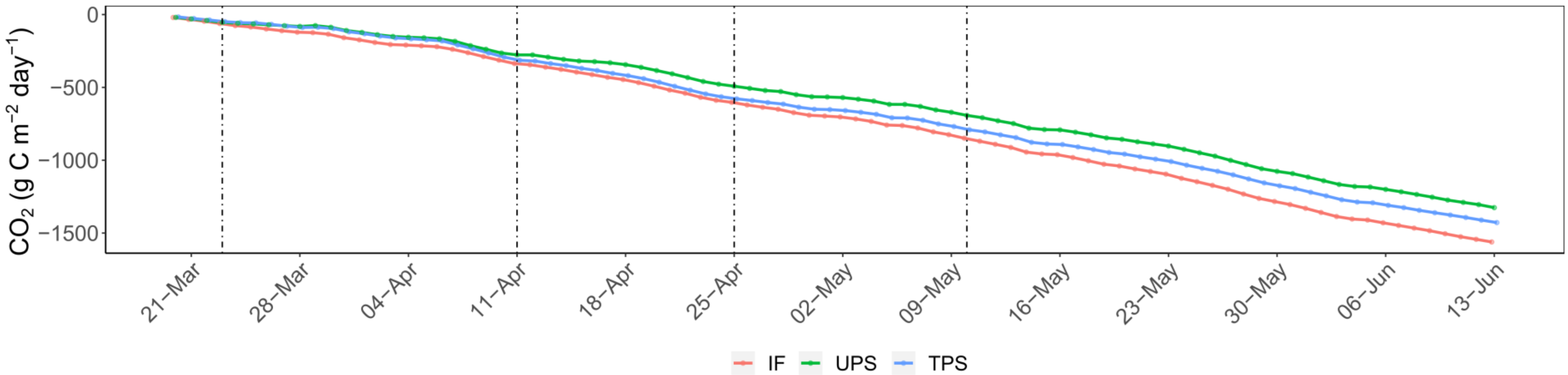


FIGURE A5.4 CUMULATIVE FLUXES OF CO_2 FOR EACH FERTILISER TREATMENT (IF = INORGANIC FERTILISER, UPS = PIG SLURRY, TPS = TREATED PIG SLURRY). EACH DATA POINT REPRESENTS THE MEAN OF THREE CHAMBERS USED PER TREATMENT. VERTICAL DASHED LINES REPRESENT THE FOUR FERTILISER APPLICATIONS. ERROR BARS HAVE BEEN REMOVED TO AID VISUALISATION.

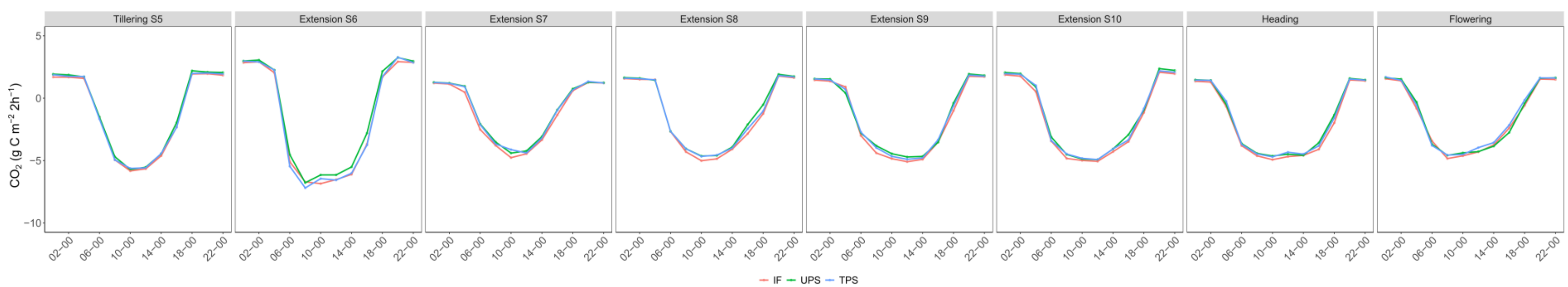


FIGURE A5.5 MEAN 2-HOUR FLUXES OF CO_2 FOR EACH FERTILISER TREATMENT (IF = INORGANIC FERTILISER, UPS = UNTREATED PIG SLURRY, TPS = TREATED PIG SLURRY) FOR EACH WINTER WHEAT GROWTH STAGE OVER THE MEASUREMENT PERIOD. EACH DATA POINT REPRESENTS THE MEAN OF THREE CHAMBERS USED PER TREATMENT. ERROR BARS HAVE BEEN REMOVED TO AID VISUALISATION. THE DATES OF EACH GROWTH STAGE, AND THE AVERAGE DAILY AIR TEMPERATURE, TOTAL RAINFALL, AND PROPORTION OF THE DATA THAT WERE GAP-FILLED PER GROWTH STAGE IS SHOWN IN TABLE A5.5.

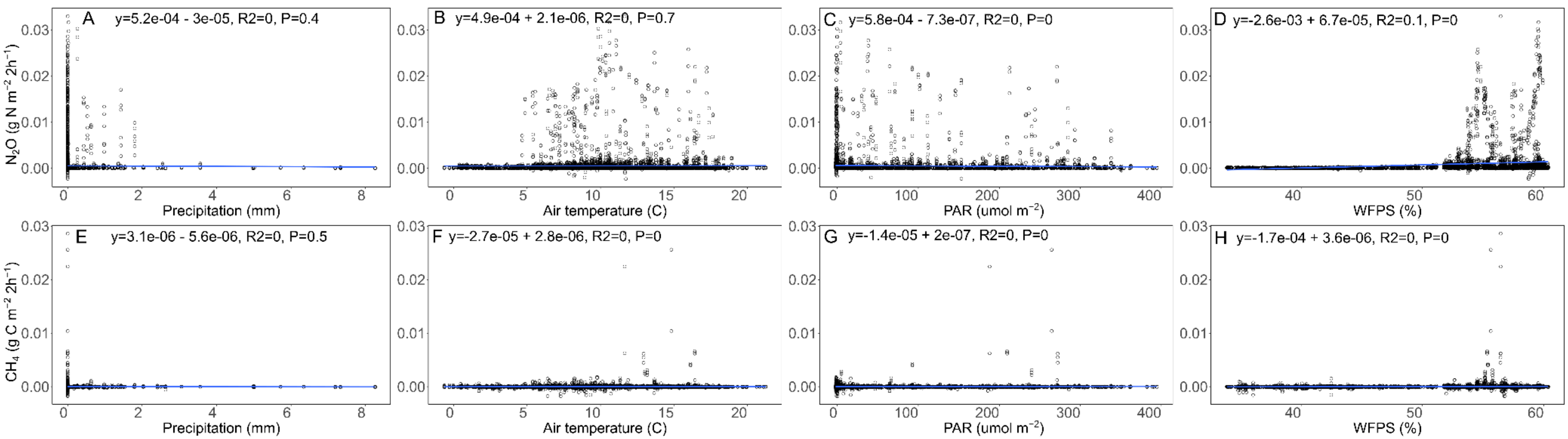


FIGURE A5.6 LINEAR REGRESSION SHOWING RELATIONSHIPS BETWEEN ALL N_2O (G N M^{-2} $2HR^{-1}$) AND CH_4 (G C M^{-2} $2HR^{-1}$) FLUXES AND (A, E) PRECIPITATION, (B, F) AIR TEMPERATURE, (C, G) PAR AND (D, H) WFPS. PAR = PHOTOSYNTHETICALLY ACTIVE RADIATION AND WFPS = WATER-FILLED PORE SPACE.

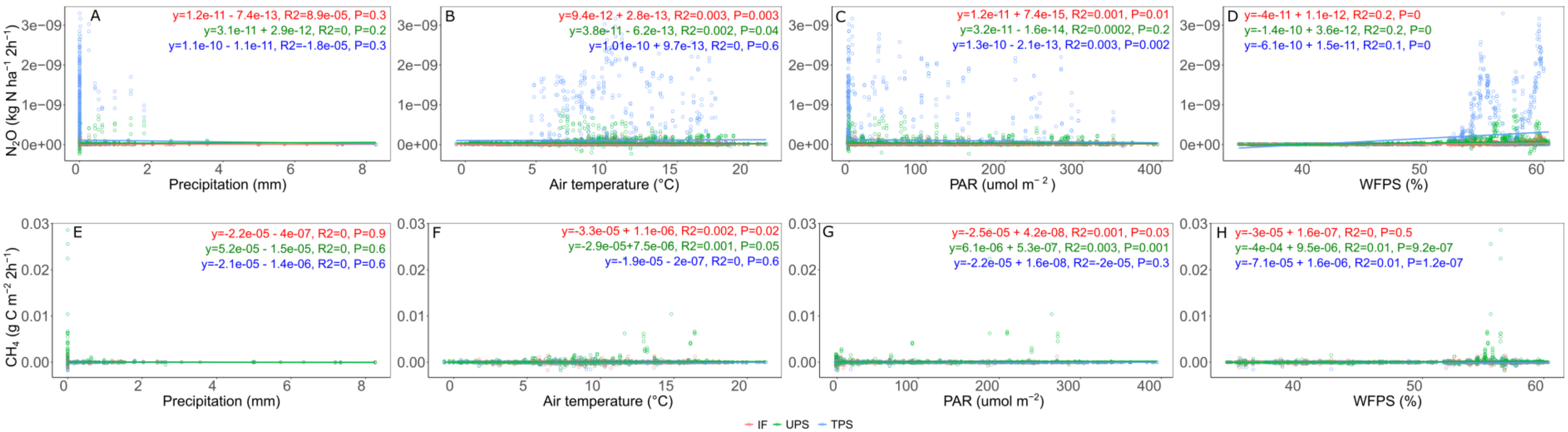


FIGURE A5.7 LINEAR REGRESSION SHOWING RELATIONSHIPS BETWEEN N_2O (G N M^{-2} 2HR^{-1}) AND CH_4 (G C M^{-2} 2HR^{-1}) FLUXES FOR EACH FERTILISER TREATMENT AND (A, E) PRECIPITATION, (B, F) AIR TEMPERATURE, (C, G) PAR AND (D, H) WFPS. PAR = PHOTOSYNTHETICALLY ACTIVE RADIATION AND WFPS = WATER-FILLED PORE SPACE. RED INDICATES THE INORGANIC FERTILISER (IF) TREATMENT, GREEN THE UNTREATED PIG SLURRY (UPS) TREATMENT AND BLUE THE TREATED PIG SLURRY (TPS) TREATMENT.

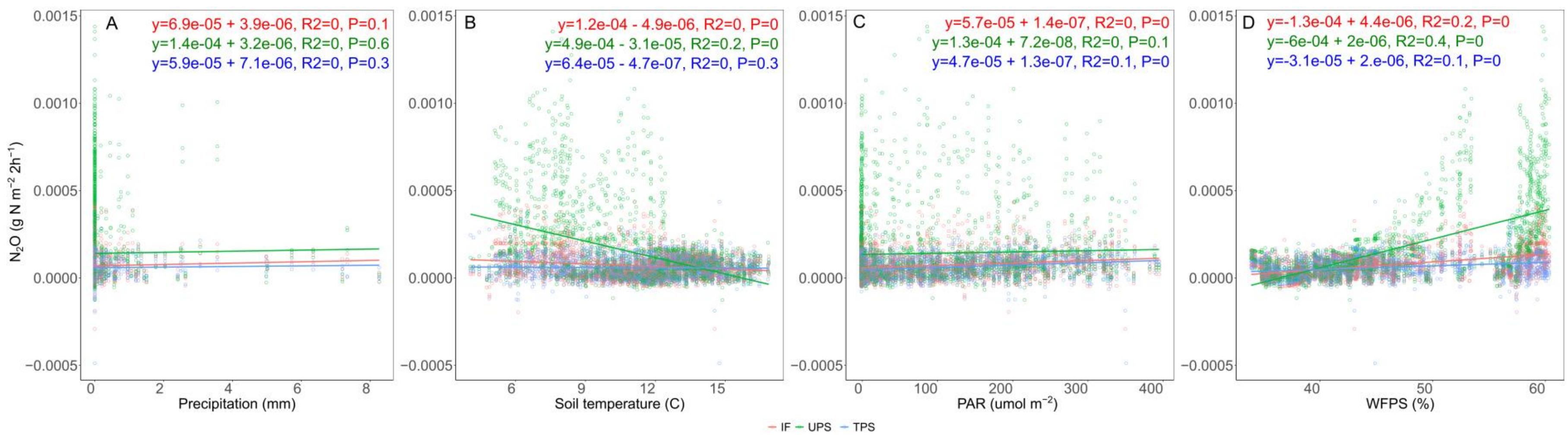


FIGURE A5.8 LINEAR REGRESSION SHOWING RELATIONSHIPS BETWEEN N_2O FLUXES (EXCLUDING FLUXES RECORDED WITHIN 0-7 DAYS OF THE FIRST TWO FERTILISER APPLICATIONS) FOR EACH FERTILISER TREATMENT (G N M^{-2} 2HR^{-1}) AND (A) PRECIPITATION, (B) AIR TEMPERATURE, (C) PAR AND (D) WFPS. PAR = PHOTOSYNTHETICALLY ACTIVE RADIATION AND WFPS = WATER-FILLED PORE SPACE. RED INDICATES THE INORGANIC FERTILISER (IF) TREATMENT, GREEN THE UNTREATED PIG SLURRY (UPS) TREATMENT AND BLUE THE TREATED PIG SLURRY (TPS) TREATMENT.