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Overriding importance of water table in the greenhouse gas balance of managed peatlands.

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The global peatland carbon (C) store may exceed 1100 Pg^{1,2}, exceeding the current size of the atmospheric carbon pool. Peatlands in many parts of the world are under severe and increasing pressure from drainage-based agriculture, timber plantations and fire. Greenhouse gas (GHG) emissions from drained peatlands have been estimated at 1.2 to 1.9 Gt CO₂e yr⁻¹^{3,4}, equivalent to 2.3-3.8% of all anthropogenic GHG emissions⁵. With growing recognition of the significance of peatlands for the climate system, efforts to curb emissions through conservation of undrained peatlands and rewetting of drained systems are intensifying. Here, we describe a unique set of 16 CO₂ eddy covariance datasets, and accompanying methane (CH₄) data, collected within the relatively narrow geographic and climate range of the United Kingdom, which span a broad gradient of land-use intensity from near-natural bogs and fens to intensive grassland, cropland and peat extraction. Combining these results with published data from a further 52 eddy covariance sites from across the temperate and boreal zones, we show that mean annual effective water-table depth (WTD_e) represents the overwhelmingly dominant control on CO₂ fluxes, apparently overriding all other climatic, vegetation and land-management factors. We show that every 10 cm reduction in WTD_e will reduce the net warming impact of CO₂+CH₄ emissions (based on 100 year Global Warming Potentials) by at least 3 t CO₂e ha⁻¹ yr⁻¹, until WTD_e is < 30 cm. Raising water levels above that level continues to have a net cooling impact until WTD_e is < 10 cm. Our results suggest significant but largely unrealised potential to reduce GHG emissions from peatlands drained for agriculture, without necessarily having to halt their productive use. Assuming similar functional relationships between CO₂, CH₄ and WTD_e for tropical peatlands, we estimate that halving WTD_e in drained agricultural peatlands could reduce peatland GHG emissions by 450 Mt CO₂e yr⁻¹, representing almost two thirds of the maximum feasible climate change mitigation for this area, and around 1% of global anthropogenic CO₂ emissions.

Peatlands form where high rainfall or restricted drainage cause waterlogging, restricting oxygen supply and suppressing decomposition. Peatlands mostly occur in the northern cool temperate and boreal zones⁶ and the humid tropics, notably Southeast Asia, Amazonia and Central Africa². Large-scale drainage of peatlands began in Europe several centuries ago, and intensified in the 20th century following the advent of electrical pumping. The resulting aerobic organic soils release nutrients through decomposition and can provide fertile land for cropland, grassland and forestry. Out of a total European peatland area of 51 million ha, 21% has been drained for forestry, and 17% for cropland and grassland⁷. In Southeast Asia, an estimated 50-75% of Malaysian and Indonesian peat swamp forests have been deforested and drained since 1990, primarily for oil palm and pulpwood plantation⁸. Drainage leads to oxidative loss of peat, land subsidence, increased fire and flood risk, and the need for energy-intensive pumped drainage⁹⁻¹². As a result, crops grown on peat have among the highest production intensities (GHG emissions per crop calorie) in the world¹³. Nevertheless, peatland drainage and cultivation have contributed significantly to economic development in many areas, and cultivated organic soils support large populations in countries such as Indonesia. Despite the urgent need to reduce GHG emissions, it is therefore unlikely that these areas can be restored to natural wetlands in time to meet international commitments to achieve net zero GHG emissions under the Paris Agreement.

In 2014, the IPCC published new guidance on GHG emissions reporting for managed peatlands¹⁴. To develop emission factors (i.e. emissions of CO₂, CH₄, and nitrous oxide (N₂O) per unit area per year) peatlands were classified by climate zone, peat type (bog versus fen) and management (e.g. cropland, grassland, forest land, re-wetted peatland). Water-table regimes were recognised as important in determining differences between categories, but were rarely taken into account explicitly. However, a range of previous studies have identified mean water-table depth (WTD) as an important influence on the emissions of all three major GHGs, with deeper water tables favouring CO₂ and N₂O emissions, and shallow water tables favouring CH₄ emissions^{15,16}. Nevertheless, it is widely considered that other factors such as climate, vegetation, agricultural practices such as crop residue management and fertilisation, and intrinsic peat properties, exert strong additional influences on GHG emissions¹⁵⁻¹⁸. Furthermore, most studies of peatland management for climate mitigation have focused on re-wetting and restoration to functional wetland systems, or on high-water table agricultural use ('paludiculture'¹⁹). While these changes in management would have desirable endpoints from the perspective of re-initiating net CO₂ sequestration, the reduction or loss of direct economic income resulting from these measures, as well as uncertain outcomes with regard to CH₄ emissions, has precluded their large-scale implementation.

We collated CO₂ flux measurements from 16 peatland eddy covariance (EC) flux towers located across the UK (Extended Data Figure 1). The sites encompass the three main temperate peatland types (fen, lowland raised bog, upland blanket bog) and a diverse range of land-use, from near-natural (including conservation-managed and re-wetted) bogs and fens to extensive and intensive grassland, arable, horticulture and peat extraction. This density of EC measurements on peat is unique globally.

Measured Net Ecosystem Exchange (net gaseous CO₂ exchange, NEE) of the sites ranged from -2.5 to +6.9 t C ha⁻¹ yr⁻¹ (where negative values indicate net CO₂ uptake). Variations were clearly associated with land-use; all net CO₂ sinks were near-natural, and all drained sites were net CO₂ sources, with the largest losses from cropland and intensive grassland (Extended Data Table 1). Between-year variations in NEE at near-natural/conservation-managed sites were comparatively small, despite variable weather conditions between years (see Extended Data Figure 2). At cropland sites, NEE was more variable between years, due to different rates of CO₂ uptake by different crops, but after accounting for removal of C in harvested biomass, net ecosystem production (NEP) was again remarkably stable.

This suggests a steady underlying rate of heterotrophic respiration and associated CO₂ loss at cultivated sites, with the entire annual gross primary production (GPP) effectively lost to autotrophic respiration and biomass harvest.

For the sites with CO₂ flux towers we found a strong correlation between NEP and WTD_e:

$$\text{NEP} = 0.1341 \times \text{WTD}_e - 1.73 \quad (R^2 = 0.90, p < 0.001, n = 16) \quad (\text{Equation 1})$$

No other climatic, hydrological or soil quality variable provided any additional explanatory power ($p > 0.05$ for residuals of NEP vs WTD_e regressed against all other variables considered), and there was no evidence of systematic deviation from the best-fit line as a function of land-use type.

Combining our EC-based NEP data with comparable values from 49 published EC studies of other temperate and boreal peatlands that also reported WTD_e (Extended Data Table 2) we obtained a similar but shallower relationship (Figure 1b):

$$\text{NEP} = 0.0927 \times \text{WTD}_e - 1.69 \quad (R^2 = 0.68, p < 0.001, n = 65) \quad (\text{Equation 2})$$

Again, we found little evidence of systematic deviation from the best-fit line as a function of land use, although there were some individual outliers. Further analysis (Supplementary Information) showed that the model was not improved (i.e. Akaike Information Criterion did not reduce) if temperature and precipitation were included as additional explanatory variables, or if a non-linear model was applied. The underlying linear relationship with WTD_e also persisted if data were grouped by land use in a linear mixed-effect model. Based on Equation 2, boreal/temperate peatlands are predicted to sequester 1.58 t C ha⁻¹ yr⁻¹ as CO₂ on average when water tables are at the peat surface, and to act as net CO₂ sinks when WTD_e < 20 cm. Peatlands with WTD_e > 25 cm were overwhelmingly net CO₂ sources.

We also analysed a total of 41 measurements of annual mean CH₄ flux, all from peatlands in the UK and Ireland, and made using comparable methods, spanning a range of land management, peat type and WTD_e (see Methods, Extended Data Table 3). This dataset shows a clearly non-linear relationship between mean annual CH₄ and WTD_e, with maximum emissions of around 0.2 t CH₄-C ha⁻¹ yr⁻¹ when WTD_e is close to the peat surface, and predominantly near-zero emissions when WTD_e > 30 cm (Figure 1c). The observations conformed to an exponential relationship:

$$\text{CH}_4 = 0.334 \times 0.5^{((\text{WTD}_e+5)/6.31)} \quad (\text{Equation 3})$$

(RMSE 0.044 t C ha⁻¹ yr⁻¹, predicted versus observed $R^2 = 0.55, p < 0.001, n = 41$)

Again, no other measured site variable provided additional explanatory power, although previous studies have shown a significant influence of aerenchymatous vascular plant species on CH₄ emissions from semi-natural peatlands, and of temperature when the water table is close to the surface^{16-17,20-21}. Comparison of our results with data collected from other biogeographic regions suggests that the obtained relationship is broadly applicable (Extended Data Figure 3).

Empirical relationships with WTD_e for CO₂, CH₄ and their combined climate forcing impact are shown in Figure 1d. For CO₂, the observed linear relationship suggests that a 10 cm rise in WTD_e within the range of observations will, on average, decrease CO₂ emissions by around 3 t CO₂ ha⁻¹ yr⁻¹. For CH₄, the non-linear relationship suggests that raising water tables in deep-drained soils to 30 cm below the surface would have a negligible influence on CH₄ emissions (maximum emission 0.3 t CO₂e ha⁻¹ yr⁻¹). Raising WTD_e from 30 to 0 cm would increase CH₄ emissions by approximately 7 t CO₂e ha⁻¹ yr⁻¹. Combining the two GHGs indicates that the change in CO₂ per unit change in WTD_e exceeds the change in CH₄ for all WTD_e > 8 cm. Consequently, any reduction in drainage depths up to that value can be

expected to have a net beneficial impact in terms of climate forcing. Between a WTD_e of 5 and 13 cm the cooling effect of CO_2 sequestration exceeds the warming impact of CH_4 emissions (based on GWP_{100}) implying that peatlands in this range (which is typical of natural systems²¹) will have a small cooling impact on a 100 year time horizon. Over longer time horizons, the longer atmospheric lifetime of CO_2 compared to CH_4 means that this cooling impact will be stronger, and will extend to peatlands with a higher WTD_e ²². However our data do suggest that where average water levels are raised above the peat surface, the net warming effect of higher CH_4 emissions is likely to exceed the cooling effect of additional CO_2 sequestration over shorter time-horizons.

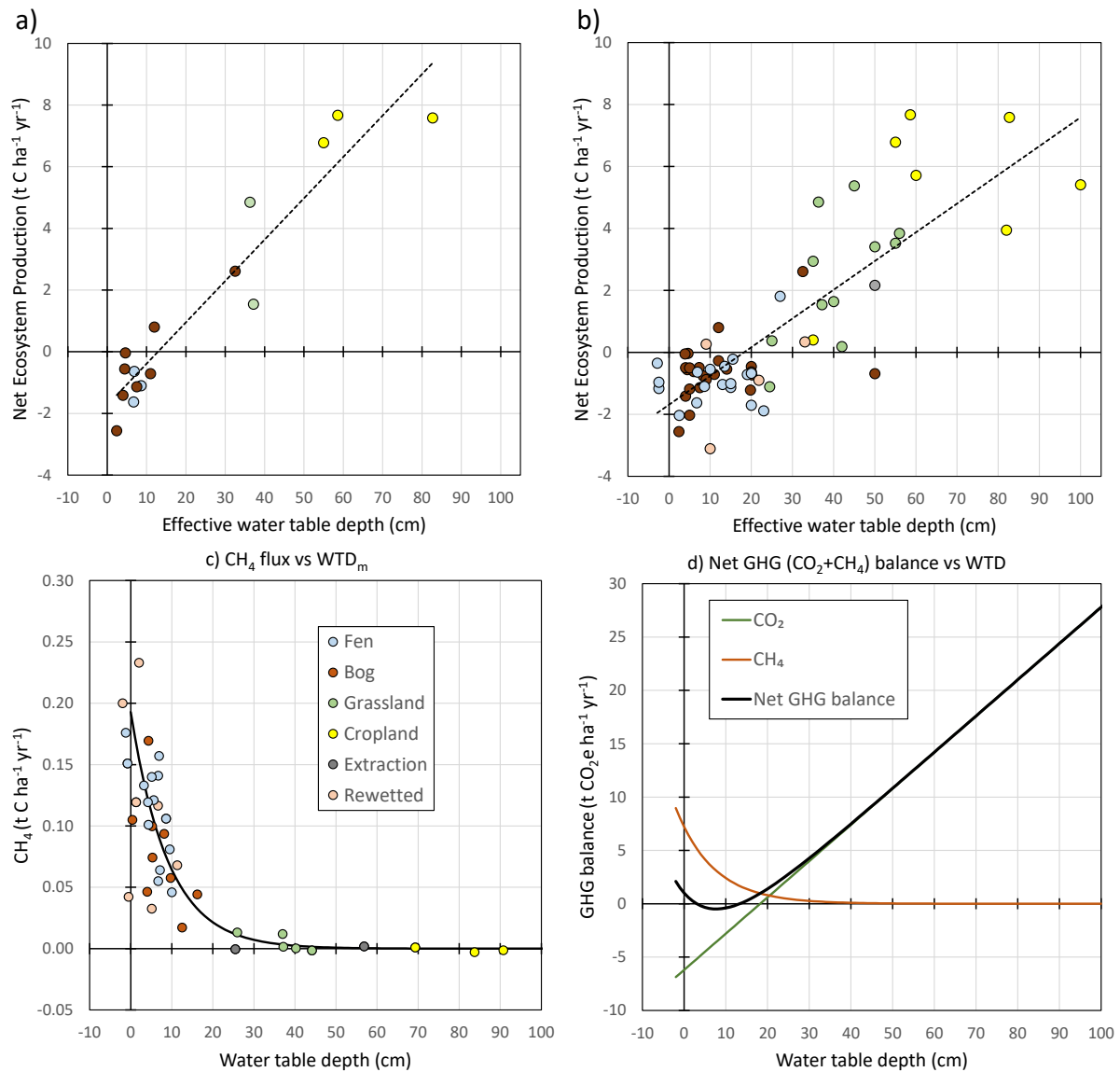


Fig 1 Annual mean flux values versus mean water table depth for: a) CO_2 , based on UK study sites; b) CO_2 based on all published boreal/temperate EC studies in boreal and temperate; c) CH_4 , based on UK and Irish sites; and d) best-fit regressions from (b) and (c) converted to $t CO_2e ha^{-1} yr^{-1}$ and combined to show the net climate impact of CO_2 and CH_4 versus water table depth (based on GWP_{100}).

In the UK, an estimated 64,000 ha of deep peat is under cropland, and 148,000 ha under intensive grassland²³. Based on typical WTD_e values of 90 cm for cropland and 60 cm for intensively managed grassland, we estimate that halving drainage depths across this area would reduce total CO_2 emissions

by 3.1 Mt CO₂ yr⁻¹, with a negligible offsetting increase in CH₄ emissions. This equates to 0.7% of the most recently reported total UK GHG emissions²⁴.

Globally, 250,000-290,000 km² of peat are under drained cropland and grassland, and estimates of CO₂ emissions from these areas, based on IPCC Tier 1 emission factors^{14,25}, range from 680 to 1030 Mt CO₂ yr⁻¹^{3,4,26}. We combined the most recent area estimates⁴ with our observed relationship between NEP and WTD_e (Equation 2) and CH₄ and WTD_e (Equation 3) to derive a revised estimate of net CO₂ + CH₄ emissions from these areas, assuming a global average WTD_e of 50 cm for grassland, and 90 cm for cropland (see Figure 1b, Methods). Emissions from tropical peatlands were scaled according to the ratio of emission factors for tropical and temperate/boreal cropland and grassland in the IPCC 2013 Wetlands Supplement¹⁴; see Methods). Our revised estimate of global CO₂ emissions from drained organic soils under cropland and grassland of 786 Mt CO₂ yr⁻¹ is within the range of previous estimates (note that CH₄ emissions are negligible in this scenario). Unlike previous studies, however, our empirical relationships with drainage depth enable us to evaluate mitigation scenarios based on land-management change, in addition to land-use change. Reapplying Equations 2 and 3 with mean WTDs halved to 45 cm in cropland and 25 cm in grassland gives a global CO₂ + CH₄ emission of 278 Mt CO₂e yr⁻¹, with CH₄ emissions comprising < 5 Mt CO₂e yr⁻¹. The net reduction of 508 Mt CO₂e yr⁻¹ (65% of present-day emissions) under this scenario equates to 11% of total CO₂ emissions from land-use (4.8 Gt CO₂ yr⁻¹⁵), and 1.3% of total global anthropogenic CO₂ emissions (39.6 Gt CO₂ yr⁻¹⁵).

We also evaluated an 'optimal re-wetting' scenario, in which all areas currently under cropland and grassland are re-wetted to a mean WTD_e of 10 cm to maximise their net climate change mitigation potential (Figure 1d). This scenario would generate net CO₂ uptake of 106 Mt CO₂ yr⁻¹, but with an offsetting emission of 91 Mt CO₂ yr⁻¹ of CH₄ emissions, giving a small net GHG removal. Comparing the two scenarios, halving drainage depths in our 'agricultural mitigation' scenario would deliver an estimated 63% of the maximum climate mitigation achievable through optimised re-wetting.

Our analysis does not consider N₂O emissions, which are believed to contribute a further 260 Mt CO₂e yr⁻¹ of emissions from organic soils under cropland and grassland²⁶. However, previous analyses suggest that N₂O emissions are also positively correlated with WTD¹⁶. Similarly, we did not quantify indirect emissions of CO₂ via aquatic organic C export, but again the available evidence suggests that this flux increases following drainage²⁷. Finally, we did not consider the potential influence of raising water levels on GHG emissions from peat fires, which were estimated by the IPCC at 0.2 Gt CO₂e yr⁻¹³. Raising water levels in drained peatlands reduces their susceptibility to deep burns, producing potentially large additional CO₂ emissions mitigation²⁸. On this basis, we consider our estimates of the climate mitigation potential of raising water levels in agriculturally managed peatlands to be conservative.

Table 1. Estimated total CO₂ + CH₄ emissions for global drained cropland and grassland on peat, disaggregated by climate zone, for present day, ‘agricultural mitigation’ and ‘optimal re-wetting’ scenarios, based on the empirical relationships shown in Figure 1 and area data obtained from Leifeld and Menichetti⁴. CH₄ emissions were converted to CO₂ equivalents using a 100 year GWP of 28, and negative values indicate net GHG removal. For details of calculations see Methods, and for emissions data disaggregated by gas and a comparison to previous emissions estimates see Extended Data Tables 4-5.

Peat areas		Mha	
Climate zone	Cropland	Grassland	Total
Boreal	7.2	2.2	9.4
Temperate	3.7	2.6	6.3
Tropical	9.4	3.9	13.3
Total	20.3	8.7	29.0
CO₂ + CH₄ emissions		Mt CO₂e yr⁻¹	
<i>Present day</i>			
Boreal	174.8	24.2	199.0
Temperate	90.2	28.0	118.3
Tropical	395.5	72.8	468.3
Total	660.5	125.1	785.6
<i>"Agricultural mitigation" scenario</i>			
Boreal	65.6	6.2	71.8
Temperate	33.9	7.1	41.0
Tropical	148.0	17.3	165.3
Total	247.5	30.6	278.1
<i>"Optimal re-wetting" scenario</i>			
Boreal	-2.9	-0.9	-3.8
Temperate	-1.5	-2.3	-3.8
Tropical	-3.8	-3.8	-7.6
Total	-8.1	-7.0	-15.1

In the context of the Paris Agreement requirement to achieve net zero GHG emissions by 2050, there is a growing emphasis on ‘negative emission’ strategies to offset continuing fossil fuel emissions, such as the ‘4 per 1000’ objective to raise soil carbon content. While such actions are clearly desirable, we argue that the urgent – and arguably more tractable – challenge of reducing existing emissions from organic soils must not be overlooked. For example, our estimated mitigation potential of 15.3 t CO₂ ha⁻¹ yr⁻¹ for halving drainage depths in cropland on organic soil compares highly favourably to estimated CO₂ sequestration potential of biochar application (0.1 to 1 t CO₂ ha⁻¹ yr⁻¹²⁹), and could thus achieve the same climate benefits within just 0.7-6.5% of the land area.

Our results present both a challenge and an opportunity. Current agricultural practices on peat rely almost entirely on crops derived from dryland ecosystems, which require drainage. Although efforts are now being made to develop wetland-adapted (‘paludiculture’) crops, these do not yet provide commercially viable large-scale alternatives to conventional agriculture³⁰. Intensifying global demands for land and food mean it may be unrealistic to expect all agricultural peatlands to be rewetted and restored to their original wetland function in the near future, particularly in countries where large populations rely on peatlands for their livelihoods. Our results indicate that the development of locally appropriate mitigation measures within agricultural peatlands could deliver substantial emissions reductions. Our results *do not* argue against full restoration of cultivated peatlands to wetlands where

this is achievable; our optimal re-wetting scenario effectively halts GHG emissions from these areas, and turns them into small net GHG sinks. Wetland restoration would also have unquestioned benefits for biodiversity, and (unlike partial raising of water tables under agriculture) can be considered truly sustainable. However, where societal need for drainage-based agriculture precludes full re-wetting, opportunities remain to achieve globally significant emissions reductions. In either case, water levels need to be raised rapidly to avoid further sustained radiative forcing due to peatland oxidation³¹, and to achieve net zero emissions on the timescale of the Paris Agreement. The development of water-tolerant, economically viable crops suitable for cultivation on organic soils should therefore be a high priority for international efforts to combat climate change.

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Author contributions

CDE conceived and led the study, undertook the global upscaling, and drafted the paper. RM led the analysis of eddy covariance data. MP and SEP supported to the global flux data synthesis, and PL undertook additional statistical analysis. All co-authors contributed to data collection, analysis and interpretation, and commented on the draft manuscript.

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METHODS

UK CO₂ flux data collection

We included new CO₂ flux data collected using the eddy covariance (EC) method at thirteen peatland sites in the UK, comprising four blanket bogs, one raised bog, three conservation-managed fens, two

grasslands and three croplands³¹⁻³⁴ (Extended Data Figure 1, Extended Data Table 1). The dataset was augmented with previously published results from two further blanket bog sites in Northern Scotland³⁵ and one in Ireland³⁶. This density of flux towers on peat within one region is believed to be unique. Peat depth at the measurement sites ranged from 0.4 m to 6 m.

All flux tower sites had at least one full year of measurements, up to a maximum of six years. All sites had fast response gas analysers which measure the atmospheric mass density of CO₂ and water vapour. Open-path analysers (either LI-7500 or LI-7500A, LI-COR Biosciences, Lincoln, Nebraska, US) were deployed across the majority of sites. (most sites), An LI-7200 enclosed-path analyser was used at the Cairngorms site, and a closed path analyser (LI-7000) was deployed at Auchencorth Moss. Simultaneous measurements of the three vector components of atmospheric turbulence were made using either a Campbell Scientific CSAT3 (Logan, Utah, USA) or a Gill Instruments Ltd. (Lymington, UK) sonic anemometer (model: Solent R3, HS, WindMaster or WindMaster PRO). Fast (20 Hz) data were logged on a CR3000 Measurement and Control System (Campbell Scientific Inc. Logan, Utah, USA) or a LI7550 data logger.

All EC data were processed using EddyPRO[®] flux calculation software (LI-COR Biosciences, Lincoln, Nebraska, USA) following widely adopted flux calculation and correction protocols to ensure consistency across sites. All data were quality checked using standardised tests for outlier removal³⁷, technical quality³⁸ and spatial representativeness³⁹. Measured net ecosystem CO₂ exchange (NEE) was gap-filled and partitioned into estimates of gross primary production (GPP) and total ecosystem respiration (ER) using standardised methods of the global Fluxnet community^{40,41}. Uncertainties in daily and annual CO₂ fluxes were calculated for the EC data based on random sampling errors³⁷ and uncertainties introduced by data gap-filling^{40,41}. Random uncertainties for measured CO₂ flux densities were estimated as standard deviations based on the method of Finkelstein & Simms⁴². Uncertainty introduced by data gap-filling was estimated as the standard deviation of the values that were averaged to fill data gaps⁴¹. Uncertainties in time integrated CO₂ fluxes were calculated as the square root sum of square of the thirty-minute uncertainty estimates for measured and gap-filled fluxes³². Energy balance closure (a check of the performance of an EC system at a measurement site) ranged from 60 to 96 % (mean slope = 0.8 ± 0.13 , $n = 11$ sites) of the independently measured available energy at sites where all terms of the surface energy balance were monitored.

Water-table depths were measured manually at all sites using PVC dipwells using a dip-meter, and then corrected to the distance from the peat surface (data were recorded as positive distance below the surface, with negative values indicating ponding above the peat surface). At most sites a calibrated water-level sensor (pressure transducer) and integrated logger was also used to record water table depth in at least one dipwell, at a minimum one-hour frequency. Peat depth was measured at multiple locations around each site by probing. For each site, we calculated 'effective' mean water table depth as whichever was the smallest out of the mean annual measured water table depth, and measured peat depth. In other words, at sites where the water table fell below the base of the peat, we considered that the depth of mean provided the most appropriate estimate of the depth of organic matter exposed to aerobic decomposition. For sites under agricultural management, C removals in harvested biomass were quantified through a combination of direct (dry weight) measurements of harvested biomass supported by information from farm managers on crop yields, and literature data or information from comparable sites where necessary. For the purposes of deriving a site CO₂ balance, we assumed that all harvested biomass would be converted to CO₂ within the same year.

Global CO₂ flux data collation

To place our results into a broader context, we collated a large dataset of published CO₂ balance data obtained from eddy covariance (EC) measurements on peatlands in the temperate and boreal climate

zones⁴³⁻⁹² (Extended Data Figure 4). We restricted the analysis to EC studies, on the basis that these provide the best and most methodologically consistent measurements of CO₂ balance, and should be relatively comparable. Sites were included if they provided sufficient data to derive a direct (gaseous) annual CO₂ balance, and an estimate of mean annual water-table depth (WTD). At natural sites with more complex (e.g. hummock-hollow) topography we expressed WTD relative to the mean surface elevation rather than (as in some studies) the hummock tops. As for own study sites, we defined effective WTD (WTD_e) as whichever was the smallest of WTD and the reported peat depth. For agriculturally managed sites, we only included studies that provided sufficient information on estimated C offtake in harvested biomass. Plantation forest sites were not included, because no studies reported C balance over a full forest rotation (and thus are not in steady state with regard to biomass uptake and removal). However, naturally treed peatlands were considered to be at steady state (with regard to biomass C) and were therefore included. For eight boreal/subarctic sites which reported fluxes for the growing season only, we assigned winter fluxes using the authors' own estimates where provided (3 sites), and for the remaining sites we applied an average winter flux estimate of 1.5 g C m⁻² month⁻¹ based on the mean of measured values from five comparable sites where winter EC data were collected.

We excluded a small number of sites where average water-table depths were more than 5 cm above the ground surface as a result of excessive re-wetting or rice cultivation⁸². Sites that had been subject to recent land-use change (e.g. forest removal³⁵) or fire⁸⁴ were also omitted, due to the contribution of decaying biomass to CO₂ emissions⁹³. In total, we extracted data from 49 published temperate/boreal EC studies (Extended Data Table 2) giving a total dataset of 65 eddy covariance studies. We also obtained data from three published tropical peatland flux tower studies^{94,95}, which are insufficient to support a separate analysis, but included in Extended Data Table 2 for reference.

CH₄ flux data collection and collation

We analysed new CH₄ flux data, measured using static chambers, from 31 UK locations within 11 discrete peatland areas of England and Wales, most of which also had at least one flux tower for CO₂ measurement as described above. Additional data spanning a range of vegetation and WTD are two of the sites were provided by Ref 96. Multiple measurements locations within individual sites were selected in order to capture local variations in vegetation, management and water table depth. As for the CO₂ flux data analysis, re-wetted sites that were frequently or continuously inundated during the study period were excluded from the analysis.

Methane fluxes were measured at multiple (usually six) locations, following standard static chamber methods^{97,98}. Chambers comprised transparent 3 mm thick acrylic sheet with a height of 50 cm, and a base of 60 x 60 cm. During measurements these were attached to 20-30 cm deep collars inserted 10 cm into the soil, which were left permanently in place wherever possible to minimise disturbance (at arable sites it was necessary to remove and re-install the collars between measurements to avoid them being damaged during routine farming activities). For taller vegetation, stackable extensions were used to increase chamber height. Chambers incorporated fans to increase air circulation within the chamber, and a double gas bag arrangement to allow air pressure equilibration.

At the majority of sites, manual chamber measurements were made 12-16 times per year (with a higher frequency during the growing season and a reduced frequency during winter), for the duration of the study at each site (typically three years, minimum one year). Measurements were mostly made between 10:00 and 16:00. On each measurement occasion, sequential chamber tests lasting 3-5 minutes each were made using dark chambers. At most sites a Los Gatos Ultra-Portable Greenhouse Gas Analyser (Model 915-0011; Los Gatos Research, Mountain View, California) was used to measure within-chamber CH₄ concentrations, with a 5 second measurement frequency. At one site where a Los Gatos analyser was unavailable, CH₄ fluxes were measured by collecting five sequential chamber gas

samples over a 21 minute period for analysis using gas chromatography. Finally, at one set of grassland sites at Cors Fochno, CH₄ fluxes were measured using a set of 14 LI-COR LI-8100A autochambers connected to a Los Gatos analyser via a LI-COR LI-8150 multiplexer unit. These smaller chambers were installed over 21 cm circular collars (other specifications as above) along a 30 m transect at each site, and flux measurements were made at 1.5 hour intervals during monthly deployments of up to 48 hours. At all sites, air pressure and soil temperature were measured at the start of each chamber test, and chamber headspace temperature and humidity were logged every 10 seconds.

Sequential CH₄ concentration data from each chamber test were converted to fluxes per unit ground surface area by linear regression against time using a standardised method, taking account of the chamber dimensions. Individual datasets were analysed visually to identify periods of linear change in concentration over time, and to exclude anomalous readings (for example associated with CH₄ ebullition events) based on pre-defined protocols. At the manually sampled chamber site it was generally not possible to derive reliable empirical models of instantaneous flux versus measured meteorological or environmental variables, and here we estimated annual means from the mean of all measurements taken within each calendar month⁹⁹. At the autochamber sites the higher temporal resolution of data allowed us to model annual CH₄ fluxes based on a non-linear regression versus soil temperature. Note that the flux measurement method is likely to have captured steady ebullition fluxes¹⁰⁰ but may have omitted some episodic ebullition at very wet sites¹⁰¹. For full details of static chamber design, operation and data processing, see Evans et al.³¹.

We augmented our primary dataset of CH₄ flux measurements with published values obtained from similar locations within the UK and Ireland^{93,101-108}, using comparable measurement methods (static chamber measurements using fast CH₄ analysers or analysis of vial samples on a gas chromatograph). We took this approach with the aim of elucidating the influence of key land-management drivers (notably drainage depth) while minimising the influence of intrinsic factors linked to climate, peat type and inter-regional differences in vegetation, as well as potential methodological differences. The full dataset is shown in Extended Data Table 3. We also evaluated the wider applicability of the relationships obtained by comparison to three independent data syntheses from the UK¹⁰⁹, continental Europe¹⁶, and a set of North American and Fennoscandian sites²¹ (Extended Data Figure 3)

Flux data analysis

We assessed the relationship between NEP and WTD_e using simple linear regression, incorporating all site types. We also tested for non-linearity using polynomial regression, and for the possible effects of site type and additional climate variables using a linear mixed effect model; for the results of this extended data analysis see Supplementary Information. For CH₄, which clearly showed a non-linear relationship with WTD_e, an exponential ‘half-life’ model was fitted with the form:

$$FCH_4 = A \times 0.5^{\frac{WTD_e+5}{B}} \quad (\text{Equation 4})$$

Where FCH_4 is the mean measured flux expressed in t C ha⁻¹ yr⁻¹, WTD_e is mean effective water table depth in cm (5 cm was added to all values, so that sites with WTD_e slightly above the surface that were included in the analysis had positive values in the equation), and A and B are fitted constants: A is the value of FCH_4 when WTD is 5 cm above the peat surface, and B is the change in WTD_e over which FCH_4 halves in magnitude. Equation 4 was fitted to the dataset by sequentially iterating B to maximise the R² of a regression through the origin between predicted and observed FCH_4 , and then adjusting A until a gradient of 1 on this relationship was obtained.

Estimates of UK and global peat emissions and mitigation potential

We obtained global estimates of grassland and cropland extent on drained organic soils, and associated GHG emissions, from two sources: the FAOSTAT website of the Food and Agriculture Organisation of the United Nations²⁵, and the analysis of Leifeld and Menichetti⁴. The estimates were derived using different data sources, aggregated differently by climate zone, and based on the IPCC's 2006²⁴ and 2013¹⁴ emissions factors for drained organic soils respectively. FAOSTAT data were also used in the IPCC 5th assessment report (AR5)³. For our analysis we took the cropland and grassland area estimates by climate zone provided by Leifeld and Menichetti⁴, and derived estimates of total CO₂ and CH₄ emissions based on Equations 2 and 3, for three illustrative water table scenarios. For the 'present day' scenario we assigned mean WTD_e values of 90 cm for cropland and 50 cm for grassland, based on the observations for these land-use classes in our collated flux dataset (Supplementary Tables 1 and 2). For boreal and temperate regions we applied Equation 3 directly.

For tropical peatlands, direct application of Equation 3 would likely produce an underestimate of emissions, because respiration rates for an equivalent drainage depth are expected to be faster in the higher temperatures experienced by tropical peatlands¹⁴. While there are currently insufficient EC data to derive an empirical NEP-WTD relationship for tropical peatlands, long-term measurements of peat subsidence suggest that peat loss in drained tropical peatlands is linearly related to WTD, as it is in high-latitude peatlands, but with a steeper response^{10,12}. On this basis, we derived indicative estimates of CO₂ emissions from agriculturally drained tropical peatlands by applying Equation 3 as above, and then scaling up the resulting CO₂ flux based on the ratio of Tier 1 EFs for tropical and boreal/temperate croplands and grasslands according to the most recent values presented in the IPCC Wetlands Supplement¹⁴. The resulting scale factors were 1.77 and 1.68 for cropland and grassland respectively. We applied the same WTD_e values to tropical cropland and grassland as for high-latitude peatlands.

For CH₄, we applied the relationship between emissions and WTD_e derived from our dataset (Equation 3), after checking for consistency with data from previous regional and global syntheses (Extended Data Figure 4). To compare the relative warming impact of CH₄ and CO₂ emissions, we used the most recent IPCC AR5 100 year Global Warming Potential (GWP) of 28, which excludes carbon-climate feedbacks.

Finally, to calculate the emissions mitigation potential of raising water levels in agriculturally managed peatlands, we re-applied Equations 3 and 4 to the FAOSTAT dataset, with WTD_e halved to 45 cm under cropland and 25 cm under grassland. This adjustment is arbitrary, and intended to illustrate the magnitude of emissions mitigation that could be attained if economically viable methods of managing cropland and grassland systems at high water levels could be implemented.

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