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Authors(s)	Wilson, David, Mackin, Francis, Tuovinen, Juha-Pekka, Moser, Gerald, Farrell, Catherine A., Renou-Wilson, Florence
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RESEARCH ARTICLE

Carbon and climate implications of rewetting a raised bog in Ireland

David Wilson¹  | Francis Mackin²  | Juha-Pekka Tuovinen³  | Gerald Moser⁴  | Catherine Farrell⁵  | Florence Renou-Wilson⁶ 

¹Earthy Matters Environmental Consultants, Donegal, Ireland

²RPS Group, Belfast, Ireland

³Finnish Meteorological Institute, Helsinki, Finland

⁴Institute for Plant Ecology, Justus Liebig University Giessen, Rostock, Germany

⁵Ecologist, Mayo, Ireland

⁶School of Biology and Environmental Science, University College Dublin, Dublin, Ireland

Correspondence

Florence Renou-Wilson, School of Biology and Environmental Science, University College Dublin, Dublin, Ireland.
Email: florence.renou@ucd.ie

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Abstract

Peatland rewetting has been proposed as a vital climate change mitigation tool to reduce greenhouse gas emissions and to generate suitable conditions for the return of carbon (C) sequestration. In this study, we present annual C balances for a 5-year period at a rewetted peatland in Ireland (rewetted at the start of the study) and compare the results with an adjacent drained area (represents business-as-usual). Hydrological modelling of the 230-hectare site was carried out to determine the likely ecotopes (vegetation communities) that will develop post-rewetting and was used to inform a radiative forcing modelling exercise to determine the climate impacts of rewetting this peatland under five high-priority scenarios (SSP1-1.9, SS1-2.6, SSP2-4.5, SSP3-7.0 and SSP5-8.5). The drained area (marginal ecotope) was a net C source throughout the study and emitted $157 \pm 25.5 \text{ g C m}^{-2} \text{ year}^{-1}$. In contrast, the rewetted area (sub-central ecotope) was a net C sink of $78.0 \pm 37.6 \text{ g C m}^{-2} \text{ year}^{-1}$, despite relatively large annual methane emissions post-rewetting (average $19.3 \pm 5.2 \text{ g C m}^{-2} \text{ year}^{-1}$). Hydrological modelling predicted the development of three key ecotopes at the site, with the sub-central ecotope predicted to cover 24% of the site, the sub-marginal predicted to cover 59% and the marginal predicted to cover 16%. Using these areal estimates, our radiative forcing modelling projects that under the SSP1-1.9 scenario, the site will have a warming effect on the climate until 2085 but will then have a strong cooling impact. In contrast, our modelling exercise shows that the site will never have a cooling impact under the SSP5-8.5 scenario. Our results confirm the importance of rapid rewetting of drained peatland sites to (a) achieve strong C emissions reductions, (b) establish optimal conditions for C sequestration and (c) set the site on a climate cooling trajectory.

KEYWORDS

carbon dioxide, climate mitigation, methane, peat, radiative forcing, rewetting

1 | INTRODUCTION

The premise that rewetting of drained or degraded peatlands can assist in climate change mitigation has gained considerable traction

in recent years (Günther et al., 2020; Parish et al., 2008; Smith et al., 2014), underpinned by the growing body of research where the main components of the peatland carbon (C) cycle: carbon dioxide (CO₂), methane (CH₄), dissolved organic carbon (DOC) have been

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quantified in a wide range of peatland types along a drainage continuum (natural-drained-rewetted) (e.g., Koehler et al., 2011; Nugent et al., 2018; Renou-Wilson et al., 2016; Tiemeyer et al., 2016). Furthermore, the key environmental factors (both abiotic and biotic) that drive these fluxes at the biosphere–atmosphere interface have also been well established (e.g., Abdalla et al., 2016; Blodau, 2002; Evans et al., 2016).

Natural (non-degraded) peatlands act as long-term C sinks (Koehler et al., 2011; Nilsson et al., 2008) and have played a vital role in regulating the global climate during the Holocene (Frolking & Roulet, 2007). Within these peatland sites, the position of the water table has frequently been identified as one of the key environmental controls that determine the amount of C that enters/exits the peatland (Chimner et al., 2016; Peichl et al., 2014; Qassim et al., 2014), with the optimum C balance likely to occur when the water table is close to the peat surface (Couwenberg et al., 2011; Jurasinski et al., 2016).

Drainage results in a radical transformation in peatland C exchange; former net C sinks (for millennia) can rapidly switch to net C sources (e.g., Wilson, Blain, et al., 2016). Globally, around 15% of peatlands have been drained; either for agriculture, forestry or peat extraction (Barthelmes, 2016). When peatlands are drained for peat extraction, they release substantial quantities of CO₂ from the highly oxidized carbon-rich environment to the atmosphere (Rankin et al., 2018; Salm et al., 2012; Wilson et al., 2015). Moreover, while CH₄ emissions tend to be relatively low from drained peat extraction fields (Rankin et al., 2018; Renou-Wilson et al., 2019) as a consequence of the extended oxic zone above the water table, substantial emissions from ditches or drains can continue (Drösler et al., 2014; Peacock et al., 2021). Furthermore, losses of DOC from peatland catchments may also increase strongly after drainage (Pickard et al., 2017; Strack & Zuback, 2013) depending on the density of the drain network (Worrall et al., 2007), with most of the DOC subsequently converted to CO₂ in the ensuing water bodies (Dinsmore et al., 2010; Evans et al., 2016).

Rewetting aims to reverse the hydrological changes brought about by drainage (Ahmad et al., 2020) and to establish the requisite conditions for the establishment of typical peatland plant communities (Richert et al., 2000; Tuittila et al., 2000). In some cases, rewetting may involve the blocking of drains only, while in others, more extensive management actions may be required (e.g., Anderson & Peace, 2017; Farrell & Doyle, 2003). An eclectic range of factors affect C dynamics post-rewetting, and include previous and current land use, the residual depth of the peat, nutrient status and vegetation composition (Wilson, Blain, et al., 2016). Consequently, the time elapsed until the C sink function is restored (if at all) at rewetted sites is ill-defined (University of Leeds Peat Club et al., 2017), and indeed may be of less importance than the immediate reduction in C emissions from the site. Research thus far has indicated a strongly site-specific response to rewetting (Renou-Wilson et al., 2019). For example, it has been demonstrated that rewetting of former peat extraction sites leads to a sizeable reduction in CO₂ emissions from bare peat surfaces (Wilson, Farrell, et al., 2016) and

vegetated areas (Järveoja, Peichl, Maddison, Soosaar, et al., 2016; Strack & Zuback, 2013), and in some cases, rewetting may result in the return of the CO₂ sink function (Nugent et al., 2018; Wilson, Farrell, et al., 2016) but not all sites show this trend (Järveoja, Peichl, Maddison, Soosaar, et al., 2016; Renou-Wilson et al., 2019; Wilson, Tuittila, et al., 2007). In addition, CH₄ emissions typically increase with rewetting (Minke et al., 2016; Strack et al., 2014; Wilson, Farrell, et al., 2016) and may remain persistently elevated in some sites decades after the blocking of drains and the implementation of other restoration management actions (Vanselow-Algan et al., 2015; Wilson et al., 2009). The magnitude of DOC losses following rewetting is ambiguous as some studies report a strong decrease (e.g., Evans et al., 2016) while others show no effect (e.g., Peacock et al., 2018).

Recent radiative forcing exercises have demonstrated that prompt peatland rewetting is paramount to prevent long-term warming effects from degraded peatland sites (Günther et al., 2020; Huth et al., 2021). Yet, despite their perceived importance for climate change mitigation efforts (Leifeld & Menichetti, 2018; Sirin et al., 2021), multi-year GHG studies (>3years) in rewetted peatlands remain scarce to date (Nugent et al., 2018; Wilson, Farrell, et al., 2016), which hinders the planning of post-rewetting peatland management.

As Ireland is home for around 50% of the near-natural oceanic raised bogs left in the Europe, there is an impetus to restore this natural resource. In order to reverse the negative trends in conservation value, habitat integrity and GHG emissions, Ireland has now committed to the rewetting/restoration of bogs within the Natura 2000 network, which comprises some 35,000ha of raised bogs, and includes approximately 1000ha of drained bogs west of the River Shannon with the potential to be restored or have been restored.

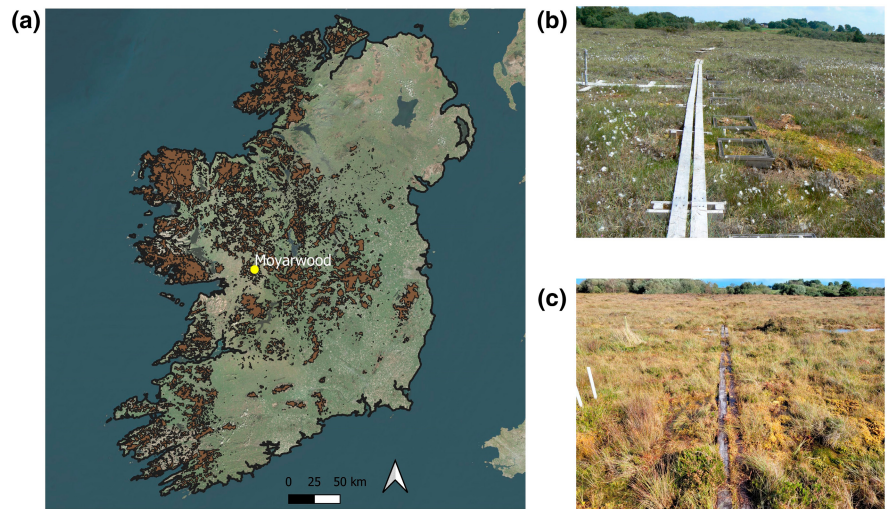
In this study, we quantified CO₂, CH₄ and nitrous oxide (N₂O) fluxes for a 5-year period in a former raised bog in the west of Ireland, immediately after the site was rewetted. We quantified and modelled fluxes from drained areas on the margins (marginal ecotope), and from a rewetted area with a persistently high-water table (sub-central ecotope) to provide the basis for an estimate of the net carbon ecosystem balance (NECB) of the two ecotopes. Hydrological modelling was then performed at the site-level to predict the future development of ecotopes at the site and to inform a radiative forcing model that was used to compare future shared socio-economic pathway (SSP) scenarios at the site.

2 | MATERIALS AND METHODS

2.1 | Site description

The study site is located at a raised bog in Moyarwood, Co. Galway, Ireland (Figure 1a). The site covers approximately 230ha and has been subject to mechanical peat extraction (for domestic use) on the margins for decades. It was extensively drained (ditches located

FIGURE 1 (a) Location of the study site at Moyarwood, Co. Galway. Brown areas indicate the location of peat soils in the Republic of Ireland. Map source: Hammond (1981). (b) Boardwalk and flux collars in the rewetted area in July 2013, and (c) same area in 2020.



every 15 m) in the 1980s in preparation for milled peat extraction. However, the site was never subsequently developed for peat extraction and a vegetation cover remained in situ between the ditches (Figure 1b,c). The ditches remained active until a rewetting programme commenced in 2012, which involved blocking of the ditches with peat dams at regular intervals (generally at any point where there was a fall in the ditch level of 10 cm). A site description can be found in Table 1.

Potential evapotranspiration varied considerably at the site during the 5-year study period (Figure 2a) with the highest annual value recorded in year 1 (528 mm) and the lowest in year 5 (491 mm). Rainfall also varied considerably throughout the study (Figure 2b) with the highest value recorded in year 3 (1541 mm) and the lowest in year 4 (1000 mm). Global radiation was relatively similar between years, although differences between years were observed from late summer (Figure 2c).

2.2 | Site instrumentation

In January/February 2013, a total of 15 stainless steel collars (60×60 cm) were established at the site along a transect perpendicular to the drainage ditches: 12 collars were located in a rewetted area (with one collar located in a former ditch), and three were located on the drained eastern margins of the site, where drainage ditches were still active during the study period. The ditch collar was placed in a similar manner to the other collars so that the channel at the top of the collar was above the water level and supported on wooden batons that extended across the ditch. No infilling (by vegetation) occurred in this ditch throughout the study period. All collars were inserted 30 cm into the peat.

Perforated plastic dipwells (3-cm internal diameter) were inserted (to a depth of approx. 100 cm) into the peat adjacent to each collar to facilitate the measurement of water table levels during each flux measurement campaign. Wooden boardwalks were established in both areas to prevent damage to the vegetation and to avoid compression

TABLE 1 Site description and meteorological data for the Moyarwood site. Air temperature and precipitation data (1981–2010) from Met Éireann Athenry meteorology station (Lat. 53°17'21" N; long. 8°47'08" W)

Site grid reference	Lat. 53°20'49.2" N; long. 8°30'55.4" W
Mean air temperature	9.9°C
Annual rainfall	1193 mm
Prevailing wind direction	South-westerly
Land use	Former raised bog, drained in 1980s in preparation for milled peat extraction. Rewetted in 2012/2013. Peat extraction on the margins.
Dominant vegetation at end of the study	Drained areas: <i>Calluna vulgaris</i> , <i>Carex panicea</i> , <i>Cladonia portentosa</i> , <i>Narthecium ossifragum</i> , <i>Trichophorum cespitosum</i> , <i>Erica cinerea</i> , <i>E. tetralix</i> Rewetted: <i>Sphagnum capillifolium</i> , <i>S. cuspidatum</i> , <i>S. magellanicum</i> , <i>S. papillosum</i> , <i>Eriophorum vaginatum</i> , <i>E. angustifolium</i> , <i>Rhynchospora alba</i> , <i>C. panicea</i> , <i>N. ossifragum</i>
Altitude	98 m asl
Area	230 ha
Mean peat depth	440 cm
Peat type	Sphagnum
Von Post Humification Scale	5–7 (0–25 cm depth)
Parent material	Limestone
Mean electrical conductivity	102 $\mu\text{S cm}^{-1}$
Bulk density	
Drained	0.13 g cm^{-3}
Rewetted	0.08 g cm^{-3}
pH	4.4
C (%)	51.5
N (%)	1.32

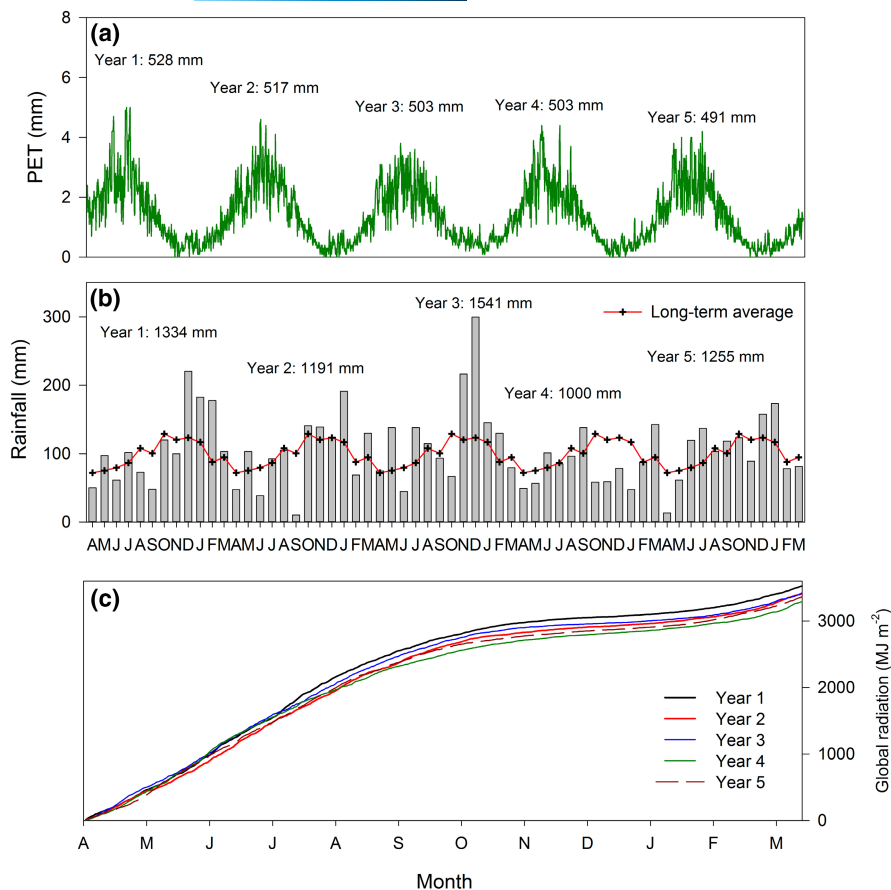


FIGURE 2 Weather variables at the Moyarwood study site from April 2013 to March 2018 (years 1–5). (a) Daily potential evapotranspiration (PET; mm), (b) monthly rainfall (mm), and (c) cumulative global radiation (MJ m^{-2}). Long-term average rainfall data shown for 1980–2010. Annual PET and rainfall sums for each 12-month period (April 1–March 31) are shown above the data. Data from Met Éireann Athenry meteorology station (Lat. $53^{\circ}17'21''\text{N}$; long. $8^{\circ}47'08''\text{W}$).

of the peat during GHG sampling. At the drained sub-site, a Watchdog weather station (Watch Dog Model 2400, Spectrum Technologies Inc., IL, USA) was established and recorded photosynthetic photon flux density (PPFD; $\mu\text{mol m}^{-2} \text{s}^{-1}$), soil temperatures ($^{\circ}\text{C}$; 5 and 10 cm depths) and soil moisture content at 10-min intervals. A second soil logger (Hobo External Data Loggers, Onset Computer Corporation, MA, USA) was installed in the rewetted site and recorded soil temperatures at 5 and 10 cm depths at hourly intervals.

2.3 | Field measurements

GHG flux measurements commenced in April 2013 and ended in March 2018. Chambers were employed at fortnightly/monthly intervals (depending on the season) during this period: each site visit consisted of a 2–4 day campaign during which CO_2 , CH_4 and N_2O fluxes were sampled simultaneously with a range of environmental variables: PPFD, soil temperature, water table level and green area index (GAI). For net ecosystem exchange (NEE) sampling, a transparent chamber ($60 \times 60 \text{ cm}$) was connected to an infrared gas analyser (EGM-4, PP Systems, UK). The chamber was also equipped with a cooling system (to prevent excessive temperature build-up) and internal fans (to ensure uniform circulation of air within the chamber). Vent holes on the chambers ensured that pressure artefacts were minimized during chamber placement. The transparent chamber was placed in the water channel of a collar and CO_2 concentration, PPFD

and temperature within the chamber headspace were recorded every 15-s over a 60–180 s period. The measurement was rejected if PPFD levels deviated by $>10\%$ or if the chamber temperature increased by $>2^{\circ}\text{C}$ during the enclosure time. After each measurement, the chamber was removed from the collar to allow the CO_2 concentration to reach equilibrium with the ambient air. NEE was measured under a range of light levels as the position of the sun changed throughout the day. In early mornings, an artificial shroud that blocked approximately 50% of incoming PPFD was placed over the chamber to permit the measurement of NEE at low PPFD levels ($<100 \mu\text{mol m}^{-2} \text{s}^{-1}$). Measurements were carried out between 8 am and 6 pm in the summer and between 9 a.m. and 3 p.m. in the winter to ensure that the maximum PPFD was reached at each measurement date. Ecosystem respiration (Reco) was measured by covering the chamber in an opaque cover and CO_2 exchange was measured as described above.

Methane (CH_4) and nitrous oxide (N_2O) fluxes were measured at monthly intervals (multiple measurements were carried out during the 4-day measurement campaign in summer) using an opaque, polycarbonate chamber ($60 \times 60 \times 25 \text{ cm}$) equipped with a battery-operated fan that mixed the air within the chamber headspace. Four 50 ml samples were withdrawn into 60 ml polypropylene syringes from the chamber headspace at 5-min intervals over a 20-min period (the measurement period was increased to 40 min during winter-time when low fluxes were expected) and the samples transferred to Exetainer® vials (12-ml Soda Glass Vials; Labco, UK) in the field

and analysed with a gas chromatograph (Bruker Greenhouse Gas Analyser 450-GC) equipped with a flame ionization detector (FID) and an electron capture detector (ECD). The detector temperatures were set at 300°C (FID) and 350°C (ECD), and five CH₄ and N₂O standards were supplied by Deuste Steining GmbH. Gas peaks were integrated using Galaxie software (Varian Inc., Palo Alto, CA, USA). During each measurement in the field, air temperature inside the chamber, soil temperature (at 5 and 10 cm depths) and water table were recorded at each collar.

To incorporate the seasonal dynamics of the plants into CO₂ exchange models, a green area index (GAI) was estimated for each of the vegetated collars. This involved measuring the green photosynthetic area of all vascular plants (leaves and stems) within the sample plot at monthly intervals. Moss % cover was estimated at the same time. Species-specific model curves were applied to describe the phenological dynamics of the vegetation of each collar, and the models (vascular plants and moss) were summed to produce a plot-specific GAI. For a more detailed description of the method see Wilson, Alm, et al. (2007).

2.4 | Flux calculations

Flux rates (mg CO₂ m⁻² h⁻¹; mg CH₄ m⁻² h⁻¹; µg N₂O m⁻² h⁻¹) were calculated as the linear slope of the CO₂, CH₄ and N₂O concentrations in the chamber headspace over time, with respect to the chamber volume, collar area and air temperature. A flux was accepted if the coefficient of determination (r^2) was at least 0.90. An exception was made in cases where the flux was close to zero, for example, in early morning/late evening when there are light constraints on photosynthetic activity or in winter time when soil processes are typically slower and the r^2 is always low (Alm et al., 2007). In these cases, the flux data were examined graphically and fluxes with obvious nonlinearity were discarded. The remainder were evaluated using Akaike's Information Criterion for small sample sizes (AICc), and fluxes that were associated with low AICc values (representing lower variance and better model fitting) were accepted.

In this study, we follow the atmospheric sign convention whereby positive values indicate a flux from the biosphere to the atmosphere (source) and negative values indicate a flux from the atmosphere to the biosphere (sink). Gross primary production (GPP) was calculated as NEE minus Reco (Alm et al., 2007). The closest Reco flux value in time to a NEE flux value was used, with care taken to ensure that air (within the chamber) and soil temperatures were similar at the time of measurement.

2.5 | Modelling

Statistical and physiological response models (Alm et al., 2007) were constructed and parameterized for each sample plot. Model evaluation was based on statistically significant model parameters ($p < .05$), the lowest possible standard error of the model

parameters and the greatest possible adjusted r^2 value (Laine et al., 2009). The relationship between Reco, GPP or CH₄ and a range of independent environmental variables (recorded in conjunction with flux measurements) was tested during model construction. Only variables that increased the explanatory power of the model were included. The models were accepted if the residuals were evenly scattered around zero. GPP was related to PPFD using the Michaelis-Menten-type relationship that describes the saturating response of photosynthesis to PPFD (Tuittila et al., 1999). The GPP model coefficients and associated standard errors were estimated using the Levenberg-Marquardt multiple nonlinear regression technique (IBM SPSS Statistics for Windows, version 21.0, Armonk, NY, USA). The Reco models were based upon the Arrhenius equation (Lloyd & Taylor, 1994) and were nonlinear models related to soil temperature (Supplementary Information 1). The CH₄ models were nonlinear models related to soil temperature (Supplementary Information 1).

2.6 | Annual NECB

Hourly fluxes were reconstructed for each sample plot in combination with an hourly time series of (1) soil temperature, (2) PPFD values recorded by the weather station, and (3) plot-specific modelled GAI that described the phenological development of the vegetation (only for collars where a statistically significant relationship between GAI and gas fluxes was observed). Annual NEE (g C m⁻² year⁻¹) was calculated for each sample plot by integrating the hourly modelled GPP (negative values) and Reco (positive values) values over each 12-month period. Annual CO₂ balances from the two study areas for the first 2 years were previously reported in and Renou-Wilson et al. (2019) and Wilson et al. (2015), and here we provide three additional years of data. The net ecosystem carbon balance (NECB) was derived from the terrestrial GHG (CO₂ and CH₄) sources and sinks and by adding fluvial C fluxes in the form of DOC (data from Regan et al., 2020). Dissolved inorganic carbon and particulate organic carbon were not included.

2.7 | Statistical analysis

Statistical analyses were performed using SPSS version 21.0 for Windows (IBM SPSS Statistics for Windows, Armonk, NY, USA). p values $< .05$ were considered statistically significant. All data were tested for normality using the Kolmogorov-Smirnov test. Where the data were not normally distributed, the repeated-measures Friedman and the Wilcoxon signed-rank test nonparametric tests were used. Uncertainty in reconstructed annual Reco and NEE was calculated by summing up the maximum and minimum standard errors associated with each of the model parameters (Drösler, 2005; Elsgaard et al., 2012; Renou-Wilson et al., 2014). Uncertainty in the NECB estimate was calculated following the law of error propagation as the square root of the sum of the squared standard errors of GPP and Reco.

2.8 | Ecohydrological modelling

All drains within the study site were mapped as polylines at a scale of 1:1000 within ESRI ArcMap 10.5® using high-resolution aerial imagery. A 1-m buffer was applied to the polylines to create a polygon of ditch extent across the study site. A 2-m resolution Digital Terrain Model (DTM) with vertical resolution of ± 0.25 m was acquired for the site. Ecohydrological modelling was completed following the methods outlined in Mackin, Barr, et al. (2017). This metric assumes that surface/near-surface hydrological processes are dominant and that there is a close relationship between topography (local surface slope, contributing catchment area), meteorological conditions and ecohydrology (hydrological regime and associated vegetation communities). In summary, the ecohydrological modelling process involves modelling of flow paths across the bog surface using hydrology tools within ESRI ArcMap 10.5®, generating a local surface slope map by generalizing to a resolution of 20 m to reduce the confounding effect of microtopography and calculating modified flow accumulation capacity (MFAC) as follows:

$$\text{MFAC} = \left(\frac{\sqrt{A}}{S} \right) \cdot K \quad (1)$$

where MFAC is the modified flow accumulation capacity (km), A is the upstream contributing catchment area (flow accumulation; m^2), S is the local surface slope (m km^{-1}) and K is the empirical climatic correction factor depending on rate of effective rainfall.

The K factor for the Moyarwood site was determined based on rates of long-term average annual rainfall and potential evapotranspiration for 1981–2010 determined from Met Éireann datasets (Walsh, 2020). Mackin, Flynn, et al. (2017) found that raised bog vegetation communities (ecotopes) that are peat-accumulating are only found on surface slopes of 0.3%–0.6% depending on rates of effective rainfall, unless there is adequate contributing catchment area to support peat-accumulating conditions within areas of steeper slopes (i.e., flushes). Areas modelled with an MFAC output ≥ 30 km were identified as topographically suitable for development of peat-accumulating ecotopes and were exported as a polygon.

The ecohydrological modelling developed by Mackin, Flynn, et al. (2017) provides an indication of the ecotopes that may develop in the short to medium term (10–30 years). A description of the biotic and abiotic characteristics of raised bog ecotopes can be found in Regan et al. (2020). However, there is some uncertainty associated with the modelling approach, which could affect the projected scenarios considered in this study. The model can over-predict rewetting potential in scenarios where there is strong connectivity between the water table in the bog and the underlying mineral substrate resulting in elevated losses of water from the bog to depth, as has been found at some Irish raised bogs, such as Clara Bog (Regan et al., 2020). Furthermore, where topographic conditions are changing, for example due to ongoing subsidence associated with drainage on the margins of the bog, the predictions may change over time. Nevertheless, the model

outputs are a useful means of identifying areas most likely to develop active raised bog vegetation. Over the longer term (> 30 years), it is possible that the model underpredicts long-term potential, since peat-accumulation in the areas predicted to become a sub-central ecotope may slowly begin to expand beyond the areas modelled.

2.9 | Radiative forcing

Greenhouse gas (GHG) emission factors derived in this study for the drained and rewetted areas were used as initial estimates for modelling the radiative forcing for the marginal and sub-central ecotopes, respectively. N_2O fluxes were not detected at the site and so were assigned a value of zero. In the sub-central ecotope, ditches were not considered as separate, as per IPCC Guidance for GHG inventory compilers (IPCC, 2014). Emission factors from ditches in the marginal and sub-marginal ecotopes were derived as the average values from the ditch sample plot at the study site and from literature values for ombrotrophic bogs (Hyvönen et al., 2013; Järveoja, Peichl, Maddison, Teemusk, & Mander, 2016; Peacock et al., 2021; Sundh et al., 2000). Average DOC values from two comparable Irish raised bog sites, namely Abbeyleix and Clara (Regan et al., 2020), were used to provide an estimate of fluvial losses from the relevant ecotopes. As outlined in Evans et al. (2016), we assumed that 90% of DOC degassed (evaded) and was released to the atmosphere.

We assume that CO_2 sequestration peaked in 2017 and that CH_4 emissions peaked in 2014 (see Results section). Thereafter, we assumed that fluxes declined linearly to values typical of near-natural sites in this region ($\sim 30 \text{ g CO}_2\text{-C m}^{-2} \text{ year}^{-1}$ and $9.8 \text{ g CH}_4\text{-C m}^{-2} \text{ year}^{-1}$) after 100 years and then stabilised. These fluxes were used as inputs for the calculation of radiative forcing, i.e., the change in Earth's radiative balance due to the perturbations to atmospheric GHG concentrations that resulted from peatland rewetting (in this instance). Radiative forcing was calculated in annual time steps with a sustained impulse-response model (Lohila et al., 2010; Mathijssen et al., 2017, 2022).

As CO_2 , CH_4 and N_2O are assumed to be instantaneously and completely mixed in the atmosphere (Myhre et al., 2013), the change in atmospheric composition due to surface exchange fluxes of these GHGs, even if occurring at a local scale, was calculated in the radiative forcing model as a change in the total mass of each GHG, assuming globally uniform concentration distribution. Here, GHG uptake was treated as negative emissions.

The atmospheric GHG pulses were modelled to decay according to characteristic time scales related to the global biogeochemical cycles of each GHG (Frolking et al., 2006). For CO_2 , this was implemented as a weighted sum of four exponential functions, where the fastest perturbation time scale was 4.3 years and the slowest decay effectively corresponded to a permanent atmospheric change for 22% of each annual pulse. This impulse response function for CO_2 decay was derived by Joos et al. (2013) from a multi-model ensemble of simulations with coupled carbon cycle climate models. The corresponding function for CH_4 was defined as a first-order decay with a

single perturbation time scale of 12.4 years (Myhre et al., 2013), i.e., a short-term atmospheric response to CH₄ emissions in comparison with CO₂ uptake. For N₂O, a first-order decay with a time constant of 121 years was assumed.

The annual emission pulses were integrated in such a way that in each year the effect of all preceding pulses and their decay with time was accounted for in the atmospheric GHG concentration change. As a major part of the CH₄ decay results from atmospheric oxidation of the emitted CH₄ molecules to CO₂, this conversion was included in the radiative forcing model. The additional CO₂ generates an indirect radiative forcing effect that was allocated to the CH₄ emissions. An efficiency of 80% was assumed for the CH₄-to-CO₂ conversion (Boucher et al., 2009).

The instantaneous radiative forcing (Wm⁻²) resulting from the modelled GHG concentration changes was calculated with a radiative efficiency parameterization derived by Etminan et al. (2016) from simulations with a detailed radiation transfer model. These parameterizations relate the radiative forcing of each GHG to the concentration change, taking into account the spectral interactions between CO₂, CH₄ and N₂O. Radiative forcing due to peatland fluxes was calculated as a marginal change with respect to a specified, temporally varying background concentration (Lohila et al., 2010). The radiative efficiency of each GHG, i.e., the radiative forcing change per concentration change, depends in a nonlinear way on this background concentration, and thus the modelled radiative forcing varies with the scenario assumed for the future concentrations. These concentrations were adopted from the shared socio-economic pathways (SSP) scenarios (Meinshausen et al., 2020).

To examine the impact of rewetting this peatland on climate perturbations, we carried out a modelling exercise where we determined the radiative forcing for the site under the five high priority SSP scenarios outlined in IPCC (2021). A brief description of the scenarios can be found in Table 2.

3 | RESULTS

3.1 | Environmental variables

In the drained area, the water table level (interpolated from manual measurements) remained between 38 and 67 cm below the peat surface for the duration of the study (Figure 3a). Seasonal variability

TABLE 2 Description of shared socio-economic pathways (SSP) scenarios (see Meinshausen et al., 2020) employed in this study

Scenario	GHG emissions level	Global surface temperature (averaged over 2081–2100 compared with 1850–1900)
SSP1-1.9	Very low	Very likely to be higher by 1.0–1.8°C
SSP1-2.6	Low	Very likely to be higher by 1.3–2.4°C
SSP2-4.5	Intermediate	Very likely to be higher by 2.1–3.5°C
SSP3-7.0	High	Very likely to be higher by 2.8–4.6°C
SSP5-8.5	Very high	Very likely to be higher by 3.3–5.7°C

was not evident. In contrast, the water table level in the rewetted area remained above the peat surface for the majority of the study period, with the exception of short periods during the summers of 2013, 2014 and 2017 (Figure 3a). Soil temperature showed strong seasonal variability in both the drained and rewetted areas (Figure 3b). The minimum and maximum values were always observed in the drained area, in winter and summer, respectively, and daily variability was always more pronounced in the drained area. However, mean annual temperature was consistently greater in the rewetted area (Figure 3b).

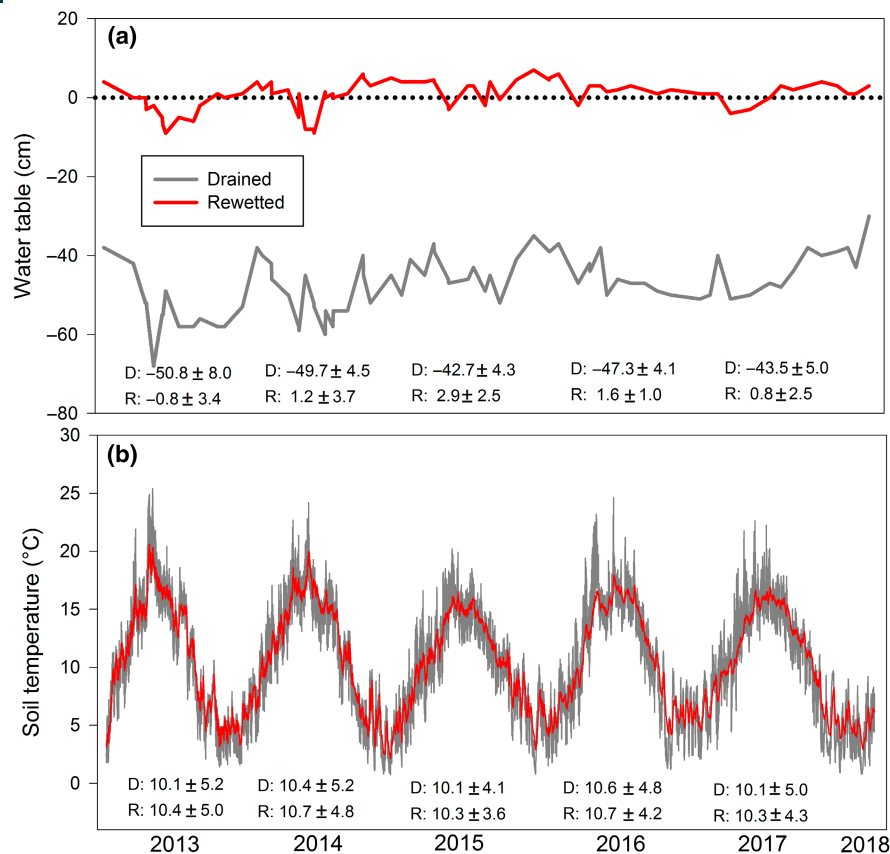
3.2 | Modelling

The GPP models were driven principally by PPFD and either GAI (for the drained plots only; see model parameters in Supplementary Information 4) or soil temperature at 10 cm depth (rewetted plots) (Equations S1-S2 in Supplementary Information). The GPP models performed well, exhibiting r^2 values that ranged from 0.73 to 0.88 (Supplementary Information 2), exhibited low associated standard error estimates (Supplementary Information 2), and good 1:1 line fits (data not shown). The Reco models were based upon the Arrhenius equation (Lloyd & Taylor, 1994) and were nonlinear models related to soil temperature (Equations S3-S4 in Supplementary Information 1). While the performance of the ditch model was relatively poor ($r^2 = 0.37$), the performance of the models for the drained and rewetted areas was better ($r^2 = 0.56$ – 0.86) (Supplementary Information 2). The CH₄ model was a nonlinear model related to soil temperature at 10 cm depth (Equation S5 in Supplementary Information 1). Again, the ditch model performed poorly ($r^2 = 0.34$), although the model performed better for the rewetted plots ($r^2 = 0.55$ – 0.78) (Supplementary Information 2).

3.3 | CO₂ exchange

In general, GPP values were similar between the drained and rewetted areas in year 1 and during the winter period in all years (Figure 4a). In the drained area, GPP was similar across all years, peaking in mid-summer. After year 1, GPP was considerably greater in the rewetted area, especially during mid-summer where it increased steadily to year 3 and was similar thereafter. Ecosystem respiration (Reco) showed a strong seasonality in both areas with emissions peaking in mid-summer, although emissions remained elevated throughout the winter, especially in the drained area (Figure 4b). On average, monthly Reco values in the rewetted area were 56% lower than in the drained area. With the exception of very short periods in the spring, the drained area was a net CO₂ source throughout the study (Figure 4c and Table 3a). In year 1, the rewetted area was a CO₂ source throughout the summer, autumn and early winter but a sink in spring and late winter. In the following years, the rewetted area was a net CO₂ sink for 9 months of the year and a small CO₂ source for 3 months (Figure 4c).

FIGURE 3 (a) Water table level (cm), and (b) hourly soil temperature ($^{\circ}\text{C}$) at 5-cm depth in the drained (D) and rewetted (R) areas of the Moyarwood study site. Water table position was manually measured during field visits and water table values were linearly interpolated between site visits. Mean annual water table and soil temperature shown \pm standard deviation.



Annual NEE in the drained area was lowest in year 3 ($\sim 112 \text{ g C m}^{-2} \text{ year}^{-1}$), and greatest in year 2 ($\sim 164 \text{ g C m}^{-2} \text{ year}^{-1}$) (Table 3a). The single collar in the ditch was also a net CO_2 source throughout the study, emitting between 93.5 and $101.8 \text{ g C m}^{-2} \text{ year}^{-1}$. In contrast, the rewetted site was a net CO_2 sink in all 5 years (Table 3b). Uptake was lowest in year 1 ($-19.5 \text{ g C m}^{-2} \text{ year}^{-1}$) but increased to between -77 and $-148 \text{ g CO}_2\text{-C m}^{-2}$ in years 2–5 (Table 3b).

3.4 | Methane and nitrous oxide exchange

Measured CH_4 fluxes at the drained site were very low and ranged from small uptake ($-0.2 \text{ mg CH}_4 \text{ m}^{-2} \text{ h}^{-1}$) to small emissions ($0.3 \text{ mg CH}_4 \text{ m}^{-2} \text{ h}^{-1}$) (data not shown). However, a statistically significant relationship between fluxes and environmental variables was not established during the modelling process. Instead, annual CH_4 emissions were estimated by linearly interpolating fluxes between measurement dates to provide estimated annual emission values of 0.1 to $0.8 \text{ g C m}^{-2} \text{ year}^{-1}$ (Table 3a).

Emissions from the rewetted ditch were much greater, with strong inter-annual and seasonal variation observed at that sample plot (Figure 5a). In particular, greater emissions were observed in the first 2 years after rewetting, particularly during the growing season, with emissions broadly similar in years 3–5 (Figure 5a and Table 3).

In the rewetted area, CH_4 emissions exhibited strong seasonal variations (Figure 5b), driven mainly by soil temperatures. Typically,

the lowest emissions were observed during the winter months (December to February), and the greatest emissions were seen in summer (June to August). Annual CH_4 emissions were very similar between years, ranging from $18.6 \text{ g C m}^{-2} \text{ year}^{-1}$ in year 5 to $20.6 \text{ g C m}^{-2} \text{ year}^{-1}$ in year 2 (Table 3).

Mean and median measured CH_4 flux values from the combined rewetted and ditch collar datasets (2.9 and $2.4 \text{ mg CH}_4 \text{ m}^{-2} \text{ h}^{-1}$, respectively) were similar to the values for the rewetted dataset only (i.e., when the ditch data were removed) (3.0 and $2.5 \text{ mg CH}_4 \text{ m}^{-2} \text{ h}^{-1}$, respectively). This confirms the guidance provided by IPCC (2014) that ditches in rewetted areas should not be considered separate to non-ditch rewetted areas. Nitrous oxide (N_2O) fluxes were not detectable at either the drained or rewetted sites during the study.

3.5 | Net ecosystem carbon balance (NECB)

In this study, the drained area was a net source of CO_2 (5-year average $141.1 \text{ g C m}^{-2} \text{ year}^{-1}$) and CH_4 ($0.5 \text{ g C m}^{-2} \text{ year}^{-1}$) (Table 3a). When combined with annual DOC values from comparable raised bog sites in Ireland (see Regan et al., 2020), this produced a NECB of $157 \text{ g C m}^{-2} \text{ year}^{-1}$ (equivalent to $1.57 \text{ tonnes C ha}^{-1} \text{ year}^{-1}$). The rewetted area was a net CO_2 sink of $-103.7 \text{ g C m}^{-2} \text{ year}^{-1}$ but was a net source of CH_4 ($19.3 \text{ g C m}^{-2} \text{ year}^{-1}$). Combined with DOC values from Regan et al. (2020), this produced an NECB of $-78 \text{ g C m}^{-2} \text{ year}^{-1}$ (equivalent to $-0.78 \text{ tonnes C ha}^{-1} \text{ year}^{-1}$).

FIGURE 4 Mean monthly (a) gross primary production (GPP), (b) ecosystem respiration (Reco), and (c) net ecosystem exchange (NEE) in the drained ($n = 3$) and rewetted ($n = 9$) areas in Moyarwood from 2013–2018 (years 1–5). Error bars denote standard deviation of the mean monthly value. Positive values indicate a net loss of carbon dioxide (CO_2) to the atmosphere and negative values indicate net uptake of CO_2 by the peatland.

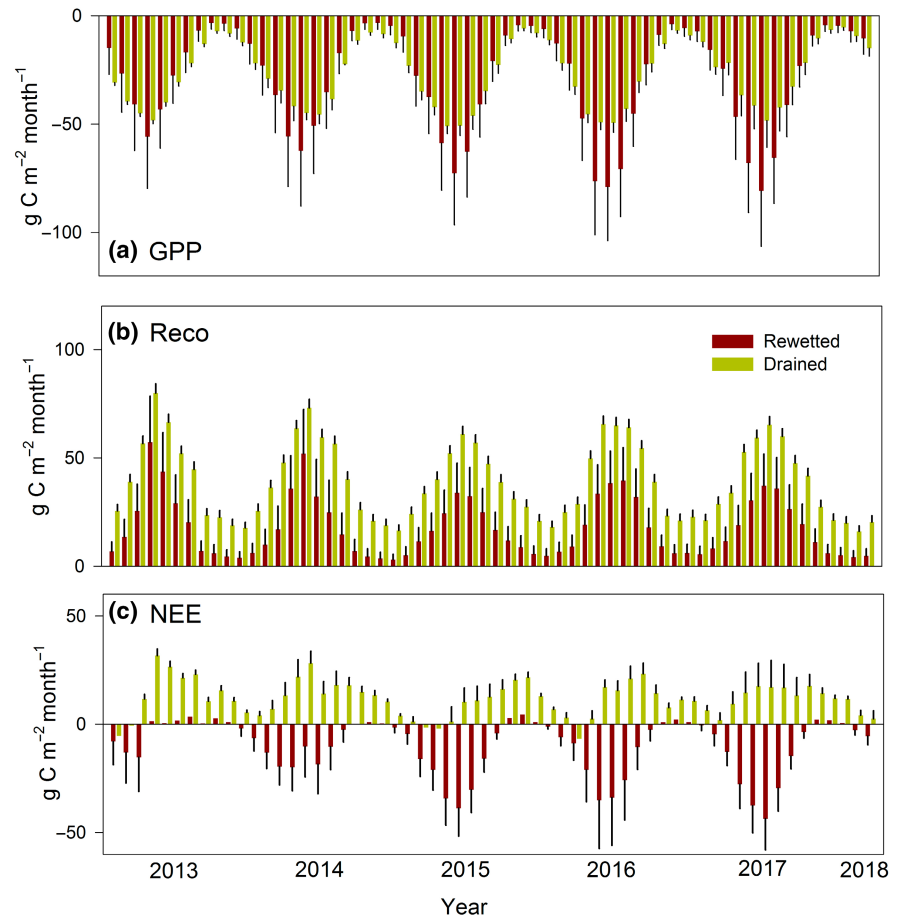


TABLE 3 Net ecosystem carbon balance (NECB: $\text{g C m}^{-2} \text{ year}^{-1} \pm$ standard error) for the (a) drained and (b) rewetted areas in Moyarwood from April 2013 to March 2018 (years 1–5). Positive values indicate a net loss of C to the atmosphere and negative values indicate net uptake of C by the peatland. DOC values taken from Regan et al. (2020)

GHG	CO_2 $\text{g C m}^{-2} \text{ year}^{-1}$	CH_4 $\text{g C m}^{-2} \text{ year}^{-1}$	DOC $\text{g C m}^{-2} \text{ year}^{-1}$	NECB $\text{g C m}^{-2} \text{ year}^{-1}$
(a) Drained				
Year 1	154.2 ± 23.2	0.6 ± 0.1	15.4 ± 0.6	170.2 ± 23.2
Year 2	163.8 ± 14.6	0.8 ± 0.5	15.4 ± 0.6	180.0 ± 14.6
Year 3	111.9 ± 24.3	0.1 ± 0.3	15.4 ± 0.6	127.4 ± 24.3
Year 4	124.9 ± 40.2	0.1 ± 0.3	15.4 ± 0.6	140.4 ± 40.2
Year 5	150.8 ± 25.0	0.7 ± 0.5	15.4 ± 0.6	166.9 ± 25.0
5-year average	141.1 ± 25.5	0.5	15.4 ± 0.6	157.0 ± 25.5
(b) Rewetted				
Year 1	-19.5 ± 11.1	18.76 ± 3.5	6.4 ± 0.4	5.7 ± 11.7
Year 2	-77.3 ± 52.0	20.58 ± 3.1	6.4 ± 0.4	-50.3 ± 52.1
Year 3	-131.1 ± 39.5	19.01 ± 6.1	6.4 ± 0.4	-105.7 ± 40.0
Year 4	-147.8 ± 44.3	19.53 ± 6.9	6.4 ± 0.4	-121.9 ± 44.8
Year 5	-143.0 ± 39.0	18.62 ± 6.5	6.4 ± 0.4	-118.0 ± 39.5
5-year average	-103.7 ± 37.2	19.3 ± 5.2	6.4 ± 0.4	-78.0 ± 37.6

3.6 | Ecohydrology

Prior to rewetting in 2012, active ditches covered approximately 23ha or 10% of the total area of the site with parallel ditches located every 15m. Ecohydrological modelling revealed that most of the site has relatively gentle surface slopes (<1%), with the surface

slope steeper towards the bog margins. Modelling of MFAC indicated that 55.8 ha (24.3%) of the site had a MFAC output of $\geq 30\text{km}$ and has the potential to develop to a sub-central ecotope (i.e., active raised bog habitat) following ditch blocking (Figure 6). This ecotope is analogous to our rewetted study area in terms of water level and vegetation composition.

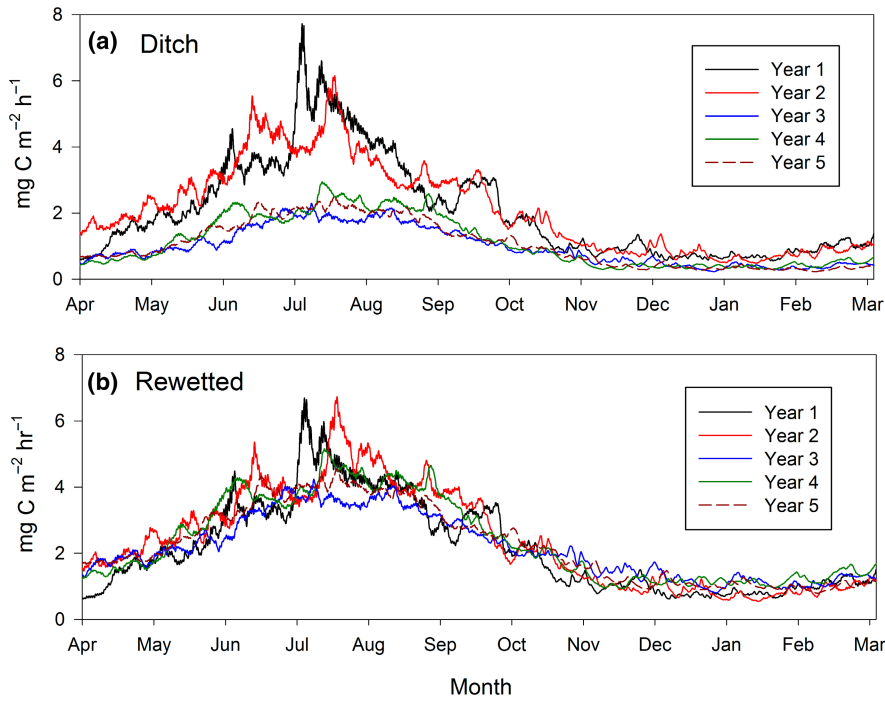


FIGURE 5 (a) Modelled methane (CH₄) fluxes (mg C m⁻² h⁻¹) in the ditch sample plot, and (b) modelled CH₄ fluxes in the rewetted area from 2013 to 2018 (years 1–5). Positive values indicate a net loss of CH₄ to the atmosphere, and negative values indicate net uptake of CH₄ by the peatland.

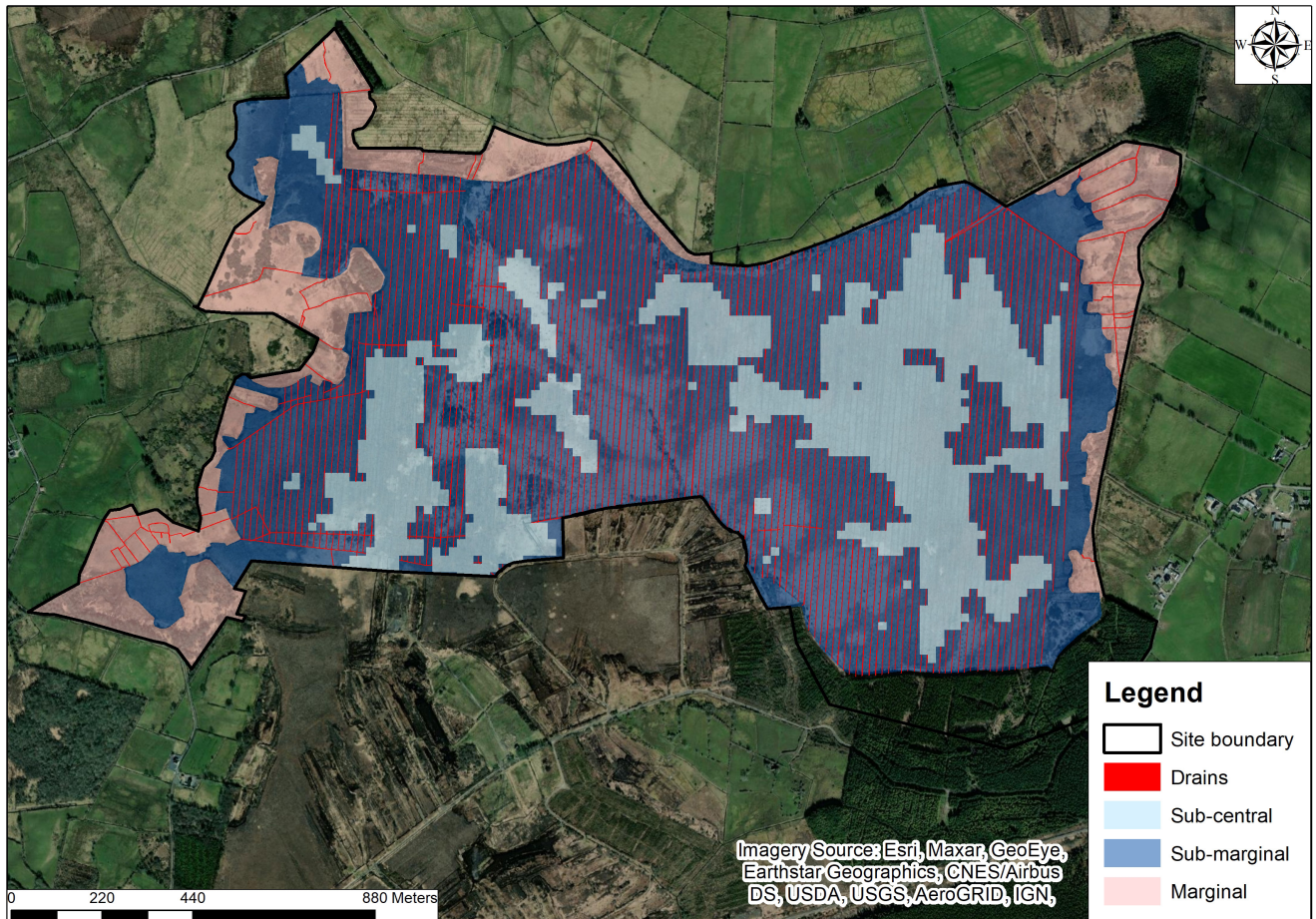


FIGURE 6 Ecohydrological modelling of the Moyarwood site. Predicted ecotopes shown in the legend.

Approximately, 136 ha (59.1%) of the site is expected to develop to a sub-marginal ecotope, an important supporting habitat, but is not anticipated to become peat-accumulating within the short to medium term (i.e., for at least 30 years, as required by the EU Habitats Directive for this habitat to be defined as Annex I habitat degraded raised bog). The modelling indicates that 37 ha (16.1%) will remain as a marginal ecotope (our drained study area) (Figure 6). As per IPCC (2014) guidance, ditches in the sub-central ecotope were not distinguished from the surrounding land, while ditches cover approximately 16 ha (7%) in the sub-marginal ecotope and 0.9 ha (0.4%) in the marginal ecotope.

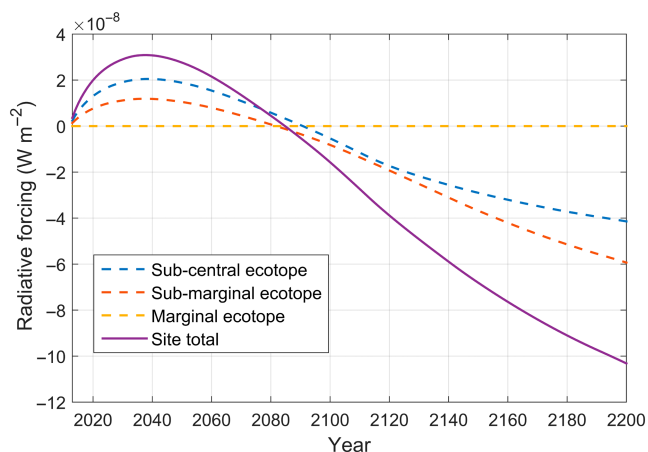


FIGURE 7 Projected radiative forcing (W m^{-2}) resulting from the rewetting-induced changes (versus business-as-usual) in carbon dioxide (CO_2) and methane (CH_4) fluxes at the Moyarwood site under the SSP1-1.9 scenario (see Table 2 for description of scenarios).

3.7 | Radiative forcing scenarios

Following rewetting, the site is projected to have much greater warming impact on the climate for a number of decades (compared with pre-rewetting or business-as-usual), primarily as a result of substantial CH_4 emissions from the sub-central and sub-marginal ecotopes (Figure 7). After 2085, the site will have a less warming impact than it would have had without rewetting. The contribution of the various ecotopes to total radiative forcing following rewetting is projected to vary considerably. The sub-central (the rewetted area in this study) and sub-marginal ecotopes are projected to follow the same general pattern (Figure 7), while no change is projected at the marginal ecotope (i.e., radiative forcing pre-rewetting and post-rewetting would be the same) (Figure 7).

There was a clear difference in radiative forcing at the site-level across the five future shared socio-economic pathway (SSP) scenarios (Figure 8). In terms of the impact on climate, the most optimal scenario was SSP1-1.9 where the effect of CO_2 sequestration (cooling) exceeded CH_4 emissions (warming) by 2085 and was then projected to have a strong cooling impact until the end of the model run (Figure 8). Scenarios SSP1-2.6 and SSP2-4.5 were projected to have a cooling impact after 2094 and 2097, respectively, although a less cooling impact than SSP1-1.9. Scenario SSP3-7.0 remained close to neutral after 2087. In contrast, under scenario SSP5-8.5, the radiative forcing effect of CH_4 emissions was always greater than the radiative forcing effect of CO_2 sequestration, with the result that the site is projected to have a warming impact until 2200 (end of the model run).

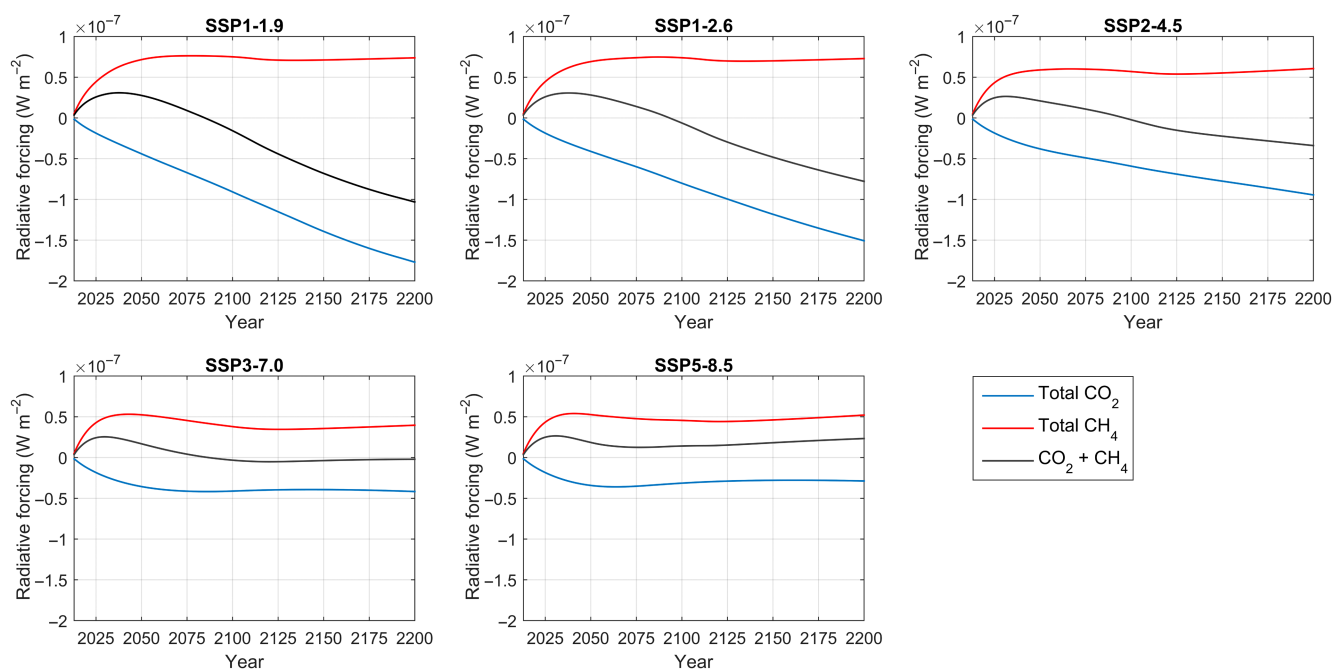


FIGURE 8 Radiative forcing (W m^{-2}) scenarios. See Table 2 for description of the five high priority shared socio-economic pathway (SSP) scenarios.

4 | DISCUSSION

4.1 | Emissions hotspots

Drainage of natural peatlands has been shown to switch former net C sinks (for millennia) into net C sources (e.g., Wilson, Blain, et al., 2016). In our study, the drained area at the margins of the site was a persistent C emission hotspot during the 5-year period (Table 3). Modelled annual values for both CO₂ and CH₄ fluxes are within the range reported by other studies for both peat extraction areas (e.g., Drösler et al., 2014; Swenson et al., 2019; Wilson et al., 2015) and for areas of eroding blanket bog (Artz et al., 2022). The degraded conditions on the margins of Moyarwood are also found extensively on the margins of all raised bogs throughout Republic of Ireland, with most of the EU Natura 2000 network of raised bogs designated for conservation affected to some extent by the indirect drainage caused by peat extraction (Crowley et al., 2021; Regan et al., 2020). However, while accurate estimation of net GHG emissions at the national level from these drained areas remains a huge challenge in the absence of high-resolution areal estimates of peatland land use categories (Connolly, 2018), their inclusion in GHG inventories could potentially add 600,000 tonnes C year⁻¹ to reported emissions from the Land Use, Land Use Change and Forestry (LULUCF) sector in Ireland (Aitova et al., Submitted).

In the absence of rewetting, drained peatlands will remain a persistent source of emissions (Waddington et al., 2002; Wilson et al., 2015). At the global scale, Joosten et al. (2016) have estimated that drained peatlands release 1.15 Gt CO₂ year⁻¹ from the oxidation of peat alone, with an additional 0.85 Gt CO₂ year⁻¹ released to the atmosphere when emissions from peat fires are included in the estimate. Under the Paris Agreement, the goal is to limit global warming to below 2°C compared with pre-industrial levels, and to aim to limit temperature increase to below 1.5°C (UNFCCC, 2015). Peatland rewetting represents an important requirement for meeting the Paris Agreement commitments (Sirin et al., 2021), although compliance will require that 50 million hectares of drained peatlands must be rewetted globally by 2070 (Kreyling et al., 2021). In Ireland, impetus for peatland rewetting (primarily peat extraction and grassland sites) has gained considerable momentum in recent years, driven by the critical need to substantially reduce emissions in the LULUCF sector (Climate Change Advisory Council, 2020).

4.2 | Rewetting and the return of C sink function

Peatland rewetting has been widely espoused as a pragmatic climate change mitigation measure (FAO, 2014; Parish et al., 2008; United Nations Environment Program, 2019) and aims to reverse the hydrological changes brought about by drainage (Ahmad et al., 2020) and to establish the requisite conditions for the establishment of typical peatland plant communities (Richert et al., 2000; Tuittila et al., 2000). Practical management actions, such as drain blocking, the installation of berms and the construction of peat cells (Mackin, Barr, et al., 2017),

can have a profound effect on peatland C dynamics (Günther et al., 2020; Strack et al., 2016; Wilson, Farrell, et al., 2016), and also on local air temperatures (Worrall et al., 2019, 2020, 2022). The rise in the water table that results from these management actions leads to an exclusion of oxygen from the peat below the water surface (Estop-Aragones et al., 2012), which in turn inhibits the activity of aerobic microorganisms (Rydin & Jeglum, 2006), thereby reducing CO₂ production. Conversely, the anoxic conditions provide suitable conditions for methanogenic activity (de Jong et al., 2020; Wen et al., 2018) and CH₄ emissions generally increase after rewetting (IPCC, 2014).

In our study, it was clear that rewetting strongly inhibited ecosystem respiration (Figure 4), while plant productivity (GPP) also increased in these newly rewetted areas (see also Purre & Ilomets, 2021). This combined to rapidly switch this area from a large CO₂ source (if we assume that the values observed in the drained area are representative of emissions from the whole site prior to rewetting) to a small net CO₂ sink in year 1, which increased further in years 3–5. Moreover, the 5-year average CO₂ sink value reported here is greater than those recorded in near-natural Irish peatland systems (cf. McVeigh et al., 2014; Regan et al., 2020) but is similar to other rewetted peat extraction sites in Ireland (Swenson et al., 2019; Wilson, Farrell, et al., 2016) and elsewhere (Nugent et al., 2018). It is probable that the magnitude of the CO₂ sink will eventually decline to values similar to those recorded at natural sites, although when this might occur is not evident from our study. Moreover, caution should be expressed when interpreting our results – rainfall during the growing seasons (April to October) throughout the study period was reasonably consistent (Figure 2), as were water table levels (Figure 3), yet rainfall at the site was much lower during a continent-wide drought period in summer 2018 (after our study had ended). A prolonged drought period would likely lead to a significant drop in water table levels at rewetted sites, with subsequent impacts on the C sink function (Beyer et al., 2021; Koebsch et al., 2020). In contrast, a persistently elevated water table has been shown to not only reduce CO₂ emissions from peatlands but can also act as an effective barrier (in combination with the presence of a moss layer) to the spread of wildfires (Granath et al., 2016).

Soil surface temperature has been suggested as a “master variable” that drives biogeochemical processes (Leonard et al., 2018), and indeed, soil temperature was found to be a very important variable driving CO₂ exchange at this site (see Supplemental Information). While the water table remained close to the peat surface for the majority of the 5-year study period, it did not (a) add to the explanatory power of the models, but (b) did act as an environmental buffer to soil temperatures, preventing the occurrence of extremes in temperatures (Figure 3), particularly during the growing season when decomposition rates could be expected to be at their greatest (Hilasvuori et al., 2013).

Methane emissions from the rewetted area in this study are substantial (Figure 5, Table 3); the mean value reported here (19.3 g C m⁻² year⁻¹) is over twice as large as the IPCC (2014) Tier 1 value for nutrient poor peatlands in the temperate zone (9.2 g C m⁻² year⁻¹). Clearly, the shallow flooding of the living vegetation layer prior to drain blocking and subsequent rewetting has proven an issue in terms of CH₄ emissions, with other peatland rewetting studies reporting

similar findings following inundation (Hahn-Schöfl et al., 2011; Huth et al., 2021). While the NECB in the drained area was primarily dominated by the CO₂ component (90%), the contribution of CH₄ was considerably greater in the rewetted area, a feature noted in other peatland rewetting studies (e.g., Schaller et al., 2022; Wilson, Blain, et al., 2016) and modelling exercises (e.g., Tanneberger et al., 2020).

Nevertheless, our results indicate that the rewetted area was a modest net C sink 2 years after rewetting, with further improvement in NECB values observed in the following years (Table 3). These results are encouraging and confirm that peatland rewetting should be given high priority in countries where degraded and/or drained peatlands are a key land use category as it achieves (a) restoration objectives (providing a number of co-benefits for hydrology, biodiversity, carbon), and (b) is also an economically sound objective (Glenk et al., 2021; Renou-Wilson & Wilson, 2018).

4.3 | Climate implications

The importance of maximizing the rewetted area within a site was also confirmed by our radiative forcing exercise. The climate impact of peatlands over the last Ice Age has been a trade-off between the reduced radiative forcing caused by CO₂ sequestration and storage, and the enhanced radiative forcing from CH₄ emissions (Frolking & Roulet, 2007; Petrescu et al., 2015). In our study, around 24% of the site is projected to become a sub-central ecotope, an active raised bog habitat, and around 59% is projected to become a sub-marginal ecotope. Over the next 200 years, both ecotopes are projected to result in a reduced climate warming impact compared with the pre-rewetting situation (Figure 7), which is in agreement with other peatland rewetting studies (Günther et al., 2020; Huth et al., 2021; Tanneberger et al., 2020). Moreover, work by Worrall et al. (2019) has demonstrated that rewetting of former peat extraction sites can lead to a cooling of the local climate, driven in the main by restoration of the water table (Worrall et al., 2022).

In contrast, the marginal ecotope in our study is projected to have the same warming impact on the climate (in the absence of rewetting) (Figure 7). Clearly, there is an incentive to increase the area of the sub-central and sub-marginal ecotopes at the expense of the marginal ecotope. For example, under current best restoration practice for raised bogs in Ireland, the main marginal ditch around peatland sites tends to remain open (and therefore active) to prevent accidental flooding of adjacent farmland (Mackin, Barr, et al., 2017). Marginal bunds located close to that ditch could be employed to facilitate a rise in the water level on the peatland side of the bund (Mackin, Barr, et al., 2017), thereby decreasing the area of the marginal ecotope with a subsequent positive impact on C dynamics. Similarly, initiatives to rewet adjoining agricultural farmland could help to increase the area of sub-central and sub-marginal ecotopes, although this would require agreement with those landowners.

Work by Swindles et al. (2019) has highlighted the precarious condition of European peatlands with regard to future climate change. In Ireland, the distribution of fens, blanket bogs and raised

bogs is likely to be severely diminished under increasing regional temperatures and precipitation changes in the decades ahead (Coll et al., 2014; Jones et al., 2006), and “second generation” peatlands, such as rewetted cutaway and cutover sites may be even more vulnerable (Renou-Wilson & Wilson, 2018). Climate envelope modelling has projected similar findings for peatlands in Britain (Clark et al., 2010; Ferretto et al., 2019; Gallego-Sala et al., 2010). Radiative forcing for our Moyarwood site under the five high priority shared socio-economic pathways (SSP) scenarios further indicates the importance of keeping below the 2°C warming threshold, although as global temperatures continue to increase (IPCC, 2021), the very low GHG emissions scenario SSP1-1.9 (and by extension the Paris Agreement targets) looks ever more challenging. While the appropriateness of Representative Concentration Pathway 8.5 (RCP8.5) as an emissions pathway has received some discussion in recent years (e.g., Hausfather & Peters, 2020a, 2020b; Schwalm et al., 2020), it has been projected to lead to a decrease in peatland C sequestration at lower latitudes and a “decline phase” in mid-latitude peatlands (Gallego-Sala et al., 2018) and lead to a switch in the C sink function of northern peatlands (from net sink to source) (Qiu et al., 2022). Under the *high* (SSP3-7.0) and *very high* emissions (SSP5-8.5) scenarios in our study, the climate cooling effect of C sequestration at our study site is projected to plateau within a few decades of rewetting (Figure 8) and the climate warming impact of sustained CH₄ emissions is likely to predominate for the next two centuries.

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CONFLICT OF INTEREST

The authors have no conflict of interest to declare.

DATA AVAILABILITY STATEMENT

Data available at <https://doi.org/10.5281/zenodo.6901318>. Detailed description of the statistical GHG and radiative forcing models used in this study can be found in Supplemental Information.

ORCID

David Wilson  <https://orcid.org/0000-0003-2776-8426>

Francis Mackin  <https://orcid.org/0000-0001-9743-0354>

Juha-Pekka Tuovinen  <https://orcid.org/0000-0001-7857-036X>

Gerald Moser  <https://orcid.org/0000-0002-0030-2370>

Catherine Farrell  <https://orcid.org/0000-0003-2940-6445>

Florence Renou-Wilson  <https://orcid.org/0000-0002-8637-9853>

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SUPPORTING INFORMATION

Additional supporting information can be found online in the Supporting Information section at the end of this article.

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