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Drained Agricultural Peatlands as Persistent Carbon Sources: Implications for Carbon and Water Use Intensity in Food Production

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ABSTRACT

Peatlands have the capacity to sequester large quantities of carbon and can therefore play an important role in climate change mitigation. However, anthropogenic activities alter their hydrological regimes, converting them from net CO_2 sinks into net sources. In England and elsewhere, lowland peatlands have been heavily drained and modified for agricultural land use, resulting in some of the most productive farmland in the UK. Estimates of CO_2 emissions and water use from the area are scarce, but these data are required to understand the consequences of maintaining agricultural output whilst simultaneously reducing GHG emissions. In this paper, we compiled a uniquely comprehensive dataset of CO_2 and H_2O flux measurements from flux towers on cropped lowland peat, and coupled this with crop calorific values to estimate carbon and water use intensities of food production on peat. Our results showed that croplands on peat emitted 23.1 ± 10.4 ton CO_2 ha^{-1} y^{-1} (mean $\pm$ SD). Sites with peat depth > 40 cm emitted 25.1 ± 9.2 ton CO_2 ha^{-1}, while wasted peat sites emitted 11.8 ± 4.8 ton CO_2 ha^{-1}. Effective water table depth and organic carbon content were the main drivers of variation in annual net ecosystem production and ecosystem respiration across sites; crop type partly followed these gradients, so may not be a direct driver of variations in emissions. ET was less variable across site-years and depended on the phenology of crop production. When considering CO_2 emissions and water use per calorie produced, lettuce and celery rotations were the most C and water use intensive crops, with values an order of magnitude larger than cereal crops. Overall, this paper highlights the scale of CO_2 emissions from managed peatlands and the importance of balancing GHG emissions and water use with ongoing food production from these economically important areas.

1 | Introduction

In their natural state, peatlands have the capacity to capture and store vast quantities of atmospheric carbon (C) (Gorham 1991; Yu et al. 2011). Despite occupying only 3% of the global land area, it is estimated that approximately one third (250–650 Pg of C) of the terrestrial C pool is stored as peat (Dargie et al. 2017; Gorham et al. 2012; Mander et al. 2025; Page and Baird 2016). The waterlogged conditions that natural peatlands experience play a defining role in the ability of these ecosystems to function as long-term C sinks. The net C accumulation is the result of a higher C assimilation via photosynthesis, compared to C losses via autotrophic and heterotrophic respiration, net methane (CH_4) emissions, and aquatic fluxes of particulate and dissolved organic C (POC and DOC, respectively) (Clymo et al. 1998; D'Acunha et al. 2019; Helbig et al. 2019). The rate of C accumulation in natural or semi-natural northern peatlands varies, with previous studies

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reporting C sequestration values that range from 0.2 to 4 ton CO_2 ha^{-1} y^{-1}, noting that dry years (i.e., when the water table was low) usually caused a reduction in the C sink strength or caused the site to become a C source because the peat was exposed to oxygen and decomposition rates increased (Artz et al. 2022; Evans et al. 2021; Peacock et al. 2019; Yu et al. 2011; Yu 2012).

Anthropogenic activities such as drainage for peat extraction, grazing, and agriculture can turn peatland ecosystems into greenhouse gas (GHG) sources (e.g., Peacock et al. 2019; Rankin et al. 2018). Drainage exposes the organic layer to aerobic conditions and promotes decomposition, releasing large quantities of C to the atmosphere as carbon dioxide (CO_2) and contributing to land subsidence. Likewise, peatland conversion to agriculture (i.e., horticulture, arable and intensive grassland) can increase nitrification, enhancing nitrous oxide (N_2O) emissions. This effect can be further promoted by the use of nitrogen-rich fertilizers and manure for crop growth (Alm et al. 1999; Haddaway et al. 2014; Martikainen et al. 1993). On the other hand, drainage can decrease or reverse natural methane emissions, as decreasing the water table depth reduces the anoxic zone (where CH_4 is produced) and increases the aerobic layer (where CH_4 is oxidized) (e.g., Alm et al. 1999; Holl et al. 2020). However, previous research has shown that the ditches created for peatland drainage can behave as large CH_4 sources to the atmosphere (Hyvönen et al. 2013; Peacock et al. 2017, 2021; Teh et al. 2011). In addition, ditches facilitate indirect CO_2 emissions, as they provide a pathway for degassing of CO_2, and for the export of POC and DOC that can be oxidized outside the peatland area.

In England, approximately 1.5% (6.5 Mt CO_{2eq}) of terrestrial GHG emissions come from drained agricultural lowland peat (Caudwell 2023) located across five main regions; the East Anglian Fenland, the Norfolk Broads, the Somerset Levels, the Humberhead Levels and Northwest England (Lloyd et al. 2023). In particular, the East Anglian Fenland ('the Fens') is the largest contiguous area of lowland fen in the UK and has been progressively modified since the 1600s, initially for grazing followed by increasingly intensive arable and horticultural crop production (Hutchinson 1980). The Fens are one of the most agriculturally fertile and profitable regions in the UK, producing more than 30% of the country's fresh produce and 21% of England's intensive crops (NFU 2019). However, the area is also one of the largest sources of GHG emissions, both historically and currently (Mitchell et al. 2024). It is estimated that the Fens originally occupied 150,000 ha, with a peat depth of 5 m or more, of which only 16,500 ha (~11%) of peat > 40 cm deep remain (Holman 2009). The rest of the area is 'wasted peat', where the original peat has been reduced to a thin but highly compacted layer of (< 40 cm) mixed organic and mineral soil.

Likewise, the Humberhead Levels contain about 6300 ha of peat with a long history of drainage that dates back to the 1600s. Currently, the area is characterized by intensive, large-scale farming, with arable farming accounting for 90% of the total farmed area (Morris et al. 2010). The region produces potatoes, field vegetables and salad crops, as well as cereals, which are used as break crops to facilitate the rotation of the other crops (Morris et al. 2010).

Estimates of the C balance of lowland agricultural peatlands remain scarce. These data are required to underpin conversations and policy around food production in the region whilst simultaneously reducing GHG emissions. Both are required to maintain domestic food security and the UK ambition to achieve net zero GHG emissions by 2050 or earlier. At the same time, having accurate estimates of the water budget is important for water resource management in one of the driest regions of the UK, as many crops grown in the Fens are increasingly dependent on irrigation. This is particularly important considering rising temperatures and atmospheric dryness due to climate change will further increase the need for crop irrigation in the region, which already has one of the highest water demands in England (Knox et al. 2020) and is under severe water stress (Jenkins et al. 2024).

This research aims to increase our understanding of the C and water (H_2O) budgets of different agroecosystems in lowland peat areas and to highlight opportunities for land management focused on reducing GHG emissions by estimating the C and water emissions, as well as C and water intensities associated to agricultural production in peat. For that, we estimated the net ecosystem production (NEP), evapotranspiration (ET) and ecosystem water use efficiency (eWUE) of eight croplands in England. In agroecosystems, the NEP accounts for the vertical C fluxes (i.e., CO_2 exchange between the surface and the atmosphere), as well as any lateral C inputs (e.g., organic fertilizers) and outputs (e.g., biomass harvest). Aquatic C (CO_2 and CH_4) exports and vertical CH_4 fluxes were not measured, therefore NEP does not represent the full net ecosystem carbon balance (NECB). ET is a major component of the water cycle and has often been used to estimate crop water requirements and facilitate land management (Allen et al. 2011; Fisher et al. 2017; Wanniarachchi and Sarukkalige 2022).

In addition to these parameters, we calculated the C and water intensities of each crop (i.e., estimates of the CO_2 emissions and water use per kilocalorie of food produced). These metrics can be used to illustrate the inefficiencies in the food systems at local and global scales as they provide information on C and water emissions per kilocalorie of crop (Carlson et al. 2017; Cassidy et al. 2013; MacDonald et al. 2015). For example, Cassidy et al. (2013) demonstrated global calorie availability could be increased by ~70% by shifting crops away from animal feed and biofuels to crops used for direct human consumption. These metrics can also help in shaping policies for achieving food security and reducing environmental impact (Cassidy et al. 2013; DeFries et al. 2015; MacDonald et al. 2015; West et al. 2014).

2 | Methods

2.1 | Study Area

We used data from a network of eight eddy covariance (EC) flux towers located on agricultural peatlands (Fluxnet site codes:

TABLE 1 | Information about the study sites.

Site ID	Latitude	Longitude	Year	Crop	Peat depth (m)	OC (%)	Bulk density (g cm^{-3})
UK-Rdm	52.44	0.42	2017	Lettuce	0.63 ± 0.20	33.0 ± 3.6	0.40 ± 0.1
			2017	Lettuce			
			2018	Lettuce[a]			
			2019	Maize			
			2020	Lettuce			
			2020	Celery			
			2021	Maize[a]			
			2022	Lettuce			
			2022	Celery			
			2023	Winter wheat			
UK-Stm2	52.32	0.23	2023[c]	Sugar beet	0.76 ± 0.22	31.5 ± 2.3	0.39 ± 0.07
UK-Swt	52.44	−0.26	2021	Peas	0.36 ± 0.42	13.5 ± 1.8	0.78 ± 0.13
			2022	Winter wheat			
			2023	Spring beans/oats			
UK-Stm	52.33	0.22	2021	Potatoes	0.68 ± 0.12	17.9 ± 2.1	0.80 ± 0.14
			2022	Winter wheat			
			2023	Spring beans			
UK-Spy	52.46	−0.18	2023[c]	Winter wheat	0.98 ± 0.33	31.8 ± 0.8	0.38 ± 11
UK-Pob	53.43	−0.92	2021	Spring wheat	0.24[b]	11.6 ± 2.9	1.01 ± 0.27
UK-Po1	53.46	−0.91	2023	Spring wheat	1.75 ± 0.60	30.0 ± 1.0	0.46 ± 0.08
UK-Po2	53.46	−0.91	2023	Spring wheat	2.04 ± 0.74	26.1 ± 0.7	0.52 ± 0.07

[a]At UK-Rdm, the tower was moved to different fields inside the farm in 2018 and 2021, so the lettuce and maize years are incomplete. The lettuce in 2018 was not included in the C budget analysis, but we considered the 2021 maize as the tower was moved after crop harvesting to a field with similar conditions.
[b]Peat depth at UK-Pob was obtained from Evans et al. (2023). OC = Organic carbon content (%).
[c]At UK-Spy and UK-Stm2, we considered 1 year of measurements from 2023-04-01 to 2024-03-31, and 2023-02-01 to 2024-01-31, respectively. More information on the instrumentation and measurements is available in Table S1.

UK-Rdm, UK-Swt, UK-Stm, UK-Stm2, UK-Spy, UK-Pob, UK-Po1, UK-Po2) (Table 1 and Figure 1). These towers are part of the UK-wide network of EC towers (UK-Flux) led and operated by the UK Centre for Ecology & Hydrology.

2.2 | EC and Ancillary Data

Each study site was equipped to measure fluxes of net ecosystem CO_2 exchange (NEE) and energy (momentum (T), latent (LE) and sensible heat (H)) using the micrometeorological eddy co-variance (EC) technique. Details of the EC instrumentation are summarised in Table S1. To complement EC measurements, a set of environmental variables was also measured at each site. Net radiation (NETRAD, W m^{-2}) including incoming shortwave radiation (SW_IN, W m^{-2}) was measured using a 4-channel SN500 radiometer (Apogee Instruments Ltd., Logan, UT, USA) or an NR01 4-component net radiation sensor (Hukseflux, Delft, The Netherlands) in the case of UK-Rdm before 2019. Soil heat flux (G, W m^{-2}) was monitored using two HFP01-SC self-calibrating heat flux plates (Hukseflux, Delft, The Netherlands) installed at a depth of 0.03 m. Photosynthetically active radiation (PAR, μmol photons m^{-2} s^{-1}) was measured using an SKP01 Quantum sensor (Skye Instruments, Llandrindod Wells, UK) or a CS310 Quantum sensor (Apogee Instruments Ltd., Logan, UT, USA). Air temperature (TA, °C), air pressure (PA, kPa) and relative humidity (RH, %) were measured using an HMP155A

(Vaisala, Helsinki, Finland) at a height of 2 m. A profile of soil temperatures (TS, °C) was measured using both Acclima digital TDTs (Acclima Inc., Idaho, USA), installed at −5, −10, −15, and −25 cm, or SoilVUE (Campbell Scientific Ltd., Logan, Utah, USA), with measurements at −5, −10, −20, −30, −40, and −50 cm. Water table depth (WTD, m) below the ground surface was measured using a CS451 pressure transducer (Campbell Scientific Ltd., Logan, Utah, USA) or a PDCR 1800 Series (Druck, UK) at UK-Rdm before 2019. WTD measurements at UK-Swt started on 2021-02-19, and at UK-Rdm they started on 2017-03-31. Precipitation (P, mm) was measured using an SBS500 tipping-bucket rain gauge with an aperture height of 1 m (Environmental Measurements Ltd. Newcastle, UK).

2.3 | Flux Calculations

We used EddyPro Version 7.0.6 (Fratini and Mauder 2014) to calculate fluxes from raw EC data measured at 20 Hz. Fluxes were computed as block averages using 30-min averaging periods. Standard corrections were applied (Sabbatini et al. 2018) such as despiking, angle of attack correction for Gill anemometers (Nakai et al. 2006); double coordinate rotations to align the sonic anemometer data to the local terrain, corrections for high- and low-frequency attenuation of the signal (Moncrieff et al. 1997, 2004); correction for CO_2 and H_2O concentrations in open path analysers due to fluctuations in atmospheric temperature and
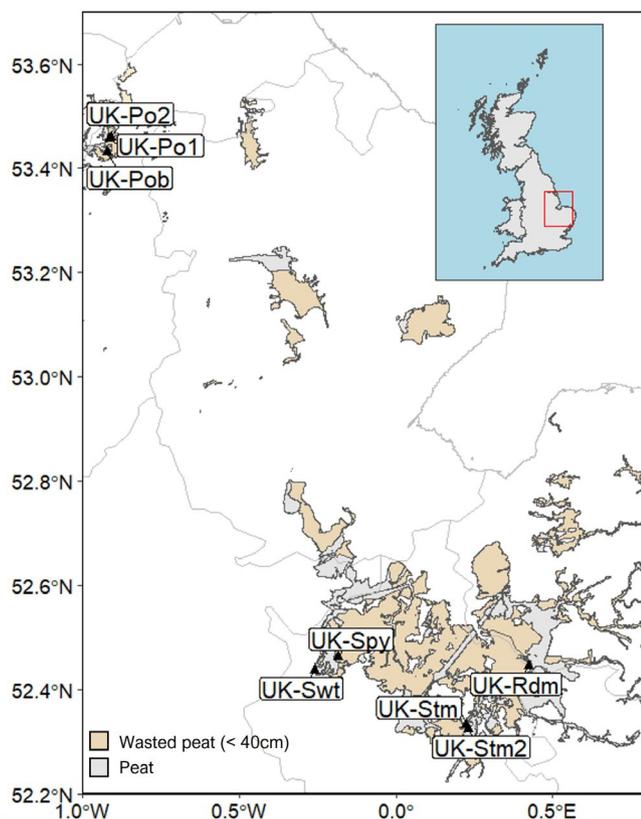
FIGURE 1 | Location of the study area in the UK and local distribution of flux monitoring sites. The base map shows the distribution of deep and wasted peat areas. Map lines delineate study areas and do not necessarily depict accepted national boundaries (Natural England 2010).

humidity (Webb et al. 1980); and time-lag compensation using covariance maximisation (Aubinet et al. 2012).

2.4 | Quality Control

Quality control of EC data involved the removal of statistical outliers and tests to ensure that the theoretical requirements for the successful application of the EC technique were met. Fluxes were removed when they fell outside of a range of values (i.e., LE [−100, 800] W m^{-2}, H [−100, 600] W m^{-2}, NEE [−50, 50] μmol m^{-2}s^{-1}). Statistical outliers were identified and excluded using the median absolute deviation approach and the recommendations provided by Papale et al. (2006). In addition, to ensure that fluxes were representative of the ecosystem, periods of low turbulence were also filtered using site-specific friction velocity (u$_*$) thresholds (Papale et al. 2006; Pastorello et al. 2020). Data availability after filtering ranged from 34.5% (UK-Stm in 2021) to 72.2% (UK-Spy), with more gaps observed at nighttime (42% available data, on average) compared to daytime (68% available data, on average) due to less turbulent conditions. More information on data availability is presented on Table S2.

2.5 | EC Gap-Filling and NEE Partitioning

We used data from nearby EC stations to gap-fill environmental variables (WS, PA, TA, RH, SW_IN, PPFD_IN, VPD, P). When

data from nearby stations were not available, we used the MDS approach, which uses the mean value observed during similar meteorological conditions, to gap-fill the data. To gap-fill fluxes at each site, we used a random forest ensemble of 500 trees and a set of variables (WS, PPFD, PA, VPD, TA, RH, WTD, SW_IN, TS (5 and 10 cm depth), SWC (5 and 10 cm depth), together with sine and cosine transformations of DOY to represent the seasonal cycle in fluxes). The random forest model was preferred over other gap-filling approaches such as the marginal distribution sampling (i.e., MDS), as studies suggest that it performs better for gap-filling longer gaps (> 1 month) compared to MDS (Irvin et al. 2021; Kim et al. 2020; Winck et al. 2023; Zhu et al. 2022).

Partitioning of NEE into estimates of gross primary production (GPP) and total ecosystem respiration (R$_{eco}$) was performed using the nighttime based partitioning method detailed in Reichstein et al. (2005). Gap-filling and flux partitioning were performed using the REddyProc Package (Wutzler et al. 2018) and the caret package (Kuhn 2008) for the R Language for Statistical Computing R version 4.5.1, (R Core Team 2025).

2.6 | Estimation of the Net Ecosystem Productivity (NEP)

The net ecosystem productivity (NEP, ton CO$_2$ ha^{-1}) was estimated as follows:

$$NEP = NEE + C_{export} - C_{import} \tag{1}$$

where NEE is the net ecosystem exchange (ton CO$_2$ ha^{-1}) estimated using eddy covariance, C$_{export}$ (ton CO$_2$ ha^{-1}) is the CO$_2$ exported from the field via crop and any residue (e.g., straw bales) harvesting, and C$_{import}$ (ton CO$_2$ ha^{-1}) is the C added to the field as fertilizer, seeds, or peat plugs (i.e., blocks of peat used to grow Celery and Lettuce crops in a greenhouse before planting into the field). The C content of each fertilizer was determined according to stoichiometric ratios or values from the literature. The C content of seeds and peat plugs was assumed to be 50% (Cumming 2018). Management information for each farm is presented in Table S3.

In this study, a positive NEP indicates C loss to the atmosphere, while a negative NEP indicates that the ecosystem is a C sink.

For the C$_{export}$, we used a combination of farm yield values obtained from land managers or biomass samples from the field when the yield from land managers was not available (Table S3). Biomass samples were manually collected from the fields within 1–2 weeks before or after the time of harvest. Five 25 × 25 cm quadrats were collected at each site by cutting to 15–20 cm above the surface, to match business as usual farm operating procedures. Samples were weighed and oven dried at 80°C to determine fresh and dry biomass weights (ton DM ha^{-1} y^{-1}). We obtained the C$_{export}$ in ton CO$_2$ ha^{-1} y^{-1} by multiplying the yield by the %C content. We assumed a %C content of 39 ± 4 for lettuce, celery, peas, beans, sugar beet and potatoes (Cumming 2018; Jia et al. 2012; Koerber et al. 2009; Terry et al. 1972), and a 45 ± 5 %C content for wheat and maize (Ma et al. 2018). When only grain values were reported as the yield for wheat, we used the crop harvest index obtained from the literature (0.51) to estimate the

total aboveground biomass removed from the field (AHDB 2025; BASF et al. 2023).

2.7 | Uncertainty Estimation

The random uncertainty of fluxes due to sampling errors (i.e., the limited number of independent samples that contribute to each sampling period) was estimated by EddyPro for each 30-min flux average using the method of Finkelstein and Sims (2001). Errors associated with flux gap-filling were estimated by obtaining the standard error of the model ensemble (i.e., the 500 trees generated for the random forest model used for gap-filling). Errors associated to the u_* threshold selection were estimated by generating 200 artificial datasets that varied in gap size, each with a u_* threshold estimate. Then the lower (5%), median (50%) and upper (95%) estimates of u_* were determined, and NEE, GPP and Reco were obtained for each u_* threshold (D'Acunha et al. 2019; Wutzler et al. 2018). The total uncertainty for each 30-min value was estimated as the combined random uncertainty and the uncertainty due to the unknown u_* threshold:

$$\sigma_{NEE} = \sqrt{\sum \sigma_R{}^2 + \sigma_{GF}{}^2 + \sum \sigma_{u*}{}^2} \qquad (2)$$

where σ_{NEE} is the annual standard deviation for NEE, σ_R is the random uncertainty from measurements, σ_{GF} is the error estimated by random forest, and σ_{u*} is the standard deviation obtained from the lower, median, and upper u_* thresholds bootstrap estimates.

The 30-min uncertainty was then propagated to the annual scale for each site and year.

The uncertainty in C_{export} was obtained by aggregating the uncertainty from the biomass sample (when available), and the uncertainty reported for the %C content of crop biomass. The uncertainty in C_{import} was derived from the C content of the peat plugs, assumed to be $50\% \pm 5\%$ (mean $\pm$ SD) (Cumming 2018).

The uncertainty in NEP for each site-year was obtained by aggregating the estimated uncertainty for NEE and the uncertainties of C_{export} and C_{import}.

2.8 | Estimation of the Ecosystem Water Use Efficiency (WUE) and Crop Water Productivity (CWP)

The eWUE is defined as the rate of C uptake per unit of water used by the crop, and is a parameter widely used to study the link between the C and water cycles, which is crucial for better crop water management (Chen et al. 2023; Tang et al. 2014). Salad crops (lettuce, celery) and potatoes are usually irrigated, which leads to higher ET values and also adds financial, energy and water costs. In this study, we estimated eWUE (in g C kg^{-1} H$_2$O) using the gross primary productivity and the evapotranspiration, as shown in Equation (3).

$$eWUE = \frac{GPP}{ET} \qquad (3)$$

where GPP is the gross primary productivity in g C m^{-2} and ET is the evapotranspiration in kg H$_2$O m^{-2}.

The crop water productivity (CWP, g kg^{-1} H$_2$O) is a measurement that relates the amount of crop produced and the water used to produce the crop, sometimes referred to as 'crop per drop'. We estimated the CWP as:

$$CWP = \frac{Crop\ yield}{ET} \qquad (4)$$

where crop yield is the biomass that is exported from the field upon crop harvesting, in g of dry weight m^{-2}, and ET is the evapotranspiration in kg H$_2$O m^{-2}.

2.9 | Estimation of C and Water Intensities

In this study we also estimated the amount of C that is emitted as CO_2 when producing 1 kcal (kcal) of a crop ($C_{intensity}$, g CO_2 kcal^{-1}), and the amount of water used when producing 1 kcal of the crop (Water$_{intensity}$, kg H$_2$O kcal^{-1}). First, we used FAO food balance sheets (FAOSTAT 2024) and the Agriculture in the United Kingdom 2024 dataset (DEFRA 2025) to obtain the percent of total crop yield allocated to direct human food consumption for each crop. Then, we obtained crop calorie data (kcal per dry gram) from Cassidy et al. (2013) and multiplied this value by the dry yield to obtain the total kilocalories available for that crop. Finally, we applied Equations (5) and (6) to obtain the C and water intensities for each crop.

$$C_{intensity} = \frac{NEP}{kcal} \qquad (5)$$

$$Water_{intensity} = \frac{ET}{kcal} \qquad (6)$$

2.10 | Soil Properties

Effective peat depth was obtained from the literature for UK-Pob (Evans et al. 2023). For the other sites, effective peat depth was measured in a square grid of 20 × 20 m centred around the flux tower, with measurements taken every 5–10 m using an extendable peat probe (Van Walt Ltd., UK). To obtain soil organic carbon (SOC, %) and dry bulk density (BD, g cm^{-3}), we collected three 40 cm intact soil cores from within the flux tower footprint at each site. The cores were cut into 10 cm sections to determine changes in bulk density and SOC with depth, and oven dried at 80°C. The SOC was determined by loss on ignition (LOI) at 550°C for 4 h (Parry and Charman 2013). LOI was then converted to C using the regression in Equation (7), developed for UK moorland soils (Bol et al. 1999; Garnett et al. 2001; Parry and Charman 2013).

$$\%C = 0.526 \times LOI - 0.167 \qquad (7)$$

2.11 | Environmental Predictors of Fluxes

We assessed the relationship between C (NEP, Reco, GPP) and water fluxes (ET) aggregated by site (i.e., mean CO_2 emissions per site), and climatic, hydrological and soil quality variables, including the WTD, effective peat depth, the peat depth exposed

to aerobic conditions (WTD$_e$), the peat C content (% OC), bulk density, temperature, precipitation and the bulk C (i.e., g C per cm^3 of soil), using a stepwise linear regression and testing both forward and backward variable selection. The best model was selected using the Akaike information criterion (AIC), which assesses models by balancing goodness-of-fit, and the number of variables included. Smaller AIC values indicate better model fitting. Final model performance was assessed using the 'performance' package for R (version 0.15.1), which provides a check of model assumptions such as linearity, normality of residuals, independence, homoscedasticity and no multicollinearity (Lüdecke et al. 2021). All statistical analyses were performed in R version 4.5.1.

3 | Results

3.1 | Site Climatic Conditions and Hydrology

All sites experienced similar mean annual air temperatures, ranging from 10°C to 12°C (Table 2). UK-Spy had the highest mean annual temperature in 2023 (11.6°C), and UK-Swt had the lowest (10.4°C) in 2021. Overall, annual temperatures were higher than the 30-year climatic normal for 1961–1990 (Table S1), particularly during the summers of 2018 and 2022, when the UK experienced record-breaking heatwaves. Annual

TABLE 2 | Total annual precipitation (P, mm), mean annual water table depth (WTD, m), mean annual air temperature (TA, °C), and mean annual vapor pressure deficit (VPD, hPa) during the measurement period for each site.

Site	Year	P (mm)	TA (°C)	VPD (hPa)	WTD (m)
UK-Rdm	2017	666	10.9	3.1	−1.15
UK-Rdm	2019	607	10.6	2.8	−1.39
UK-Rdm	2020	695	11.1	2.4	−1.17
UK-Rdm	2021	540	9.9	3.4	−1.20
UK-Rdm	2022	542	11.2	3.9	−1.11
UK-Rdm	2023	706	10.9	3.1	−1.20
UK-Stm	2021	535	10.7	3.2	−1.14
UK-Stm	2022	467	11.4	4.1	−1.21
UK-Stm	2023	877	11.2	3.4	−1.21
UK-Stm2	2023	744	11.5	3.2	−0.81
UK-Swt	2021	501	10.4	3.1	−0.89
UK-Swt	2022	523	11.4	4.0	−1.25
UK-Swt	2023	701	11.2	3.3	−0.83
UK-Po1	2023	764	10.9	2.9	−0.88
UK-Po2	2023	729	11.1	3.0	−0.70
UK-Pob	2021	596	10.4	3.1	−1.81
UK-Spy	2023	876	11.2	3.1	−0.31

means of vapor pressure deficit (VPD, hPa) for the sites ranged between 2.5 and 4.1 hPa, with the highest value observed at UK-Swt and UK-Stm in 2022, and the lowest at UK-Rdm in 2020. Overall, UK-Pob received the least amount of precipitation, with an annual p value of 507 mm, followed by UK-Swt with a mean p value of 575 mm. The value for UK-Pob is below the 30-year annual means (1961–1990) for the region (539 mm) (Robinson et al. 2015). All the other sites, on average, received more than 500 mm of precipitation (Table 2). In terms of water table, all the sites were actively managed to maintain a water table that is suitable for crop production. UK-Spy had the highest average water table (mean WTD = −0.32 m) during the measurement period, whereas UK-Pob had the lowest WTD in 2021, with an average of −1.81 m (Table 2, Figure 2).

3.2 | Field Scale Fluxes

3.2.1 | Net Ecosystem Exchange (NEE)

NEE provides a measure of direct CO_2 exchange between the field and the atmosphere, without adjusting for lateral C inputs and outputs (including crop harvest). Followed a similar seasonal pattern across sites (Figure 3), with most of the CO_2 uptake occurring during spring and summer (which coincided with the crop growing season), and CO_2 emissions during periods without crops and during senescence of cereals. UK-Rdm showed a bimodal pattern in years when the site was double cropped with horticultural crops (i.e., two crops planted in the same growing season), with a lettuce/lettuce cycle in 2017, and a lettuce/celery rotation in 2020 and 2022. As a result of the double-cropping system, fluxes at UK-Rdm showed two peaks in C uptake at the period of maximum crop growth instead of one, with a period of high R$_{eco}$ during the intercropping period under the warm summer conditions (Table 3).

The magnitude of GPP, R$_{eco}$ and NEE varied across sites and crops (Figures 3 and 4). At UK-Stm, the winter wheat in 2022 had higher GPP than the potato in 2021 and the beans in 2023, while R$_{eco}$ was similar for the three crops. This resulted in a negative NEE for the winter wheat year compared to positive NEE in the other crop-years. At UK-Rdm, winter wheat had the highest GPP, R$_{eco}$ and lowest (but still positive) NEE compared to the other crop-years. Short term negative NEE (i.e., GPP > R$_{eco}$) was observed for the wheat crop between April and October 2023, whereas the lettuce and celery crops only had about a month were GPP was higher than R$_{eco}$. UK-Swt also exhibited a variability in GPP, R$_{eco}$ and NEE between years and crops. For the peas 2021, R$_{eco}$ was higher than GPP at the beginning of the growing season and until June, when the site turned into a weak net sink for about a month and then started emitting C again during autumn and winter. Winter wheat 2022 had a larger GPP and R$_{eco}$ than the peas, and negative annual NEE, with a similar seasonal pattern to that observed with winter wheat at UK-Stm in the same year. Finally, during the beans and oats (2023), the site acted as a net C source during the spring and autumn periods, and as a net C sink between June and September. At UK-Stm2, the site was a C source for the first half of 2023 while vegetation was absent. Starting July, GPP started to become larger than R$_{eco}$ and the site turned into a C sink for the rest of the year once a dense beet canopy had developed.
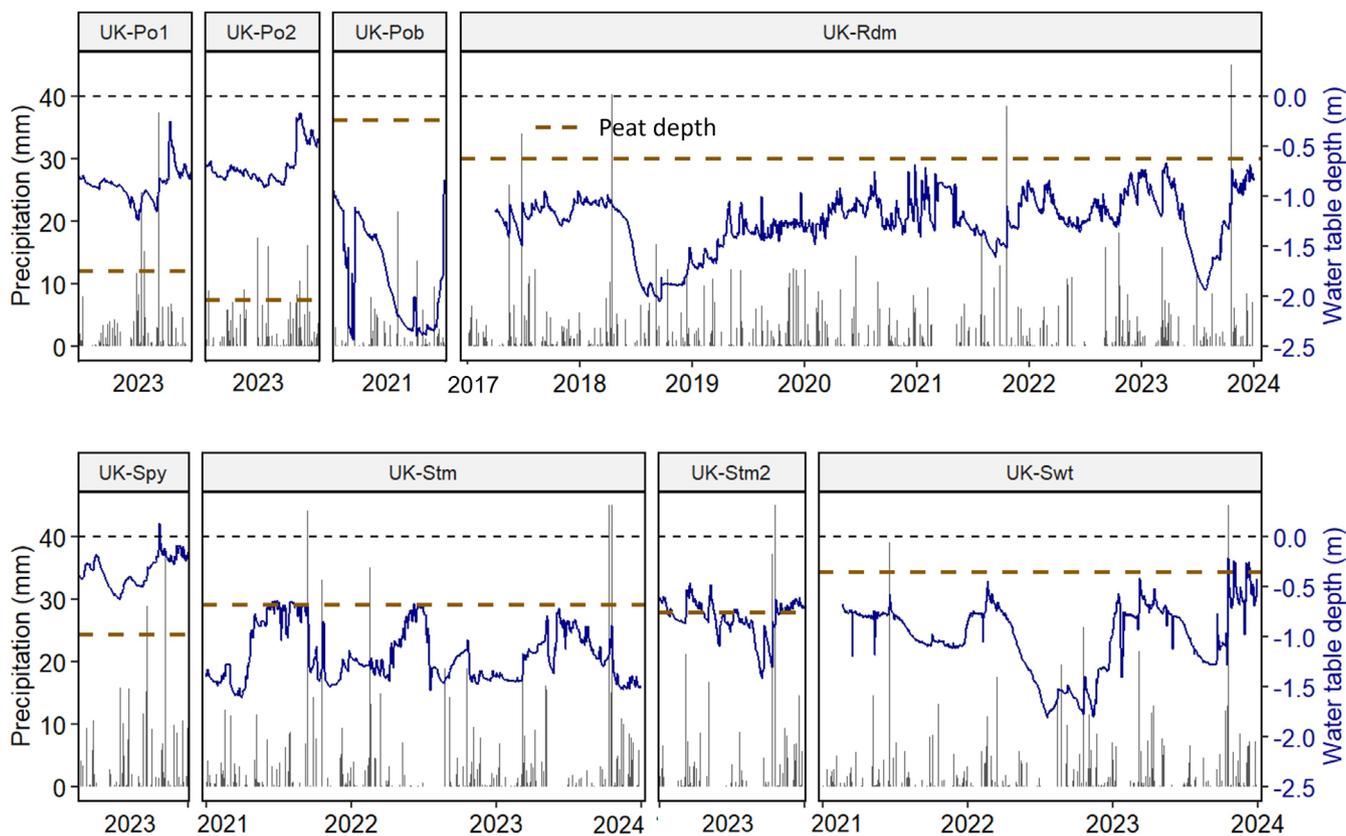
FIGURE 2 | Precipitation (P, mm), water table depth (m, blue line) and peat depth (m, brown dashed line) for each site during the measurement period. Water table depth measurements at UK-Swt started on 2021-02-19, and at UK-Rdm they started on 2017-03-31.

In general, annual R_{eco} exceeded annual GPP for most site-years, which resulted in a positive NEE (i.e., direct net CO_2 emissions from the field) (Figure 3). However, the in situ NEE was negative at UK-Swt in 2023 (-18.3 ± 1.5 ton CO_2 ha^{-1} y^{-1}), at UK-Pob in 2021 (-4.4 ± 0.9 ton CO_2 ha^{-1} y^{-1}), and at UK-Stm2 in 2023 (-14.9 ± 1.4 ton CO_2 ha^{-1} y^{-1}). This pattern reflected the high productivity of the crops during these years, during winter wheat production, as well as during a sugar beet cycle at UK-Stm2 (Figures 3 and 4).

3.2.2 | Net Ecosystem Productivity (NEP)

In terms of NEP, which considers both the import of C via fertilizer addition and/or peat plugs and CO_2 export from harvesting activities, all the sites were strong net C sources in all years. At UK-Swt, UK-Pob, and UK-Stm2 where annual NEE had been negative, the high CO_2 export in harvested biomass completely offset the in situ NEE and turned the systems into a C source. At the other farms, off-site CO_2 emissions from harvested biomass represented between 28% and 95% of the total emissions.

The highest CO_2 emissions were observed at UK-Stm2 in 2023 (sugar beet), with 38.0 ± 5.5 ton CO_2 ha^{-1} y^{-1} emitted to the atmosphere, followed by UK-Po1 in 2023 (wheat) with 31.3 ± 2.1 ton CO_2 ha^{-1} y^{-1} (Figure 4b). The lowest emissions were observed at UK-Swt in 2023 (wheat) with 4.2 ± 3.0 ton CO_2 ha^{-1} y^{-1}. UK-Pob also had lower emissions compared to the other sites in 2021 (spring wheat) with 6.3 ± 1.5 ton CO_2 ha^{-1} y^{-1}. Averaging by farm, UK-Stm2, UK-Po1 and UK-Po2 were the sites with the highest emissions with 38.0 ± 5.5, 31.3 ± 2.1, and 27.3 ± 2.0 ton

CO_2 ha^{-1} y^{-1}, respectively. UK-Pob and UK-Swt were the sites with the lowest mean annual emissions with 6.6 ± 1.5 ton CO_2 ha^{-1} y^{-1} and 6.4 ± 3.6 ton CO_2 ha^{-1} y^{-1} (Figure 3b). Considering all the site-years, the agricultural peatlands emitted, on average, 23.1 ± 10.4 ton CO_2 ha^{-1} y^{-1}, with a large variation observed across sites (6.4 ton CO_2 ha^{-1} y^{-1} to 38.0 ton CO_2 ha^{-1} y^{-1}). Sites with peat depth >40cm emitted 25.1 ± 9.2 ton CO_2 ha^{-1} y^{-1}, while wasted peat sites emitted, on average, 11.8 ± 4.8 ton CO_2 ha^{-1} y^{-1}.

The multiple winter wheat crops grown at different study sites allowed us to compare NEP for the same crop across sites. Interestingly, winter wheat exhibited a large variation in in situ NEE when comparing across sites. It was positive (C emission) at UK-Rdm, UK-Po1, UK-Po2 and UK-Spy in 2023, but negative (C uptake) at UK-Pob in 2021 and UK-Swt in 2022 (Figure 4a). When including C imports and exports, all the sites had similar crop yields (19.0 ± 3.9 ton CO_2 ha^{-1}) although UK-Pob had the lowest CO_2 export (11.0 ton CO_2 ha^{-1}), about half of what was observed at the other sites. UK-Pob and UK-Swt were the smallest C sources in NEP~6 ton CO_2 ha^{-1} y^{-1}, while the other sites were larger C sources (> 20 ton CO_2 ha^{-1} y^{-1}).

3.2.3 | Evapotranspiration (ET)

The variance in ET was less than that observed for NEE, but there were some noteworthy differences across site-years (Figure 4c). The lowest ET values were observed at UK-Po1 with
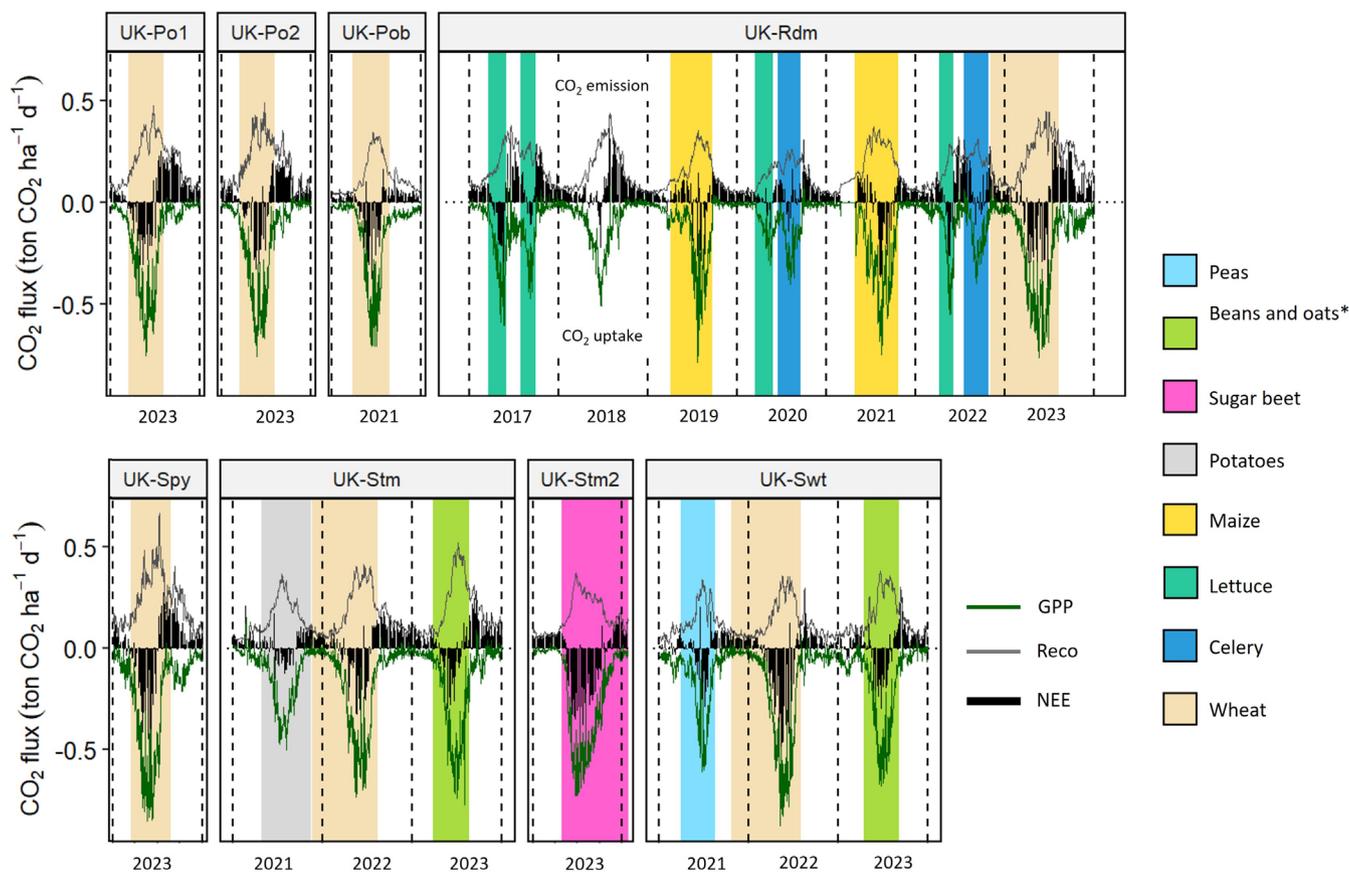
FIGURE 3 | NEE (ton CO_2 ha^{-1} d^{-1}, black bars), GPP (ton CO_2 ha^{-1} d^{-1}, green line) and R_{eco} (ton CO_2 ha^{-1} d^{-1}, grey line) for croplands on lowland agricultural peat in the Fens. The shaded areas represent the crop growing period for each crop. *Beans were planted as a single crop at UK-Stm, but they were intercropped with oats at UK-Swt. The fluxes at UK-Rdm during 2018 were not considered as the tower was moved to a different field in the same farm.

TABLE 3 | Mean and standard deviation of annual net ecosystem exchange (NEE, ton CO_2 ha^{-1} y^{-1}), gross primary productivity (GPP, ton CO_2 ha^{-1} y^{-1}), ecosystem respiration (Reco, ton CO_2 ha^{-1} y^{-1}), net ecosystem carbon balance (NEP, ton CO_2 ha^{-1} y^{-1}), and evapotranspiration (ET, mm y^{-1}) for each study site.

Site	NEE	NEE SD	GPP	GPP SD	Reco	Reco SD	NEP	NEP SD	ET	ET SD
UK-Po1	14.0	0.8	59.0	3.8	73.0	2.1	31.3	2.1	309.6	15.9
UK-Po2	10.4	0.8	60.2	3.9	70.6	2.2	27.3	2.0	390.4	16.0
UK-Pob	−4.4	0.9	51.4	3.5	46.9	1.7	6.6	1.5	478.0	28.9
UK-Rdm	14.9	3.5	45.7	7.2	61.7	4.1	24.2	5.5	609.7	70.2
UK-Spy	4.0	1.0	81.4	4.5	85.4	2.3	24.8	2.5	484.4	19.6
UK-Stm	10.6	2.7	53.1	6.0	63.7	3.5	26.1	4.2	593.1	65.8
UK-Stm2	−14.9	1.4	77.3	4.3	62.4	1.6	38.0	5.5	610.1	33.2
UK-Swt	−4.0	2.4	55.0	6.3	51.0	2.9	6.4	3.6	457.7	44.3

309±16 mm in 2023, and UK-Po2 with 390±16 mm in 2023. UK-Rdm had the highest ET values in 2017 (698±27 mm) and 2020 (692±23 mm), which corresponded to a lettuce/lettuce cycle in 2017 and a lettuce/celery rotation in 2020, both of which received irrigation (300–350 mm, Table S3). On average, UK-Rdm and UK-Stm2 had the highest ET out of all farms, with 610±70 mm and 610±33 mm, respectively.

3.2.4 | Ecosystem Water Use Efficiency (eWUE) and Crop Water Productivity (CWP)

Winter wheat at UK-Po1 was the crop with the highest eWUE, with a value of 22.4 g CO_2 kg^{-1} H_2O, followed by sugar beet at UK-Stm2 with 17.8 g CO_2 kg^{-1} and wheat at UK-Spy with 17.7 g CO_2 kg^{-1}. By contrast, salad crops had the lowest eWUE. The
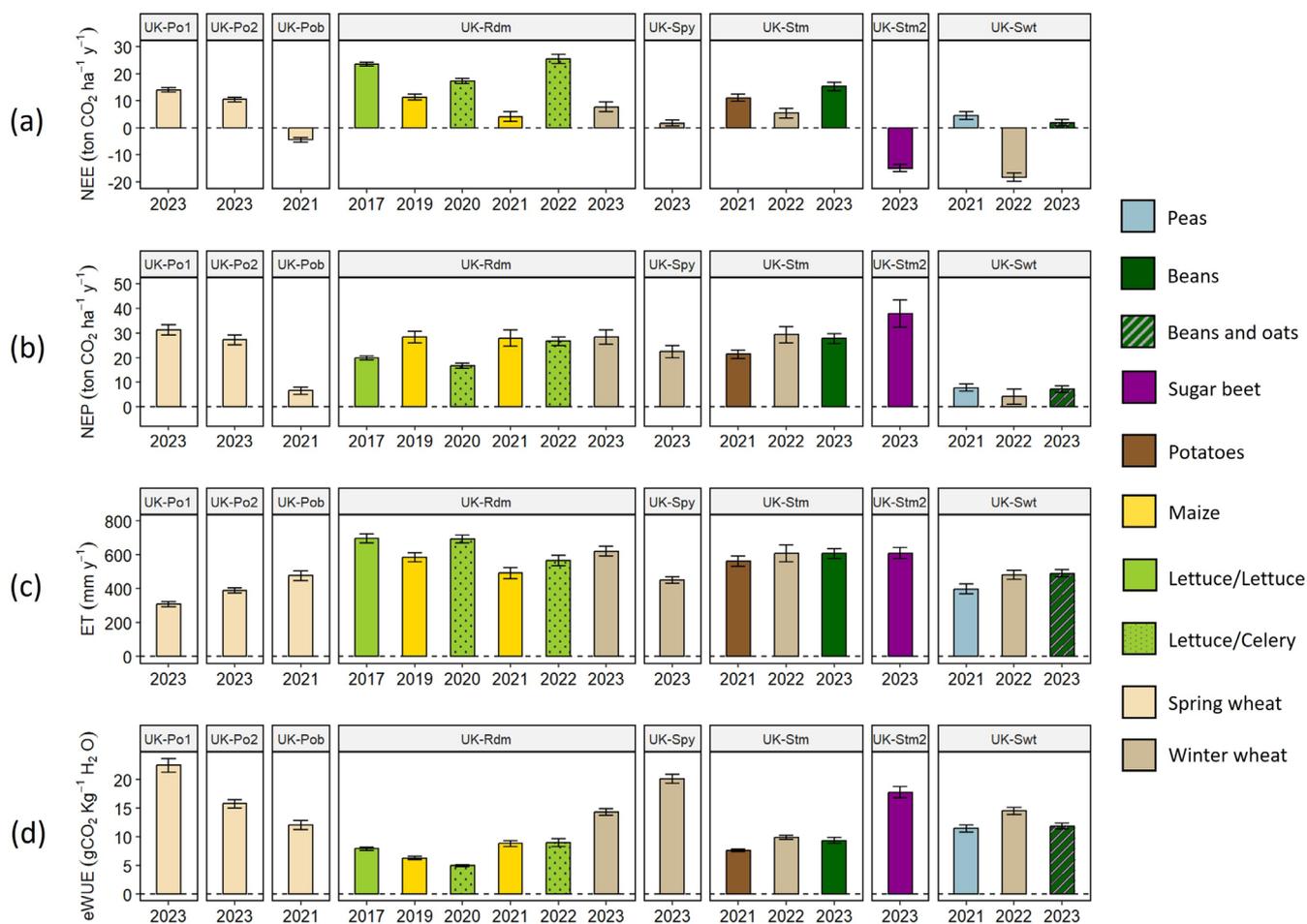
FIGURE 4 | (a) Total annual NEE (ton CO_2 ha^{-1} y^{-1}), (b) NEP (ton CO_2 ha^{-1} y^{-1}), (c) ET (mm y^{-1}), and (d) eWUE (g CO_2 kg^{-1} H_2O) for each site-year. Calendar years were used for all sites except at UK-Stm2 where fluxes were considered from 2023-02-01 to 2024-01-31 to account for a full crop year (sugar beet), and at UK-Spy from 2023-04-01 to 2024-03-31 as measurements started in March 2023. The fluxes at UK-Rdm during 2018 were not considered as the tower was moved to a different field in the same farm.

lettuce/lettuce and lettuce/celery cycles had values ranging from 4.9 g CO_2 kg^{-1} H_2O to 8.8 g CO_2 kg^{-1} H_2O (Figure 4d). In terms of CWP (Figure S2), UK-Stm2 with sugar beet in 2023 had the highest CWP (88 g kg^{-1} H_2O), followed by UK-Po1 with wheat in 2023 and UK-Rdm with Maize in 2021 (CWP = 4.2 and 3.55 g kg^{-1} H_2O, respectively). The salad crops had the lowest CWP, with values ranging from 0.4 to 0.6 g kg^{-1} H_2O.

3.3 | Carbon and Water Intensities

Although peat depth is a confounding variable in interpreting emissions by crop type, grouping the data from all the site-years by crop allowed us to compare how each crop year fared in terms of emissions per crop calorie (Figure 5).

Out of all the crops in this study, fields with sugar beet showed the highest CO_2 emissions, closely followed by fields with beans and maize (38.0 ± 5.5, 27.7 ± 2.0 and 26.7 ± 4.0 ton CO_2 ha^{-1} y^{-1}, respectively). Fields with the mix of spring beans and oats and peas, on the other hand, showed the lowest emissions with 7.3 ± 1.3 ton CO_2 ha^{-1} y^{-1} and 7.8 ± 1.5 ton CO_2 ha^{-1} y^{-1}, respectively (Figure 5a).

In terms of water loss (Figure 5b), the irrigated fields had the highest annual ET. Lettuce/lettuce year had the highest value overall (697 ± 27 kg m^{-2} y^{-1}), followed by lettuce/celery (630 ± 38.1 kg m^{-2} y^{-1}) and sugar beet (610 ± 33 kg m^{-2} y^{-1}). Peas and maize had the lowest ET, with values of 399 ± 28 and 482 ± 76 kg m^{-2} y^{-1}, respectively.

When considering the annual CO_2 emissions to produce a kcal of harvestable produce and the water use per kcal for each crop, we accounted for the total intensity (all available kcal) and the intensity for food (kcal destined for direct human consumption) (Figure 5c). In terms of total C intensity, the site-years with lettuce/lettuce or lettuce/celery rotations had the highest values, with 3.7 and 3.6 g CO_2 $kcal^{-1}$. The lowest intensities were observed at peas and sugar beet site-years, with values of 0.20 and 0.29 g CO_2 $kcal^{-1}$. When focusing only on kcal delivered directly for human sustenance, fields with salad crops were the most CO_2 intensive (3.2 and 3.0 g CO_2 $kcal^{-1}$ for lettuce/lettuce and lettuce/celery, respectively), whereas peas were the least intensive (0.11 g CO_2 $kcal^{-1}$) followed by wheat (0.23 g CO_2 $kcal^{-1}$) (Figure 5d). We note that maize and sugar beet have dual uses as food and energy crops but focus solely on the food use here to enable comparisons across crops.
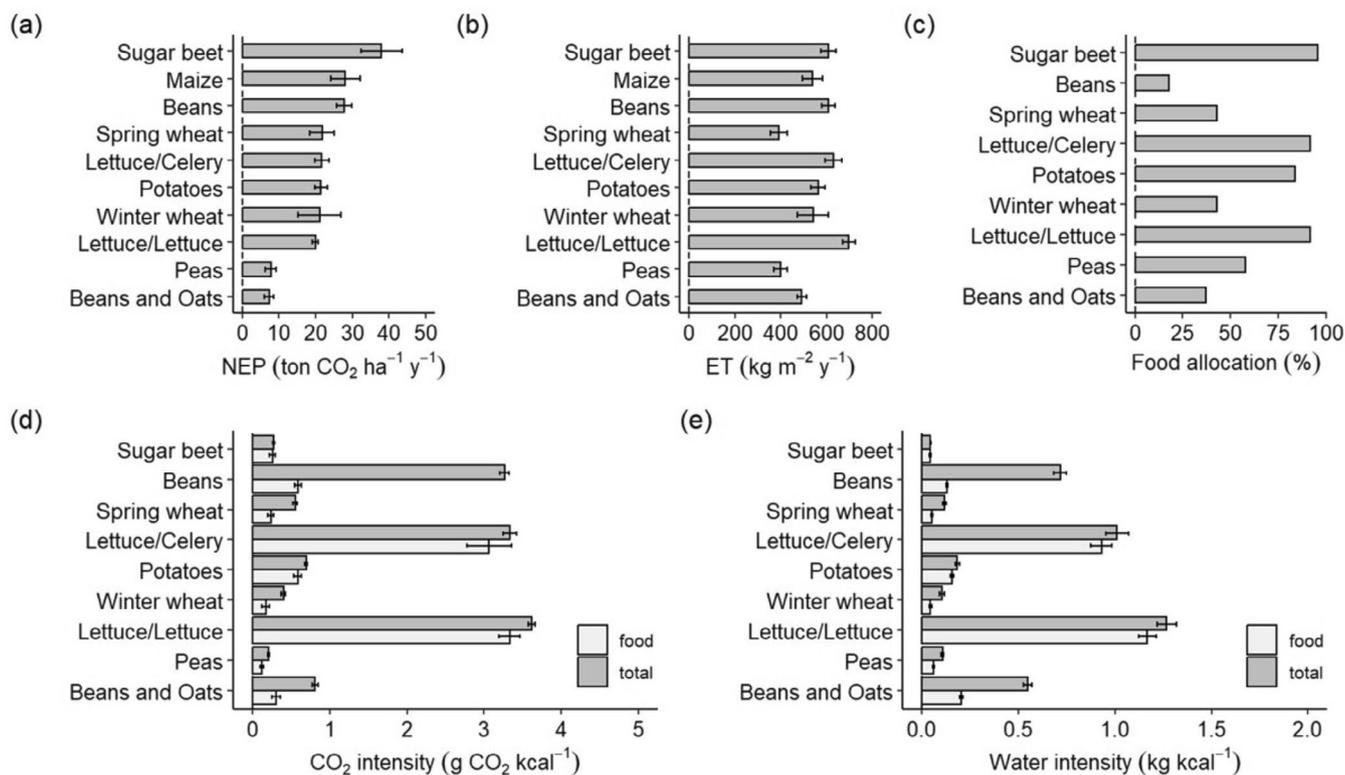
FIGURE 5 | (a) Net ecosystem C balance (NEP, ton CO_2 ha^{-1} y^{-1}); (b) Evapotranspiration (ET, kg m^{-2} y^{-1}); (c) Percent of crop production allocated for direct human consumption (%) based on FAO data for the UK in 2024; (d) Carbon intensity (g CO_2 $kcal^{-1}$) and (e) Water intensity (kg $kcal^{-1}$) for each crop in this study. Note that maize was removed from the CO_2 and water intensities calculations as the maize produced in the fens is solely for fodder.

For the total water intensity, again, the lettuce/lettuce and lettuce/celery rotations were the most intensive (1.3 and 1.0 kg $kcal^{-1}$), whereas the peas and sugar beet were the least intensive (0.09 and 0.04 kg $kcal^{-1}$). When only considering the kcal for direct human consumption, lettuce/lettuce and lettuce/celery remained as the most water intensive crops (1.20 and 0.9 kg $kcal^{-1}$) and sugar beet (0.03 kg $kcal^{-1}$) and wheat (0.04 kg $kcal^{-1}$) were the least water intensive (Figure 5e).

3.4 | Environmental Predictors of Fluxes

We found that the model that included the percent of organic carbon (% OC, g cm^{-3}), effective water table depth (WTD_e, cm) (i.e., the mean depth of peat exposed to aerobic conditions) (Evans et al. 2021), and mean annual temperature (MAT, °C) was the best predictor of the annual net ecosystem productivity (ton CO_2 ha^{-1}) (Figure 6a–c), with the best model taking the form of:

$$NEP = 0.2 \times WTD_e + 9.9 \times MAT + 0.6 \times OC - 111.1, R^2 = 0.84, p < 0.05, n = 8$$

A simple linear model between the WTD_e and the NEP was only able to explain 69% in the variation of NEP across sites. Including the % OC improved the Akaike information criterion (AIC) from 32.9 to 28.0, and when the MAT was included in the model, the AIC decreased to 26.1.

In terms of Reco, we found that the best model included the MAP, bulk C and WTD_e (Figure 6d–f). The annual precipitation alone was able to explain 90% of the variation in Reco across sites (Figure 6e). Including the WTD_e and bulk C improved the model slightly.

$$Reco = -0.1 \times WTD_e + 0.08 \times MAP + 417.5 \times Bulk_C - 40.8$$

$$R^2 = 0.96, p < 0.01, n = 8$$

We also evaluated the site-years with wheat only ($n = 7$) and found that the NEP was best predicted by the bulk C ($R^2 = 0.8$, $p < 0.01$). We also found a strong linear relationship between ecosystem respiration and OC ($R^2 = 0.90$, $p < 0.001$) (Figure S2).

4 | Discussion

4.1 | Environmental Predictors of Fluxes

Our results show that crop type alone was not a major influence on NEP. Instead, emissions across sites were strongly related to WTD_e, organic carbon content (OC) and the mean annual temperature (MAT). Previous studies have shown a similar relationship between the NEP and these variables (Evans et al. 2016, 2021; Koch et al. 2023; Musarika et al. 2017; Tiemeyer et al. 2016), where CO_2 emissions increase as the WTD decreases until it reaches the bottom of the peat layer, at which point emissions plateau because
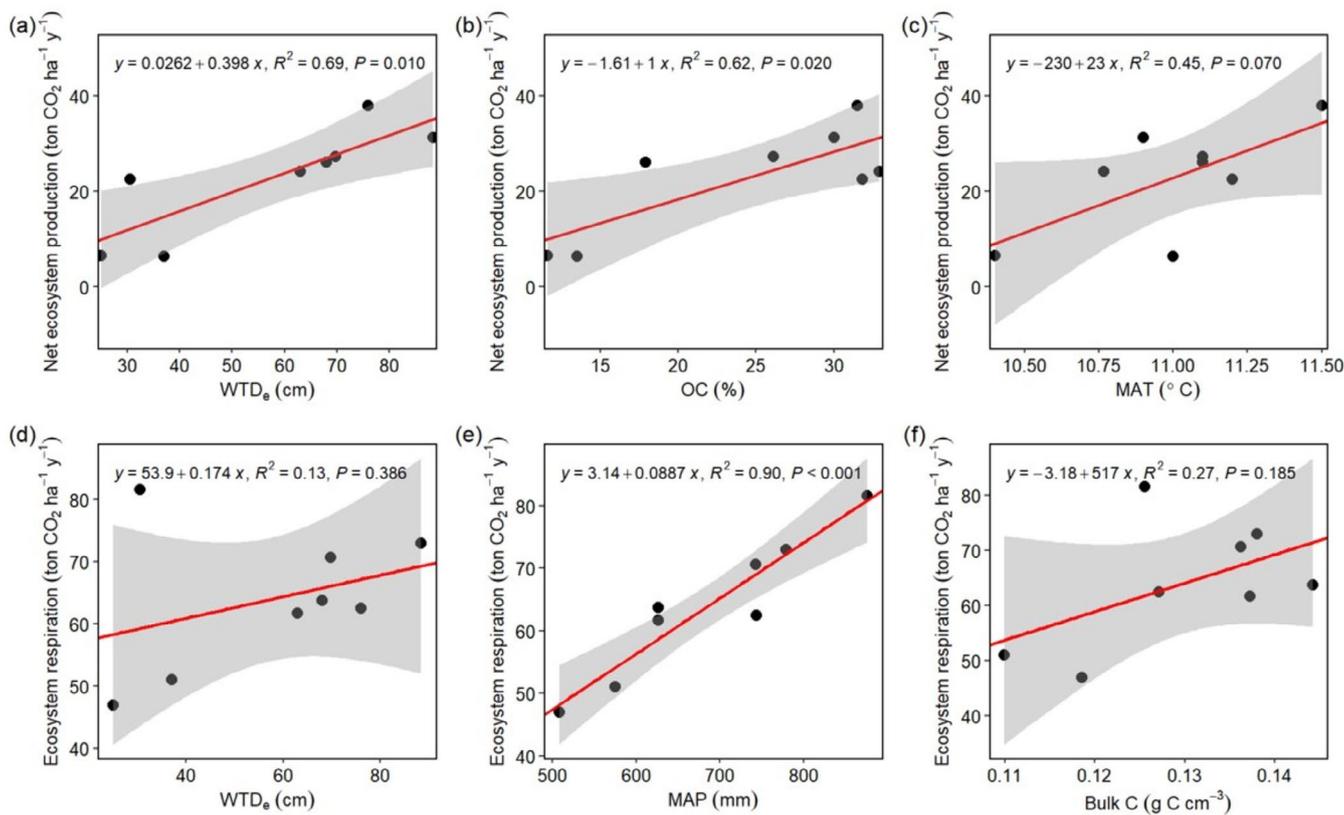
FIGURE 6 | (a) Relationship between net ecosystem productivity (NEP, ton CO_2 ha^{-1} y^{-1}) and effective water table (WTD$_e$, cm). (b) Relationship between NEP and OC (%). (c) Relationship between NEP and mean annual temperature (MAT, °C). (d) Relationship between R$_{eco}$ (ton CO_2 ha^{-1} y^{-1}) and WTD$_e$ (cm); (e) relationship between R$_{eco}$ and mean annual precipitation (MAP, mm); and (f) relationship between Reco and bulk carbon (g C cm^{-3}). The shaded grey area represents the standard error of the fit.

the water table reaches the mineral layer. Because all the peatlands in this study, except for UK-Spy, consistently have a water table below the peat depth, sites with deeper peat have more soil OC exposed to aerobic microbial decomposition, which is in turn reflected in higher respiration rates compared to sites with shallow peat depth or peat with reduced OC content. This explains the lowest emissions observed at the sites with wasted peat (UK-Pob and UK-Swt) compared to sites with deeper, C-rich peat.

Evans et al. (2021) analysed 16 UK and Irish CO_2 flux towers and found a similar linear relationship between NEP and WTD$_e$ (NEP = 0.1341 × WTD$_e$—1.73, R^2 = 0.90), although our sites show a lower gradient, and including the % OC and MAT greatly improved the predictive power of the model. This shows that, at agricultural sites, WTD$_e$ has a limited influence on NEP because the sites are heavily drained, thus WTD$_e$ reflects a more static site characteristic (e.g., peat depth, OC) rather than a dynamic hydrological driver. Therefore, it may be also capturing the effects of the variation in OC and BD across sites. However, we did not observe a strong relationship between WTD$_e$ and OC or BD. In addition, other climatic variables (e.g., MAT, MAP) can exert control on the CO_2 exchange between the surface and the atmosphere and is likely related to the influence of temperature and water availability on crop growth or on rainfall-induced CO_2 pulses during dry periods, as explained below.

We also found a strong relationship between MAP and Reco (R^2 = 0.90), with sites that received more precipitation

generally emitting more than drier sites. Because water tables at most sites remain consistently low, often below the peat layer (Figure 2), precipitation appears to have little influence on overall water table depth. Rather than substantially rewetting the peat profile, rainfall likely produces short-lived increases (i.e., 'pulse') in soil moisture conditions within the upper, oxygenated peat layer, enhancing microbial activity and respiration. This pulse effect has been observed in previous studies for multiple ecosystems (Nguyen et al. 2025; Nijman et al. 2024; Placella et al. 2012; Säurich et al. 2019). For our sites, this would mean that as the water table becomes deeper, its direct regulatory effect on greenhouse gas emissions likely diminishes, whereas the importance of rainfall dynamics and irrigation inputs increases.

The relationship observed between NEP and OC, and Reco and OC content is different from the findings of other studies on CO_2 emissions from managed peatlands (Eickenscheidt et al. 2015; Leiber-Sauheitl et al. 2014; Liang et al. 2024; Renou-Wilson et al. 2014; Veenendaal et al. 2007). Liang et al. (2024), for example, measured CO_2 emission rates from peat cores collected across Denmark in an incubation experiment at standardised water potentials and found that OC was a poor predictor of area specific CO_2 emissions. Instead, they hypothesised that the differences across sites were due to the combined effects of soil physics, geochemical properties, and microbial communities at the study sites. Similarly, Leiber-Sauheitl et al. (2014) measured GHG emissions from deep and shallow organic soils

and found that R_{eco} was correlated with the WTD, but not to the SOC content. By contrast, Renou-Wilson et al. (2014) and Veenendaal et al. (2007) attributed some of the variations in CO_2 emissions to the soil OC content. These discrepancies might be due to the different methods used to estimate the fluxes. Liang et al. (2024) conducted a 60-day experiment with samples taken every 5–7 days and measured heterotrophic respiration, whilst we used high frequency measurements of CO_2 fluxes to obtain the annual NEE and modelled the R_{eco} (i.e., heterotrophic and autotrophic) from it. Likewise, Leiber-Sauheitl et al. (2014) and Eickenscheidt et al. (2015) measured fluxes using manual chambers, with measuring intervals of 2 weeks for the former, and 8–60 days for the latter. Fluxes were then extrapolated to obtain annual CO_2 emissions. The different flux aggregation methods depending on the time scale could be the reason for a lack of relationship.

The soil organic matter quality and peat decomposition levels might also play a role in determining peat R_{eco} rates. Although we did not investigate the quality of the organic matter of our sites, we hypothesise that the soils presented here are in a higher state of decomposition than the ones previously reported in other studies as our study sites are heavily managed, with the WTD around 1 m belowground for most of the sites throughout the year, with the exception of UK-Spy which had a WTD of ~30 cm. In fact, dry bulk density for our study sites had a mean value of $0.60 \, g \, cm^{-3}$ which is higher than the mean bulk density of $0.49 \, g \, cm^{-3}$ for the sites in Liang et al. (2024). Future studies that combine measurements of soil chemical and physical properties and autotrophic and heterotrophic respiration rates might help elucidate the main drivers of CO_2 emissions from managed peatlands. Moreover, soil properties can vary significantly across space and time due to natural variations in soil hydrology and biogeochemistry, but also due to crop and field management, and climatic conditions (Ceschia et al. 2010; Elsgaard et al. 2012; Pagliai et al. 2004; Peacock et al. 2017). Scaling soil properties over the flux tower footprint at high temporal resolutions could improve the relationships between NEP and soil predictors.

While we could not identify a direct influence of crop type on peat CO_2 emissions, it is worth noting that drainage depths may be partly influenced by the types of crop present at either a field or farm scale, for example root crops generally require deeper drainage than salad crops. Furthermore, cropping across the Fenland region is partly determined by the depth of remaining peat, with salad and vegetable crops preferentially growing on areas of deeper peat, and thinner ('wasted') peat soils mainly used for cereals. Crop type may therefore be an indirect driver of CO_2 emissions, or an effective response variable, with both crop type and emissions varying as a function of peat depth. Moreover, studies have shown that interannual climatic variability, field and crop management (i.e., tillage, irrigation regimes, fertilisation, planting depths, drainage) and harvest characteristics (e.g., belowground biomass removal from sugar beet and potatoes harvest) influence soil CO_2 emissions (Abdalla et al. 2016; Bilandžija et al. 2016; Glenn et al. 2011; Kallenbach et al. 2010), all of which make disentangling the crop type effects from interannual and site-specific conditions much more complicated. We cannot rule out more direct influences of crop type and management on emissions, for example as a function of the degree of soil

disturbance, irrigation or fertiliser use, but any such relationships could not be detected within our current dataset.

4.2 | Carbon and Water Intensities

The East of England is expected to experience warmer and drier conditions by 2050 because of climate change (Jenkins et al. 2024). Previous studies have shown that the water balance in East Anglia is shifting progressively towards an increasing supply deficit (Henriques et al. 2008). Moreover, the highest water abstractions for irrigation in England are in Lincolnshire and East Anglia (Jenkins et al. 2024; Knox et al. 2020). This indicates that the current water management practices, including irrigation for agricultural production, will likely become increasingly unsustainable in a drier future (Jenkins et al. 2024). In addition, more frequent extreme events (i.e., droughts, floods) and overall increasing climatic variability are expected to increase crop vulnerability to climate change as yields become more unstable (Putelat et al. 2021). Here, we found that, in terms of the NEP, effective peat depth and carbon content were more important than crop type in determining total emissions. However, when expressed in terms of the intensity of CO_2 per kilocalorie of different food crops, the emissions associated with lettuce and celery on peat were one order of magnitude larger than those for sugar beet, peas, potatoes and wheat. Likewise, the water used to produce 1 kcal of salad crops (which includes substantial irrigation) is more than one order of magnitude larger than that used to produce sugar beet, peas and wheat. If we only consider the percentage of crop production that is used for direct human consumption, lettuce and celery, again, stand out as the most C and water intensive crops. The high CO_2 intensity observed is due to the low number of calories these crops provide (0.1 kcal/g of fresh weight). In this context, maximizing the calories produced per tons of CO_2 emitted could be a pathway towards maintaining food security while mitigating GHG emissions. High-value salad and other vegetable crops such as lettuce, celery, radish and onions are preferentially grown on peat soils in the UK due to their high water retention capacity, controlled drainage, fertility and ease of cultivation (Rhymes et al. 2024). Increased consumption of fresh produce is an important for a healthy diet and forms part of the recommendations of the UK's National Food Strategy (Dimbleby 2021), but our data show that comes at an environmental and societal cost, both in terms of C emissions and water use. This is particularly acute when considering the amount of calories produced. Given that East Anglia and Lincolnshire are among the most vulnerable regions of the UK in terms of both water security and risks to soil carbon stocks, a shift towards crops with a higher calorific content and/or lower demand for water (including supplementary irrigation) might improve the sustainability of agriculture in the region. However, this would likely come at an economic cost in terms of the value of agricultural produce, and relocating fresh vegetable production off peat remains challenging, with a risk of displacing CO_2 emissions and even exacerbating water scarcity issues (Rhymes et al. 2024). Reducing the use of high-grade agricultural land on organic soils for non-food crops such as wheat for animal feed or maize for bioenergy (e.g., Evans et al. 2024) would provide an alternative mitigation strategy, by releasing some land for other uses (e.g., higher water table agriculture, restoration to wetland). These are, however, ultimately

societal and policy choices, that the data presented in this study can help to inform.

An important caveat of this study is that although caloric intake is often used as a proxy for food security (Cassidy et al. 2013), it does not account for the full spectrum of nutritional needs or dietary preferences. Considering the nutritional value of the crops (e.g., proteins, vitamins, fibre content) is important for a more comprehensive assessment of nutritional adequacy. Future studies could focus on integrative metrics that account for carbon emissions and water use per nutrient density (i.e., nutrient profile), which will consider both an environmental and nutritional perspective (Doran-Browne et al. 2015; Pradhan et al. 2013; Rippin et al. 2021; Smedman et al. 2010). Finally, we used national proportions of crop production for food allocation to estimate the C and water intensities for food production for each crop, but these proportions may vary regionally. More detailed life-cycle analyses will benefit from the dataset presented here.

4.3 | Comparison to Other Ecosystems

Northern peatlands can store vast amounts of C in their soils and therefore have the potential to play an important role in either mitigating or exacerbating climate change. Peacock et al. (2019) found that a conservation-managed site in East Anglia, with a peat depth of almost 4 m, was a net C sink of -3.8 ton CO_2 ha^{-1} yr.$^{-1}$. Likewise, other natural or semi-natural fens and bogs have also shown moderate rates of C accumulation, (e.g., -1.2 ton CO_2 ha^{-1} yr.$^{-1}$) (Yu 2012), -3.8 ton CO_2 ha^{-1} yr.$^{-1}$ (Lund et al. 2010), -1.8 ton CO_2 ha^{-1} yr.$^{-1}$ (Humphreys et al. 2014; Peichl et al. 2014; Pelletier et al. 2015). In contrast, all the sites in this study were C sources to the atmosphere when accounting for the C_{export} via harvest, ranging from 6.4 to 38.0 ton CO_2 ha^{-1} yr.$^{-1}$. These values are similar to the ones reported in studies that focus on managed peatlands (Table S4), with NEP values ranging from 11 to 55 ton CO_2 ha^{-1} yr.$^{-1}$ (Ceschia et al. 2010; Elsgaard et al. 2012; Haddaway et al. 2014; Peacock et al. 2019). This is as expected as drainage exposes the organic layer to oxic conditions, where it is quickly converted to CO_2 (Haddaway et al. 2014).

Our results indicate an oxidative loss rate of 0.04–0.9 cm y^{-1} of the peat layer, which would suggest a total peat loss within the next 85–600 years. These values agree with the findings of Cumming (2018), who estimated an oxidative loss rate of 0.8 cm y^{-1} for a managed peatland in the East Anglian Fens with a bulk density of 0.26 g cm^{-3} and a peat depth between 1 and 2 m, and also with the results presented in Evans et al. (2016), who estimated a total loss of the peat layer of 5 managed lowland peat soils in England within the next 200–500 years.

Crops grown on peat also exhibit higher emissions than the same crops grown on mineral soils (Table S4). For example, celery was a net C sink of ~11 ton CO_2 ha^{-1} yr.$^{-1}$ when grown in a mineral soil in the Jiangsu province in China (Jia et al. 2012), whereas the lettuce/celery years at UK-Rdm emitted 15.4 ton CO_2 ha^{-1} yr.$^{-1}$ and 25.6 ton CO_2 ha^{-1} yr.$^{-1}$. Winter wheat years for mineral soil sites were also small to moderate C sinks with values of -1.2, -3.3, and -5.9 ton CO_2 ha^{-1} yr.$^{-1}$ (Ceschia et al. 2010), or small sources of C with a mean NEP of 3.4 ton

CO_2 ha^{-1} yr.$^{-1}$ (Ceschia et al. 2010). Interestingly, NEP from the winter wheat years at UK-Swt and UK-Pob with highly wasted peat soils are only slightly higher than these values, indicating that these sites behave more similar to a mineral site due to the thin peat layer. Emissions from winter wheat at sites with true peat layers (UK-Stm and UK-Rdm) were double or triple the emissions reported for wheat on mineral soils.

Raising the water table could decrease CO_2 emissions from agricultural peatlands (Evans et al. 2021; Freeman et al. 2022; Wilson et al. 2016). However, more research is needed to fully ascertain the effect of a higher water table on GHG emissions, as it can reduce CO_2 and potentially N_2O emissions, but may increase CH_4 emissions if water levels are raised close to the peat surface (Evans et al. 2021; Taft et al. 2017, 2018). Moreover, maintaining a higher water table during the growing season could have a negative impact on crop yields and impede in-field operations (Matysek et al. 2019, 2022; Taft et al. 2018). Moving from conventional agriculture to paludiculture (i.e., biomass production on wet peatlands) could be a sustainable alternative for peatland use, especially as many countries move towards net zero by 2050 goal. Although paludiculture seems promising in terms of climate mitigation, there is a limited understanding of the implications of large-scale conversion of agricultural peatlands to paludiculture, which would modify the whole food supply system, with potentially negative effects on farm incomes and food security (Lahtinen et al. 2022; Page and Baird 2016; Tan et al. 2021). This change would need to be accompanied by an increase in agricultural yields from mineral soils and/or alternative (e.g., indoor) farming systems to account for the loss in agricultural output from peatlands, as well as a reduction in the demand for animal feed (Lahtinen et al. 2022).

Finally, it is important to consider that in this study we focused on the exchange of CO_2 between the surface and the atmosphere, and the lateral C losses via crop harvest. We did not account for the additional emissions from the fuel used for farm machinery, from pesticide and fertilizer manufacturing, packaging, transport, and storage, or from irrigation. However, Ceschia et al. (2010) found that emissions from field operations only represented a small percentage of the total greenhouse gas budget (< 10% of emissions), with soil emissions dominating the overall C balance in mineral soils. In addition, agricultural soils are usually treated with mineral and organic fertilizers that, through nitrification and denitrification, release nitrous oxide (N_2O) to the atmosphere. Peatlands can also emit CH_4 as higher water tables create anoxic conditions that promote methanogenesis. The presence of ditches to manage the water tables also contributes to the C budget of the fields in terms of CO_2, CH_4 and N_2O emissions (Hendriks et al. 2024; Peacock et al. 2017, 2021), as well as lateral losses of C through the movement of POC and DOC (D'Acunha et al. 2019; Roulet et al. 2007; Strack et al. 2015). Peacock et al. (2017) and Evans et al. (2016) report ditch CO_2 emissions ranging from 0.2 to 1.3 ton CO_2 ha^{-1} for agricultural peatlands in the fens. It is important to mention that ditch water is also used to irrigate some crops (e.g., potatoes, lettuce, celery) and could be an additional, although small, C input into the field that we are not considering in this study. Moreover, the low bulk density of peat, often exposed to wind in cultivated peatlands, can lead to peat loss through aeolian sediment transport. Cumming (2018) estimated an additional loss of 3.3–18.0

ton CO_2 ha^{-1}yr.$^{-1}$ via wind erosion and found that this loss is exacerbated by dry conditions during the summer months and management events (e.g., tilling, harvesting). However, the fate of this carbon (i.e., within the field, in a ditch, on a neighbouring field) and its mineralization rate remains uncertain.

Therefore, accounting for the different components of the C budget is necessary to estimate the full greenhouse gas budget and should be the focus of future studies.

5 | Conclusion

All the sites in this study were strong C sources, emitting on average 23.1 ± 10.4 ton CO_2 ha^{-1} y^{-1}. Sites with peat depth > 40 cm emitted 25.1 ± 9.2 ton CO_2 ha^{-1}, while wasted peat sites emitted, on average, 11.8 ± 4.8 ton CO_2 ha^{-1}. We found that NEP is mainly determined by the amount of peat organic matter exposed to oxidation by drainage, rather than crop, but the C and water intensities depend very strongly on crop type, with salad crops being one order of magnitude more C and water intensive than cereals. The findings in this study can help inform policies for better land management to reduce C emissions from these ecosystems whilst maintaining food production. Future research should focus on understanding the full GHG budget of agricultural peatlands, as well as on alternative management practices such as paludiculture.

Author Contributions

Brenda D'Acunha: conceptualization, investigation, writing – original draft, methodology, formal analysis, data curation, visualization, writing – review and editing. **Chris D. Evans:** funding acquisition, writing – review and editing. **Alanna Bodo:** investigation, writing – review and editing. **Hollie Cooper:** data curation, writing – review and editing. **Dafydd Egryn Crabtree:** investigation, writing – review and editing. **Alexander Cumming:** data curation, investigation, writing – review and editing. **Jennifer M. Rhymes:** investigation, writing – review and editing. **Daniel Rylett:** investigation, writing – review and editing. **Rebekka R. E. Artz:** conceptualization, funding acquisition, writing – review and editing. **Ross Morrison:** conceptualization, funding acquisition, methodology, writing – review and editing.

Conflicts of Interest

The authors declare no conflicts of interest.

Data Availability Statement

The data that support the findings of this study are openly available in Zenodo at 10.5281/zenodo.18669937.

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Supporting Information

Additional supporting information can be found online in the Supporting Information section. **Table S1:** Flux towers measurement period and instrumentation. MAT = mean annual temperature from 1961 to 1990, MAP = mean annual precipitation from 1961 to 1990. **Table S2:** Friction velocity threshold (u_*, m s^{-1}), available data (%) and energy balance closure (EBC) for each site-year in this study. **Table S3:** Management information for all the sites. **Table S4:** Comparison with other sites. For the method, CH, chamber measurements; EC, eddy covariance; GC, gas chromatography. **Figure S1:** Mean monthly air temperature for all the study sites. The shaded area represents the maximum and minimum temperatures observed during the day. **Figure S2:** Relationship between Reco and OC (left), and NEP and MAP (right). **Figure S3:** Crop water productivity for each site-year. **Figure S4:** Relationship between NEP and Bulk C, NEP and MAP, and Ecosystem respiration and OC for wheat crop only.