Global Change Biology

Global Change Biology (2016) 22, 4080–4095, doi: 10.1111/gcb.13325

Multiyear greenhouse gas balances at a rewetted temperate peatland

DAVID WILSON[1], CATHERINE A. FARRELL[2], DAVID FALLON[2], GERALD MOSER[3], CHRISTOPH MÜLLER[3,4] and FLORENCE RENOU-WILSON[4]

[1]*Earthy Matters Environmental Consultants, Glenvar, Co. Donegal, F92 HX03, Ireland,* [2]*Bord na Móna, Tullamore, Co. Offaly, Ireland,* [3]*Justus Liebig University Giessen, Ludwigstraße 23, 35390, Giessen, Germany,* [4]*University College Dublin, Belfield, Dublin 4, D04 V1W8, Ireland*

Abstract

Drained peat soils are a significant source of greenhouse gas (GHG) emissions to the atmosphere. Rewetting these soils is considered an important climate change mitigation tool to reduce emissions and create suitable conditions for carbon sequestration. Long-term monitoring is essential to capture interannual variations in GHG emissions and associated environmental variables and to reduce the uncertainty linked with GHG emission factor calculations. In this study, we present GHG balances: carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O) calculated for a 5-year period at a rewetted industrial cutaway peatland in Ireland (rewetted 7 years prior to the start of the study); and compare the results with an adjacent drained area (2-year data set), and with ten long-term data sets from intact (i.e. undrained) peatlands in temperate and boreal regions. In the rewetted site, CO_2 exchange (or net ecosystem exchange (NEE)) was strongly influenced by ecosystem respiration (R_{eco}) rather than gross primary production (GPP). CH_4 emissions were related to soil temperature and either water table level or plant biomass. N_2O emissions were not detected in either drained or rewetted sites. Rewetting reduced CO_2 emissions in unvegetated areas by approximately 50%. When upscaled to the ecosystem level, the emission factors (calculated as 5-year mean of annual balances) for the rewetted site were (±SD) -104 ± 80 g CO_2-C m^{-2} yr^{-1} (i.e. CO_2 sink) and 9 ± 2 g CH_4-C m^{-2} yr^{-1} (i.e. CH_4 source). Nearly a decade after rewetting, the GHG balance (100-year global warming potential) had reduced noticeably (i.e. less warming) in comparison with the drained site but was still higher than comparative intact sites. Our results indicate that rewetted sites may be more sensitive to interannual changes in weather conditions than their more resilient intact counterparts and may switch from an annual CO_2 sink to a source if triggered by slightly drier conditions.

Keywords: carbon dioxide, climate change mitigation, interannual variation, methane, peat soils, rewetting

Received 23 February 2016 and accepted 14 April 2016

Introduction

Drained peat (organic) soils are a major source of greenhouse gas (GHG) emissions to the atmosphere (Joosten *et al.*, 2012, IPCC, 2014; Smith *et al.*, 2014). When an intact peatland is drained for peat extraction purposes it results in fundamental changes in GHG dynamics within the ecosystem. For example, while intact peatlands may be small net annual carbon (C) sinks (Roulet *et al.*, 2007; Nilsson *et al.*, 2008; Koehler *et al.*, 2011), drainage results in increased carbon dioxide (CO_2) emissions from the peatland to the atmosphere (Haddaway *et al.*, 2014; Wilson *et al.*, 2015) and a decrease in methane (CH_4) emissions (Strack & Waddington, 2007), although emissions from drainage ditches may remain high (IPCC, 2014) and increased emissions of nitrous oxide (N_2O) (Alm *et al.*, 2007a).

Correspondence: David Wilson, tel. + 353 74 9177613, e-mail: david.wilson@earthymatters.ie

The cessation of peat extraction offers opportunities for the creation of new habitats and ecosystems that may positively contribute to a wide range of environmental objectives (Renou *et al.*, 2006). For example, the rewetting of drained organic soils has received considerable attention in recent years with research focused on the prospects of stimulating biodiversity (Parish *et al.*, 2008; Ramshunder *et al.*, 2011), for erosion prevention (Wilson *et al.*, 2011), the development of C offset markets (Dunn & Freeman, 2011; Bonn *et al.*, 2014), and the reduction of GHG emissions and the potential for C sequestration (Wilson *et al.*, 2013; Dixon *et al.*, 2014).

Rewetting defined here as 'the deliberate action of raising the water table on drained soils to re-establish water saturated conditions' (IPCC, 2014) may include a wide palette of management actions and is highly site-specific (Wheeler, 1995; Quinty & Rochefort, 2003). In industrial extracted peatlands methods to ensure the maintenance of a water table (WT) level close to the surface of the peat commonly include drain blocking, bund construction and landscaping of the peat surface (Farrell & Doyle, 2003; Smolders *et al.*, 2003).

Rewetting of drained peat soils has been promoted as a key climate mitigation tool (Joosten *et al.*, 2012; Bonn *et al.*, 2014; Smith *et al.*, 2014). However, the effect of rewetting on GHG dynamics in these new ecosystems has been somewhat unpredictable with some studies reporting high CO_2 and CH_4 emissions postrewetting (Wilson *et al.*, 2007b, 2009; Vanselow-Algan *et al.*, 2015), while others have shown that the CO_2 sink function can be re-established relatively quickly (Tuittila *et al.*, 1999; Kivimäki *et al.*, 2008; Wilson *et al.*, 2013). Clearly, GHG dynamics postrewetting are influenced by a wide range of factors (e.g. time since rewetting, climate, peat type, nutrient status, vegetation, hydrology), which suggests that a one size fits all approach in deriving generic emission factors may not be the most appropriate (Wilson *et al.*, 2016). Indeed, the large range in uncertainty values that are associated with the Tier 1 emission factors values for rewetted organic soils in the 2013 IPCC *Wetlands Supplement* is indicative of a highly heterogeneous response to rewetting (IPCC, 2014, Wilson *et al.*, 2016).

Long-term monitoring (here defined as the monitoring of GHG fluxes and associated environmental variables for periods greater than 4 years) is essential to capture interannual variations (IAVs) in GHG fluxes and to reduce uncertainty in emission factors (e.g. Teklemariam *et al.*, 2010; Moore *et al.*, 2011; McVeigh *et al.*, 2014). Variations in weather inputs, such as rainfall, light and temperature directly affect internal peatland functioning (Nijp *et al.*, 2015) through fluctuations in WT, soil temperature and plant growth dynamics (Teklemariam *et al.*, 2010). In rewetted industrial cutaway peatlands, many ecosystem properties have been impaired by decades of drainage, a lack of vegetation cover (Graf *et al.*, 2012), modified peat structure (Kennedy & Price, 2005) and altered hydrological parameters (Schlotzhauer & Price, 1999; Waddington & Price, 2000; Price & Whitehead, 2001). In the absence of a functioning acrotelm, large WT fluctuations are common (Price, 1996; Wilson *et al.*, 2007b), which in turn may result in the increased decomposition of both the residual peat substrate and more recent organic matter deposits. Similarly, the rapid successional changes that can be a feature of industrial peatlands postrewetting (Tuittila *et al.*, 2000b; Wilson *et al.*, 2013), combined with IAV in both external and internal factors, may further accentuate the unpredictability of GHG exchange in these new ecosystems.

A previous 3-year study at a rewetted industrial cutaway peatland in Bellacorick, north-west Ireland, by Wilson *et al.* (2013) reported high annual CO_2 uptake and moderately high CH_4 emissions from various vegetated microsites and noted a wide variation in annual GHG values across all microsites (including nonvegetated). The current investigation builds on this study and uses an extended data set from the same measurement location, in combination with new drained microsites (Wilson *et al.*, 2015). Although numerous multiyear data sets (encompassing 4- to 12-year monitoring periods) have been published in regard to GHG exchange in intact/semi-intact peatlands (e.g. Roulet *et al.*, 2007; Peichl *et al.*, 2014; Aurela *et al.*, 2015; Helfter *et al.*, 2015), the multiyear data set presented here is, to our knowledge, the longest GHG study at a rewetted peatland.

The objectives of this study were as follows: (1) to estimate annual CO_2, CH_4 and N_2O balances for a range of drained and rewetted microsites at an industrial cutaway peatland, (2) to examine the IAV in weather inputs and in the annual CO_2, CH_4 and N_2O balances, (3) to upscale our results to the ecosystem level and derive emission factors for the rewetted site and (4) to compare our findings with long-term GHG data sets from intact/semi-intact peatlands.

Materials and methods

Study site

The study site is located at Bellacorick, Co. Mayo, on the north-west coast of Ireland (rewetted site: 54°7′29.9″N, 9°33′22.2″W; drained site: 54°7′40.8″N, 9°33′21.6″W). The site is a former Atlantic blanket bog and milled peat was extracted from the site from 1961 to 2002 (Fallon *et al.*, 2012). At the cessation of peat extraction, the majority of the site was bare peat (87%), with the remaining 13% comprised of open-water, *Sphagnum cuspidatum–Eriophorum angustifolium*-, and *Juncus effusus*-dominated communities. In 2002, a larger-scale rehabilitation plan was implemented in a sequential fashion across the peatland, which involved drain blocking, the establishment of bunds and the creation of shallow pools. Since then, the landscape has developed as a composite of microsites: bare peat fields (some with active drainage systems), and recolonized drained and rewetted areas (Fallon *et al.*, 2012). The average thickness of the residual peat layer within the study site is 50 cm and the peat is composed mainly of highly humified cyperaceous peat overlying glacial till (Farrell & Doyle, 2003). The peat is nutrient-poor with a C : N ratio of 58 (n = 6) and average pH of 3.8 (±0.3). For a detailed description of the site and postextraction rewetting measures, see Farrell & Doyle (2003) and Wilson *et al.* (2013).

Climatic conditions

The sites are located within the temperate zone as defined by IPCC (2006) and are characterized by an oceanic climate with prevailing south-west winds, mild mean annual air temperatures (10.3 °C) and high annual rainfall (1245 mm) (Met Éireann, Belmullet Station, 1981–2010).

Microsites

In 2008, 12 square aluminium collars (60 × 60 cm) were inserted to a depth of 30 cm into the peat in the following microsites: *Juncus effusus–Sphagnum cuspidatum* (JS)-dominated communities (n = 3); *Sphagnum cuspidatum–Eriophorum angustifolium* (SE)-dominated communities (n = 3); *Eriophorum angustifolium* (EA)-dominated communities (n = 3); and rewetted bare peat (BP_R, n = 3). In November 2011, additional collars were established in two microsites where the drainage ditches remained functional: drained bare peat (BP_D, n = 3) and *Juncus effusus*-dominated communities (J_D n = 3). Here, we define 'drained' as sites with a mean annual water table (WT) level deeper than −20 cm (Couwenberg & Fritz, 2012; Strack *et al.*, 2014).

Environmental monitoring

A weather station (WatchDog Model 2400; Spectrum Technologies Inc., Aurora, IL, USA) was established at the rewetted site and recorded photosynthetic photon flux density (PPFD; μmol m^{-2} s^{-1}) and soil temperature (°C) at 5, 10 and 20 cm depths at 10-min intervals. Additional soil loggers (Hobo External Data Loggers; Onset Computer Corporation, Bourne, MA, USA) were installed within the SE and BP_D microsites and recorded hourly soil temperatures (°C) at 5, 10 and 20 cm depths. Water table level (WT) was manually measured from dipwells (internal diameter 2 cm) inserted adjacent to each gas measurement collar and linearly interpolated between measurement days to provide continuous data (Alm *et al.*, 2007b; Wilson *et al.*, 2015). Wooden boardwalks were established at each site to facilitate GHG measurements and to prevent damage to the vegetation and compression of the peat.

Field measurements

Greenhouse gas (GHG) fluxes were measured at fortnightly or monthly intervals from November 2008 to December 2013. Net ecosystem exchange (NEE) was measured with a static polycarbonate chamber (60 × 60 × 33 cm) equipped with two internal fans (flow speed ~ 1 m s^{-1}) to ensure mixing of the air. A radiator cooling system (with submerged ice packs) was used to maintain the temperature within the chamber close to the ambient air temperature (Alm *et al.*, 2007b). NEE was measured under a range of ambient light levels (PPFD; μmol m^{-2} s^{-1}) and PPFD was recorded from a sensor (PAR-1; PP Systems, King's Lynn, Norfolk, UK) located inside the chamber. NEE measurements were carried out between 8 am and 6 pm in the summer and between 9 am and 3 pm in the winter (3–9 measurements per collar per measurement day) to ensure that the maximum PPFD was reached at each measurement date. For each measurement, the chamber was placed in a water-filled channel at the top of the collar, and the CO_2 concentration (ppmv) in the chamber headspace was measured at 15-s intervals over a period of 60–180 s using a portable CO_2 analyser (EGM-4; PP Systems, UK). 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 μmol m^{-2} s^{-1}). The chamber was removed from the collar between measurements until the CO_2 concentration inside the chamber returned to ambient levels. Ecosystem respiration (R_{eco}) was then measured by covering the chamber in an opaque cover and CO_2 exchange was measured as outlined above. Concurrently with each CO_2 measurement, air temperature (°C) within the chamber and soil temperatures at 5, 10 and 20 cm depths were recorded at each collar (soil temperature probe; ELE International, Bedfordshire, UK). The WT position relative to the soil surface was manually measured with a water level probe (Eijkelkamp Agrisearch Equipment, Giesbeek, the Netherlands).

Methane (CH_4) and nitrous oxide (N_2O) fluxes were measured at monthly intervals using an opaque, polycarbonate chamber (60 × 60 × 25 cm) equipped with a battery-operated fan which mixed the air within the chamber headspace (flow speed ~ 1 m s^{-1}). 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 wintertime when low fluxes were expected. During each measurement, air temperature inside the chamber, soil temperature (at 5, 10 and 20 cm depths) and WT were recorded at each collar. From 2009 to 2011, CH_4 and N_2O concentrations of the gas samples were measured in a gas chromatograph with attached autosampler unit (Shimadzu GC-2014; LAL, Göttingen, Germany) within 24 h of collection. For a more detailed description, see Wilson *et al.* (2013). In 2012–2013, samples were transferred to Exetainers® 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 CH_4 and N_2O standards were supplied by Deuste Steininger GmbH. Gas peaks were integrated using Galaxie software (Varian Inc., Palo Alto, CA, USA).

To incorporate the seasonal dynamics of the plants into CO_2 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 *et al.* (2007a).

Flux calculations

Flux rates (mg CO_2 m^{-2} h^{-1}, mg CH_4 m^{-2} h^{-1}, μg N_2O m^{-2} h^{-1}) were calculated as the linear slope of the CO_2, CH_4 and N_2O 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 (e.g. 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.*, 2007b). In these cases, the flux data were examined graphically and fluxes with obvious nonlinearity (due to chamber leakage, fan malfunction, ebullition, etc.) were discarded. The remainder were accepted provided that some of the environmental variables measured at the same time (e.g. PPFD, soil temperature) were sufficiently low to account for the low flux values (Wilson *et al.*, 2013). In this study, we followed the sign convention whereby positive values indicate a GHG flux from the peatland to the atmosphere (source) and negative values indicate a flux from the atmosphere to the peatland (sink). Gross primary production (GPP) was calculated as NEE minus R_{eco} (Alm *et al.*, 2007b) and the closest R_{eco} flux value in time to a NEE flux value was used.

Modelling

Statistical and physiological response models (Alm *et al.*, 2007b) were constructed and parameterized for each study site (see S1). Model evaluation was based on the following criteria: (1) statistically significant model parameters ($P < 0.05$), (2) lowest possible standard error of the model parameters and (3) highest possible coefficient of determination (adjusted r^2) (see Laine *et al.*, 2009b). During model construction, the relationship between R_{eco}, GPP or CH_4 and a range of independent environmental variables (recorded in conjunction with flux measurements) was tested. 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 light (Tuittila *et al.*, 1999), and to GAI and/or water table (S1a). 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 R_{eco} models are based upon the Arrhenius equation (Lloyd & Taylor, 1994) and are nonlinear models related to soil temperature and water table (S1b). The CH_4 models are nonlinear models related to soil temperature, water table and GAI (S1c).

Annual carbon dioxide and methane balances

The response functions estimated for R_{eco}, GPP and CH_4 were used for the reconstruction of the annual CO_2 and CH_4 balances. Fluxes were reconstructed for each sample plot in combination with an hourly time series of (1) $T_{5\ cm}$, (2) WT levels linearly interpolated from weekly measurements, (3) PPFD values recorded by the weather station and (4) plot-specific modelled GAI. Annual NEE was calculated as the sum of annual GPP and annual R_{eco} (negative values). The annual CO_2 and CH_4 balances (g C m^{-2} yr^{-1}) were calculated for each sample plot by integrating the hourly values over each 12-month period.

Upscaling

All long-term published CO_2 data sets in intact/semi-intact peatlands are measured with the eddy covariance (EC) method, which spatially integrates CO_2 fluxes over a wide footprint (Baldocchi, 2003) to produce an emission factor with an associated level of uncertainty. Therefore, to compare the values measured in this study with estimates from the other long-term sites, we upscaled the annual balances from the microsite to the ecosystem level ($site_{upscaled}$). Emission factors (g CO_2-C m^{-2} yr^{-1} and g CH_4-C m^{-2} yr^{-1}) were then calculated for the rewetted site as the average of the annual $site_{upscaled}$ NEE and CH_4 balances.

A high resolution habitat map (area 5.69 ha) was created with the weather station at the centre. A vegetation and habitat mapping survey was carried out in the summer of 2011 and aerial colour photographs at a scale of 1 : 6000 were used as a base map. The minimum habitat unit size for the survey was set at 0.5 ha. Habitat classifications (Fossitt, 2000) were applied on site using the aerial photos as a guide. Features on the ground, such as vegetation, watercourses and roads were also noted. Once the fieldwork was completed, the maps were digitized using ArcMap 10 GIS package, with every unit (in this case vegetation communities, open water, roads, etc.) mapped digitally. Guidelines as outlined by Smith *et al.* (2011) were followed. The proportion of JS, SE, EA and BP_R microsites in $site_{upscaled}$ were 14.3, 60.6, 18.5 and 6.6%, respectively. Although the habitat map was created in 2011 (the mid-point of the GHG study), we used the same microsite cover values for upscaling in the preceding 2 years (2009 and 2010) and in the subsequent 2 years (2012 and 2013).

Global warming potential

The global warming potential (GWP) of each microsite and $site_{upscaled}$ was calculated to determine the effect on the global climate of rewetting at the site. Dissolved organic carbon (DOC) losses from the drained and rewetted sites were estimated according to guidance provided by IPCC (2014). CH_4 and N_2O fluxes were converted to CO_2 equivalents (t CO_2-eq ha^{-1} yr^{-1}) according to their GWP, which include climate–carbon feedbacks (see Myhre *et al.*, 2013) over a 100-year horizon: $CH_4 = 34$ and $N_2O = 298$.

Statistical analysis

Statistical analyses were performed using SPSS version 21.0 for Windows (IBM SPSS Statistics for Windows, Armonk, NY, USA). P values smaller than 0.05 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 R_{eco} and NEE was calculated by summing up the maximum and minimum standard errors associated with each of the model parameters (e.g. Drösler, 2005; Elsgaard *et al.*, 2012; Renou-Wilson *et al.*, 2014). Uncertainty in the annual R_{eco} or NEE estimate was calculated following the law of error propagation as the square root of the sum of the squared standard errors of GPP and R_{eco}.

Results

Photosynthetic photon flux density

Total PPFD varied by around 7% between years, with the highest annual values observed in 2010 and 2013, and the lowest value in 2011. A strong seasonal trend was evident in PPFD values across all years with maximum values observed in May (2012), June (2009, 2010, 2011) or July (2013), and minimum values in December of each year (Fig. 1a). Significant differences were observed in mean annual values between years (Friedman test $\chi^2 = 45.46$, $P < 0.001$) and the *post hoc* Wilcoxon signed-rank test showed that significant differences existed in six of the annual comparisons ($P < 0.001$) but were nonsignificant in the other four ($P > 0.05$).

Annual rainfall

Annual rainfall varied between years with 2011 recorded as the wettest year and 2013 as the driest year (Fig. 1b). Years 2009 and 2011 were 7 and 11% wetter, and 2010, 2012 and 2013 were 11, 10 and 15% drier, respectively, than the long-term average for the area (Met Éireann, Belmullet Station). The highest rainfall in all years occurred in the October–December period and the lowest in the May–July period (Fig. 1b).

Soil temperatures

Soil temperatures in the rewetted site varied considerably between years (Fig. 2a) and annual means were significantly different between years (Friedman test $\chi^2 = 6621$, $n = 8759$, $P < 0.001$). Mean annual soil temperatures in 2010 were influenced by the abnormally cold conditions for this region at both the start and end of the year, while temperatures during the summer 2011 were approximately 2 °C lower than the other years. Soil temperatures were highly

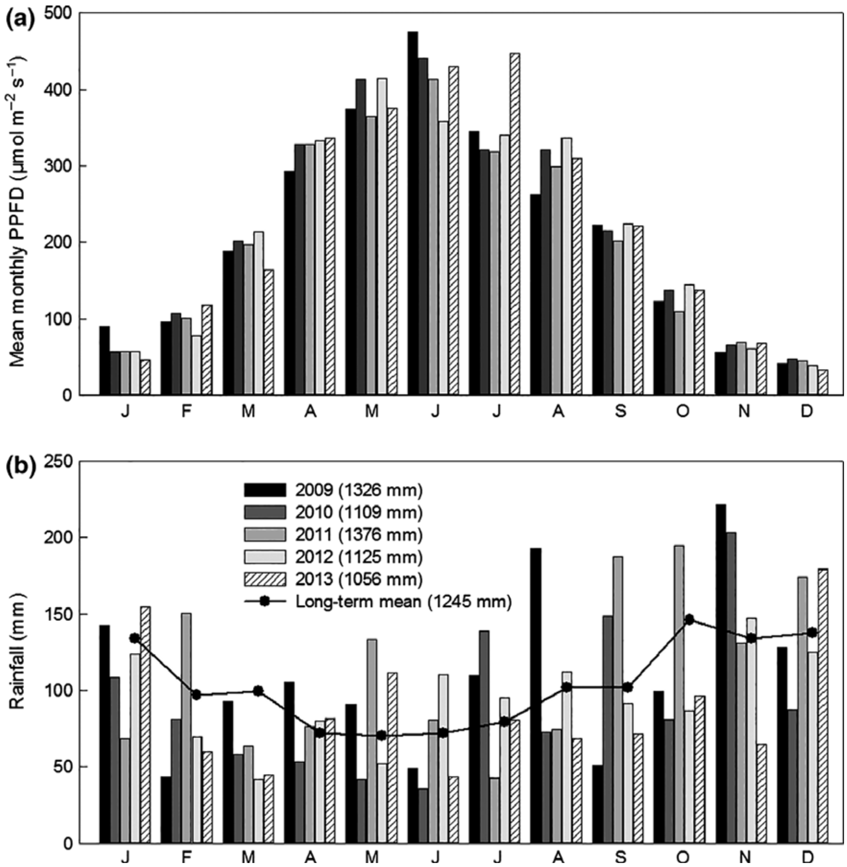

Fig. 1 (a) Mean monthly photosynthetic photon flux density (PPFD, µmol m^{-2} s^{-1}) and (b) monthly rainfall (mm) for 2009–2013. Annual rainfall for the study period and long-term annual mean are shown in parentheses. Rainfall data and long-term mean values (1981–2010) courtesy of Met Éireann, Belmullet Station (www.met.i.e).

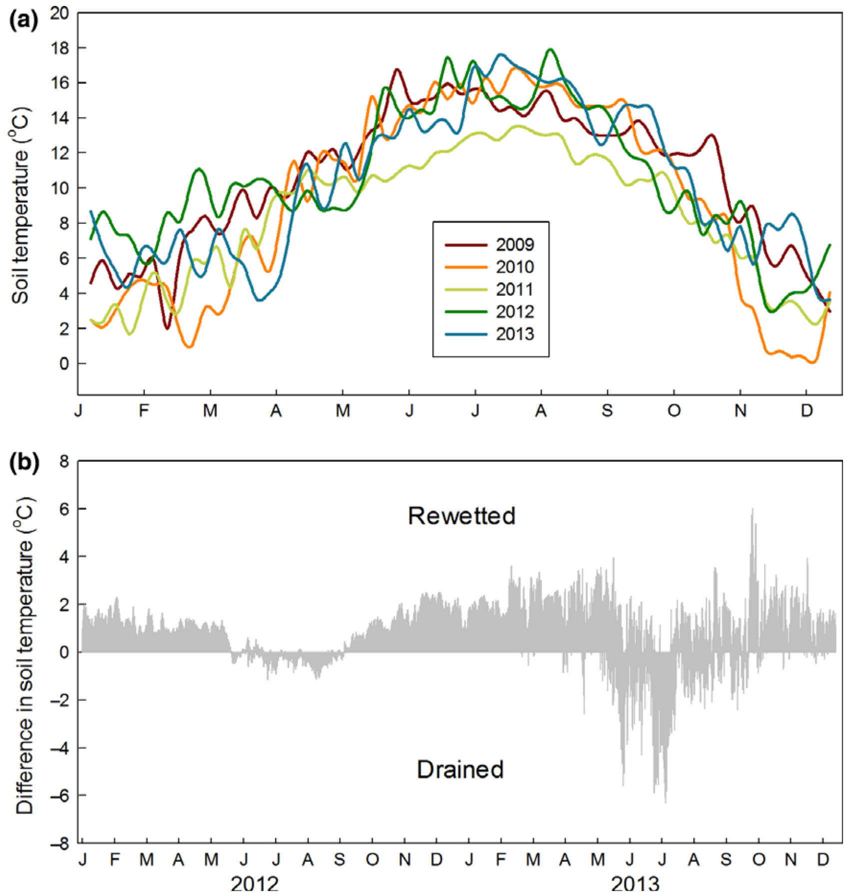

Fig. 2 (a) Mean weekly soil temperature (°C) at 5 cm depth ($T_{5\ cm}$) at Bellacorick rewetted site for 2009–2013 and (b) difference in hourly soil temperature between the rewetted and drained sites for 2012–2013 (comparative data for 2009, 2010 and 2010 not available as drained site established Nov 2011).

seasonal across all years. The lowest value (−0.4 °C) was observed in December 2010 and the highest value (19.6 °C) was recorded in June 2012 and July 2013.

The annual mean soil temperatures were significantly different (Wilcoxon signed-rank test, $z = -80.78$, $P < 0.001$) between the rewetted and drained sites (Fig. 2b). In the winter and spring periods, temperatures in the rewetted site were up to 6 °C higher than those in the drained site, whereas in summer and early autumn the drained site was the warmer site with the difference close to 6 °C in 2013 (Fig. 2b).

Water tables (WTs)

Mean annual WT levels were relatively similar at all microsites in 2009, 2011 and 2012 (Table 1) and lower (i.e. deeper) in 2010 and 2013. For all rewetted vegetated microsites, the mean annual WT level remained positive (i.e. above the soil surface) throughout the study. However, all the rewetted microsites showed considerable drops in WT levels in summer 2010 with the lowest value (approximately 40 cm below the surface) recorded in BP_R (Table 1). The mean annual WT at the drained sites were negative (i.e. below the soil surface) in the 2 years of the study. Mean annual WT values were similar in both BP_D and in J_D in both years, although deeper WT levels (during the summer) were evident in BP_D (Table 1). Water retention curves showed that the WT position remained above the soil surface for over 90% of the time in the rewetted vegetated microsites during the 5-year period (Fig. 3). For BP_R, the water table was above the soil surface for 50% of the time (Fig. 3).

Green area index

Mean annual GAI values were similar between the rewetted vegetated microsites (~1 $m^2\ m^{-2}$) but were much lower in the drained J_D site (~0.4 $m^2\ m^{-2}$)

Table 1 Mean annual water tables (cm) at the *Juncus effusus–Sphagnum cuspidatum* (JS), *Sphagnum cuspidatum–Eriophorum angustifolium* (SE), *Eriophorum angustifolium* (EA), rewetted bare peat (BP_R), drained bare peat (BP_D) and *Juncus effusus* drained (J_D) microsites at Bellacorick. Positive values indicate mean annual values above the soil surface; negative values indicate mean annual values below the soil surface. Minimum and maximum values (cm) shown in parentheses

Microsite	JS	SE	EA	BP_R	BP_D	J_D
Year						
2009	5.6 (−4, 10.5)	12.5 (8, 16)	6.3 (−9, 14)	−0.4 (−19, 5)	–	–
2010	3.5 (−29, 8.5)	9.5 (−11, 20)	6 (−18, 20)	−3.9 (−40, 5)	–	–
2011	7.1 (−4.5, 9.5)	13.2 (9, 21)	7.4 (2, 14)	−0.4 (−5, 2)	–	–
2012	8.0 (3, 12)	15.5 (8, 21)	6.6 (−2, 16)	−0.1 (−11, 2.5)	−22 (−45, −13)	−23.0 (−41, −16)
2013	6.1 (−12, 13)	12.8 (2, 19)	4.8 (−17, 14)	−3.0 (−32, 2.5)	−30 (−70, −16)	−29.6 (−30, −14)

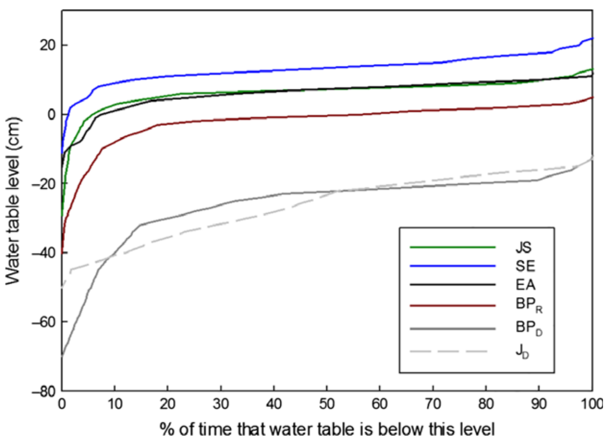

Fig. 3 Water table (cm) residence time curves for each microsite for the duration of the study. Microsite abbreviations are JS (*Juncus/Sphagnum*), SE (*Sphagnum/Eriophorum*), EA (*Eriophorum*), BP_R (bare peat rewetted), BP_D (bare peat drained) and J_D (*Juncus* drained).

(Table 2). Mean annual GAI decreased in 2012 and 2013 at JS (a decline of approximately 17% over the duration of the study) reflecting the increased dominance of *Sphagnum cuspidatum* (with little seasonal variation) and the decline of *Juncus effusus* in that microsite. The vascular plants (*Eriophorum angustifolium*) in SE were a very minor component of the microsite and their small decrease over the study was compensated for by a large increase in moss cover (see minimum values in Table 2). In the EA microsite, mean annual GAI values remained relatively constant between 2009 and 2013 although, intra-annual variation in the EA microsite was very high reflecting the large changes in GAI values (driven by a large vascular plant component) that occurred both spatially and temporally at that microsite (Table 2). In J_D, mean annual GAI was dominated by vascular plants (*Juncus effusus*) and a small decline was observed in mean values in the second year of the study (Table 2).

Table 2 Mean annual green area index (GAI, $m^2\ m^{-2}$) in the vegetated microsites for 2009–2013. Maximum and minimum GAI values observed in each year are also shown. Microsite abbreviations are JS (*Juncus/Sphagnum*), SE (*Sphagnum/Eriophorum*), EA (*Eriophorum*) and J_D (*Juncus* drained)

	GAI ($m^2\ m^{-2}$)				
Microsite	2009	2010	2011	2012	2013
JS					
Mean	1.21	1.20	1.13	1.02	1.00
Max.	1.83	1.81	1.84	1.06	1.02
Min.	0.99	0.99	0.99	0.96	0.96
SE					
Mean	0.94	0.95	0.99	1.03	1.01
Max.	1.17	1.19	1.17	1.06	1.06
Min.	0.52	0.53	0.52	0.95	1.00
EA					
Mean	0.93	0.97	0.92	0.94	0.94
Max.	2.05	2.10	2.06	2.05	2.04
Min.	0.06	0.05	0.06	0.06	0.05
J_D					
Mean	–	–	–	0.48	0.42
Max.	–	–	–	1.29	1.25
Min.	–	–	–	0.12	0.14

Seasonal patterns of carbon dioxide and methane exchange

The JS and SE microsites acted as a net CO_2 sink from January onwards (with the exception of 2012 where net uptake did not start in JS until March) (Fig. 4). However, in 2010 (late May) and 2013 (June) both microsites became a net CO_2 source and, with the exception of SE in 2013, remained a net CO_2 source for the rest of the year. In general, the EA microsite was a net CO_2 source at the start of each year and switched to a net CO_2 sink at different times in the spring over the 5 years of the study. Uptake of CO_2 was generally strong throughout the summer period (with the exception of 2010 and 2013) until approximately September when net uptake ceased in most years (Fig. 4). The bare peat sites were

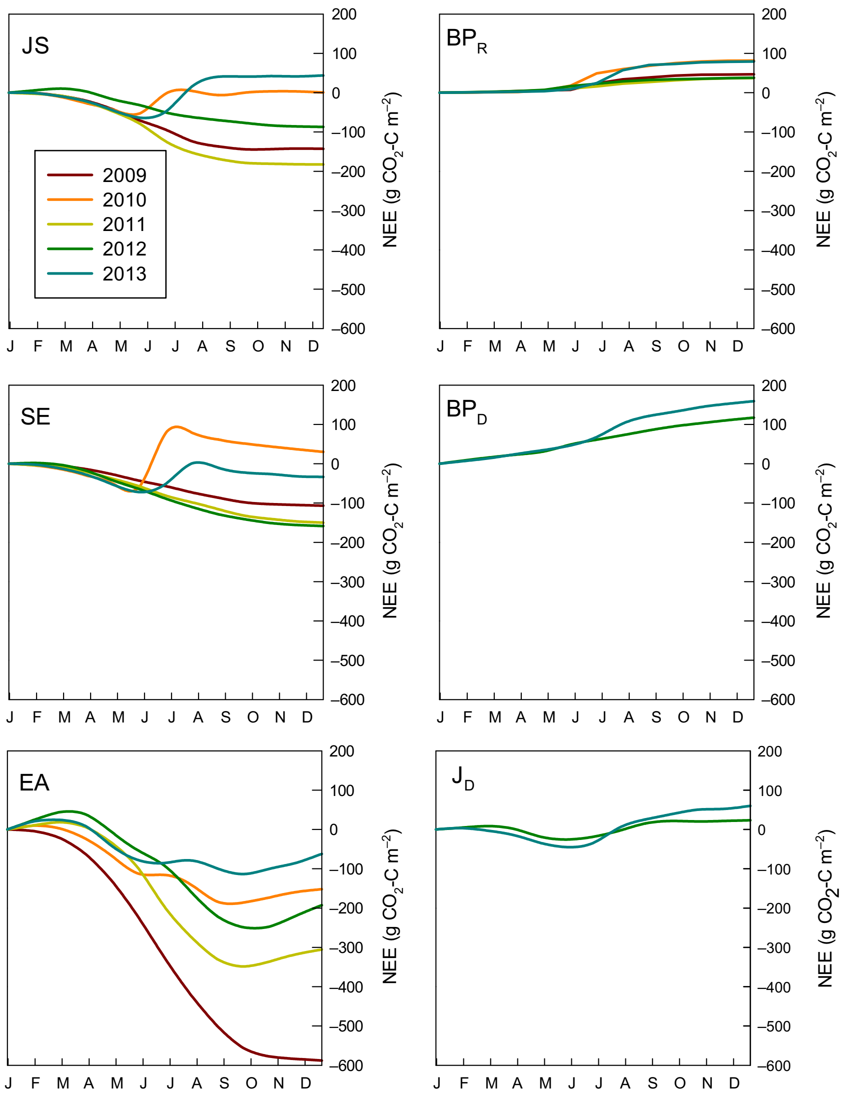

Fig. 4 Annual net ecosystem exchange (NEE; g CO_2-C m^{-2}) at the microsites at Bellacorick. Negative values indicate CO_2 flux from the atmosphere to the peatland (sink). Value at end of the curve indicates the total annual NEE value. Microsite abbreviations are JS (*Juncus/Sphagnum*), SE (*Sphagnum/Eriophorum*), EA (*Eriophorum*), BP_R (bare peat rewetted), BP_D (bare peat drained) and J_D (*Juncus* drained).

net CO_2 sources from the start of the year (Fig. 4). CO_2 emissions from BP_R remained very low until May in all years and then showed a moderate (2009, 2011 and 2012) or a sharp increase (2010 and 2013). In contrast, CO_2 emissions from BP_D increased steadily from the start of each year and increased rapidly in June 2013. The J_D microsite was a CO_2 sink for approximately 5 months of each year and a source for the remainder (Fig. 4).

Annual carbon dioxide and methane balances

The annual cumulative CO_2 balance (NEE) was generally negative (i.e. sink) in the rewetted vegetated sites and always positive (i.e. source) in the bare peat (both rewetted and drained) and J_D microsites (Fig. 4). Annual emissions from BP_R were reduced by over 50% in comparison with BP_D. In 2010, both JS and SE were a small annual source of CO_2, and JS was a larger source (~44 g C m^{-2} yr^{-1}) in 2013. In EA, uptake was reduced in both these years although the microsite remained a net CO_2 sink (Fig. 4). There was considerable spatial and temporal variation in the annual NEE with the highest uptake observed in EA in 2009 (588 g C m^{-2} yr^{-1}) and the highest emissions in BP_D in 2013 (159 g C m^{-2} yr^{-1}). At the rewetted site, the 5-year mean annual NEE ranged from 57 ± 30 g C m^{-2} yr^{-1} in BP_R to −260 ± 179 g C m^{-2} yr^{-1} in EA (Table 3). At

Table 3 Five-year mean annual gross primary production (GPP), ecosystem respiration (R_{eco}), net ecosystem exchange (NEE) and methane (CH_4) (g C m^{-2} yr^{-1}) values for the rewetted microsites and 2-year mean values for the drained microsites at Bellacorick. Standard errors are shown in parentheses. Microsite abbreviations are JS (*Juncus*/*Sphagnum*), SE (*Sphagnum*/*Eriophorum*), EA (*Eriophorum*), BP_R (bare peat rewetted), BP_D (bare peat drained) and J_D (*Juncus* drained). Discrepancies in number calculations are due to rounding-off artefacts

	CO_2			
Microsite	GPP (g C m^{-2} yr^{-1})	R_{eco} (g C m^{-2} yr^{-1})	NEE (g C m^{-2} yr^{-1})	CH_4 (g C m^{-2} yr^{-1})
BP_D	–	138 (11)	138 (11)	0
J_D	378 (21)	419 (8)	42 (23)	0
BP_R	–	57 (30)	57 (30)	0.1
JS	355 (16)	281 (65)	−74 (67)	8.7 (8)
SE	259 (24)	175 (100)	−84 (103)	11.2 (9)
EA	882 (64)	622 (167)	−260 (179)	5.3 (3)

the drained site, the 2-year mean annual NEE was 42 ± 23 and 138 ± 11 g C m^{-2} yr^{-1} in BP_D and J_D, respectively.

Considerable spatial variations in CH_4 emissions were observed at the study site (Fig. 5). The rewetted vegetated microsites were annual CH_4 sources in all years, with the highest annual values observed in SE (range from 8.2 to 14.6 g C m^{-2} yr^{-1}) and the lowest in EA (range from 5.1 to 5.4 g C m^{-2} yr^{-1}). Emissions from BP_R were very low (~0.1 g C m^{-2} yr^{-1}, data not shown) and CH_4 fluxes were not detected in either BP_D or J_D. 5-year mean annual CH_4 emissions were highest in SE (11.2 ± 9 g C m^{-2} yr^{-1}) and lowest in BP_R (0.11 g C m^{-2} yr^{-1}) (Table 3). N_2O fluxes were below the detection level of the equipment throughout the study period (data not shown).

Difference from the long-term average

Interannual variation in annual NEE was driven mainly by R_{eco} at the microsites (Fig. 6). The variation in annual values around the 5-year average (expressed here as % difference) showed that GPP typically varied between ±17% at the vegetated microsites. In contrast, the variation in R_{eco} was more pronounced at all microsites with the widest range (between −49 and +90%) observed at SE. With limited exceptions (mainly in JS), the divergence in GPP and R_{eco} within the same year were in the same direction (i.e. they were either both positive or both negative). Variation in CH_4 emissions around the 5-year average were highest in SE (±28%), intermediate in JS (−24 and +27%) and lowest in EA (−4 and +2%).

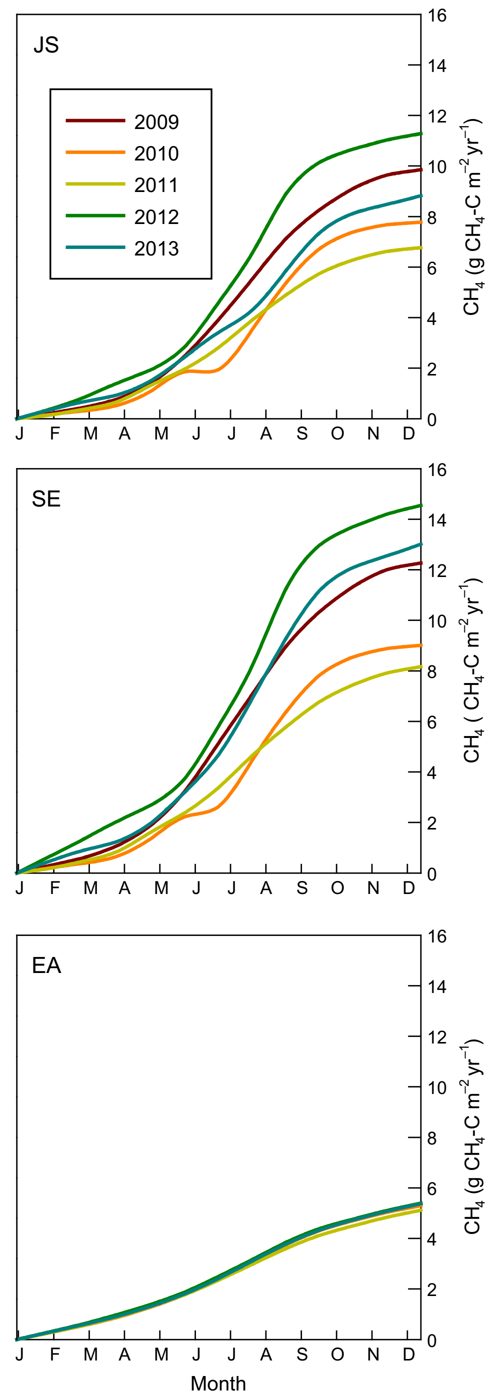

Fig. 5 Annual methane (CH_4) flux (g CH_4-C m^{-2} yr^{-1}) from the microsites at Bellacorick. Positive values indicate emissions to the atmosphere. Microsite abbreviations are JS (*Juncus*/*Sphagnum*), SE (*Sphagnum*/*Eriophorum*) and EA (*Eriophorum*).

Global warming potential and greenhouse gas mitigation

With the exception of EA, all microsites had a warming impact on the climate when the global warming potential of GHG fluxes and DOC were taken into account

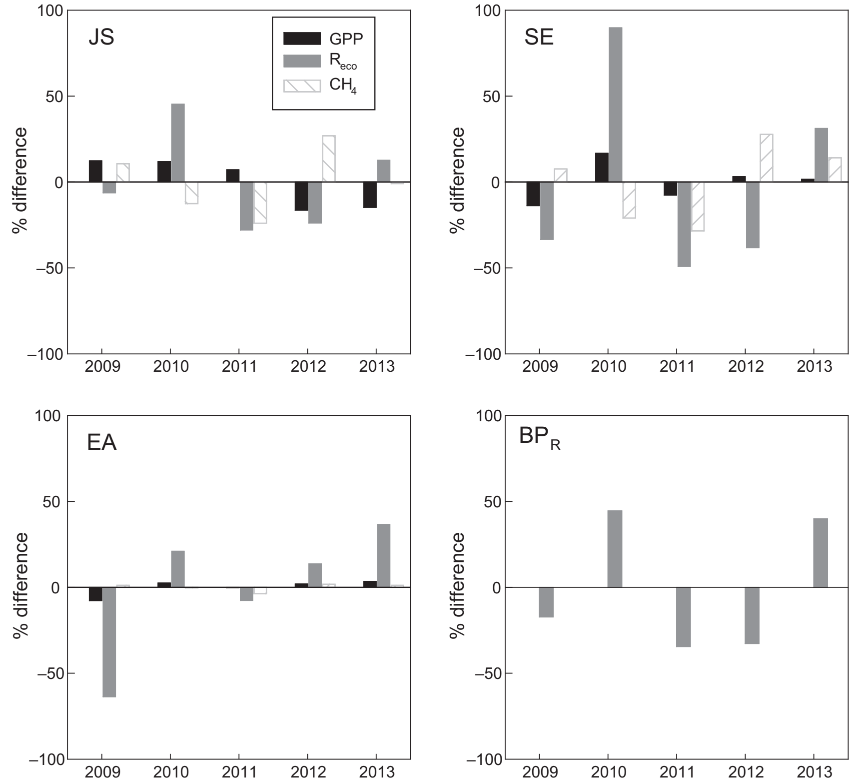

Fig. 6 Percentage (%) difference in annual gross primary production (GPP), ecosystem respiration (R_{eco}) and methane (CH_4) from the respective 5-year mean value (2009–2013) in the rewetted microsites at Bellacorick. Microsite abbreviations are JS (*Juncus/Sphagnum*), SE (*Sphagnum/Eriophorum*), EA (*Eriophorum*) and BP_R (bare peat rewetted).

(Table 4). The highest warming impact occurred in the drained BP_D microsite (7.12 t CO_2-eq ha^{-1} yr^{-1}) followed by J_D (3.60), SE (3.36), BP_R (3.02) and JS (2.11), respectively. EA had a large climate cooling impact (−6.25 t CO_2-eq ha^{-1} yr^{-1}) driven by very high CO_2 uptake and moderate CH_4 emissions. In the drained microsites, GWP was dominated by CO_2 emissions, with lesser amounts from CH_4 emissions from drainage ditches. In contrast, GWP in the rewetted sites JS and SE was dominated by CH_4 emissions.

Management actions (rewetting and/or recolonization) had a strong impact on GHG mitigation at the site. Recolonization of the drained bare peat (BP_D) substrate by *Juncus effusus* (J_D) resulted in an estimated GHG mitigation (i.e. avoided losses) of 3.52 t CO_2-eq ha^{-1} yr^{-1} and rewetting of the bare peat (BP_R) produced a saving of 4.1 t CO_2-eq ha^{-1} yr^{-1} (Table 4). The combination of rewetting and recolonization (i.e. all rewetted vegetated microsites) resulted in a GHG mitigation of between 3.8 and 13.4 t CO_2-eq ha^{-1} yr^{-1}.

Site comparisons

The upscaled emission factors (±SD) for the rewetted site were −104 ± 80 g CO_2-C m^{-2} yr^{-1} (Fig. 7) and 9 ± 2 g CH_4-C m^{-2} yr^{-1}. The CO_2 emission factor value is within the range of mean values reported for long-term natural and semi-natural sites, which ranged from −5 g CO_2-C m^{-2} yr^{-1} in Fäjemyr in Sweden to −114 g CO_2-C m^{-2} yr^{-1} in Forsinard, Scotland (Fig. 7). Interannual variation in NEE was considerably higher in Bellacorick (this study) than at any of the other long-term study sites (Fig. 7).When compared to other long-term intact sites within the same climate region (Fig. 8) for the same temporal period (2009–2013), Bellacorick was an annual CO_2 sink in 2009, 2011, 2012 and 2013, but a source in 2010. In contrast, the other sites (where data were available) were CO_2 sinks in all years (Fig. 8).

Discussion

The two main prerequisites for C sequestration in former drained and degraded peatlands are the re-establishment of hydrological conditions similar to those in intact ecosystems, and recolonization by vegetation (Smolders *et al.*, 2003). Rewetting at Bellacorick has resulted in an increase in the residence time for the water table above the soil surface: from 0% of the time in drained bare peat areas to 50% in rewetted bare peat

Table 4 Greenhouse gas balance (t CO_2-eq. ha^{-1} yr^{-1}) at the drained and rewetted microsites at Bellacorick. Dissolved organic carbon (DOC) and CH_4 emissions from drainage ditches (CH_{4ditch}) and were calculated using guidance provided by IPCC (2014). CH_4 and N_2O fluxes are converted to CO_2-eq. (t CO_2-eq ha^{-1} yr^{-1}) according to their global warming potential (GWP) on a 100-year timescale including climate –carbon feedbacks: CH_4 = 34 and N_2O = 298 (Myhre *et al.*, 2013). Positive values indicate a net warming impact on the climate and negative values indicate a net cooling impact. Intact sites at Glencar (Koehler *et al.*, 2011; McVeigh *et al.*, 2014) and Forsinard (Levy & Gray, 2015) are shown for comparative purposes. Microsite abbreviations are JS (*Juncus/Sphagnum*), SE (*Sphagnum/Eriophorum*), EA (*Eriophorum*), BP_R (bare peat rewetted), BP_D (bare peat drained) and J_D (*Juncus* drained)

	t CO_2-eq. ha^{-1} yr^{-1}					
Microsite	NEE	DOC	CH_{4soil}	CH_{4ditch}	N_2O	GWP
Drained						
BP_D	5.06	1.14	0	0.92	0	7.12
J_D	1.54	1.14	0	0.92	0	3.60
Rewetted						
BP_R	2.09	0.88	0.05	–	0	3.02
JS	−2.71	0.88	3.94	–	0	2.11
SE	−2.60	0.88	5.08	–	0	3.36
EA	−9.53	0.88	2.40	–	0	−6.25
Upscaled	−3.81	0.88	4.07	–	0	1.14
Natural						
Glencar	−2.04	0.46	1.86	–	0	0.28
Forsinard	−4.18	0.34	1.95	–	0	−1.89

areas (Fig. 3). Furthermore, the water table in the rewetted vegetated microsites was located above the soil surface for over 90% of the time. While this is primarily due to the postextraction rewetting measures at the site (i.e. ditch blocking, bund creation), it may also be partly caused by the water retention capacity of the vegetation, particularly *Sphagnum cuspidatum*, or it may be a reflection of the underlying topography of the residual peat (i.e. the artificial creation of hollows). The residence time values compare favourably with those reported elsewhere for restored (Wilson *et al.*, 2010) and intact peatlands (Holden *et al.*, 2011; Brown *et al.*, 2014; Malmer, 2014), and would indicate that the rewetting management actions have succeeded in capturing and containing the rainfall on the site.

Rewetting and the subsequent recolonization of the site have had a considerable impact on GHG emissions (Figs 4 and 5). The almost permanently inundated conditions at the rewetted site resulted in substantially reduced CO_2 emissions at the rewetted bare peat microsite (in comparison with the drained site) and high CO_2 sink values in the rewetted vegetated microsites (Table 3). The high WT precludes oxidation of both the residual (and highly recalcitrant) peat, and the more labile material that has accumulated over the last decade. Although the former is highly decomposed at this stage (Farrell & Doyle, 2003), CO_2 emissions remain significant when this peat is drained (see Table 3 and Fig. 4). In the rewetted microsites, approximately 10 cm of new organic material has amassed on top of the older peat in the decade since rewetting (data not shown), somewhat lower than the accumulation values reported for other rewetted/restored sites (Waddington *et al.*, 2003; Lucchese *et al.*, 2010). This discrepancy may be due to the fact that *Sphagnum cuspidatum* has flourished at this site postrewetting, has spread extensively at the expense of its nurse species (e.g. *Juncus effusus*,

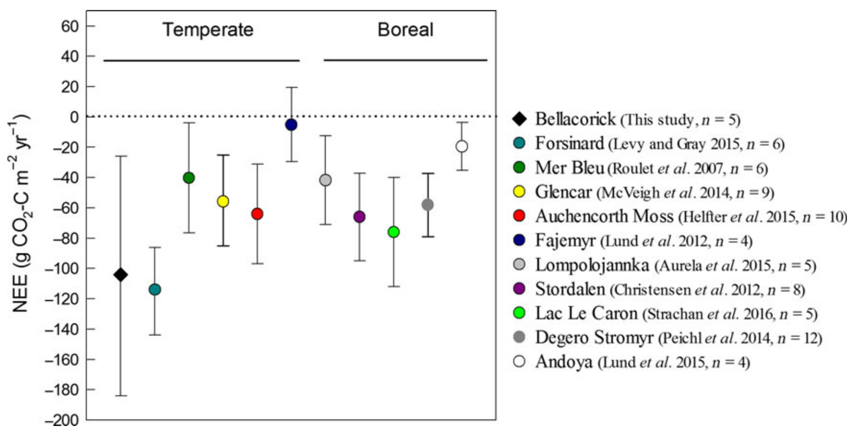

Fig. 7 Mean annual net ecosystem exchange (NEE, g CO_2-C m^{-2} yr^{-1}) at peatland sites with long-term (n > 4 years) annual carbon dioxide (CO_2) data sets. Error bars are standard deviations (SD) of the mean. Coloured symbols represent intact/semi-intact peatland sites and the black symbol represents the rewetted site in this study. Negative values indicate CO_2 flux from the atmosphere to the peatland (sink). n = number of measurement years at each site.

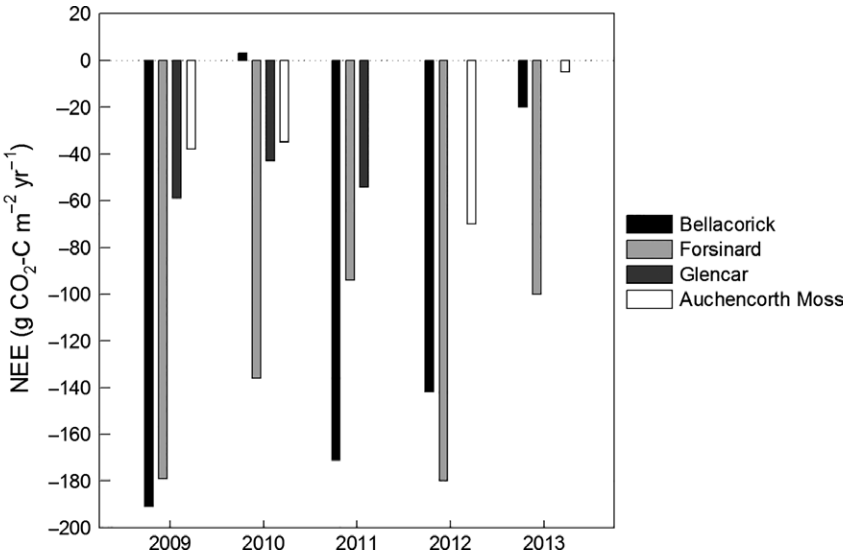

Fig. 8 Net ecosystem exchange (NEE; g CO_2-C m^{-2} yr^{-1}) for the period 2009–2013 for long-term carbon dioxide (CO_2) monitoring peatland sites in the Republic of Ireland and Scotland. Negative values indicate CO_2 flux from the atmosphere to the peatland (sink). No data were available for Glencar in 2012 and 2013 (see McVeigh *et al.*, 2014) and for Auchencorth Moss in 2011 (see Helfter *et al.*, 2015). Annual NEE values for Forsinard courtesy of P. Levy (pers. comm.).

Eriophorum angustifolium) in much of the rewetted areas and now covers over 60% of the rewetted site. Studies have demonstrated that *Sphagnum cuspidatum* litter readily decomposes (Hogg, 1993; Belyea, 1996), particularly the young stems (Limpens & Berendse, 2003). It is highly likely that the drop in the water tables in the summer of 2010 resulted in oxidation of some of this very labile material, and directly contributed to the higher R_{eco} values observed in that year in the JS and SE microsites.

The CO_2 emission factor of -104 ± 80 g CO_2-C m^{-2} yr^{-1} for Bellacorick compares well with the annual NEE values reported for intact sites (Fig. 7). Mean annual NEE values across long-term intact/semi-intact peatland monitoring sites appear relatively similar regardless of climate zone (mean of −58 and −52 g m^{-2} yr^{-1} for temperate and boreal peatlands, respectively), although the mean values from the Fäjemyr (Lund *et al.*, 2012) and Andøya (Lund *et al.*, 2015) sites are conspicuously lower than the others (Fig. 7). Upscaling GHG flux values measured at the microsite scale to provide an estimate of emissions/removals at the ecosystem level is a challenge (Gray *et al.*, 2013). However, a strong relationship between GHG exchange and vegetation communities has been reported in numerous studies (e.g. Levy *et al.*, 2012; Urbanová, 2012; Wilson *et al.*, 2013) and novel approaches using vegetation communities as 'proxies' for GHG exchange have been adopted for upscaling purposes (Couwenberg *et al.*, 2011). Key to such approaches is the availability of reliable GHG flux data and sufficiently detailed landscape-scale vegetation surveys (Gray *et al.*, 2013).

Interannual variation in NEE between the intact/semi-intact sites also appear comparable (Fig. 7), which undoubtedly reflects the mature developmental stage of these ecosystems and their resilience to some extent to variations in climatic factors (McVeigh *et al.*, 2014; Peichl *et al.*, 2014). Annual NEE values at these sites are influenced by a wide range of factors, such as rainfall (Levy & Gray, 2015), high vapour pressure deficit (Lund *et al.*, 2015), winter air temperature (Peichl *et al.*, 2014; Helfter *et al.*, 2015) and water table depth (McVeigh *et al.*, 2014; Helfter *et al.*, 2015; Strachan *et al.*, 2016). In contrast, IAV in NEE at our site is much greater as evidenced by the very wide range in annual NEE values at Bellacorick (+3 to −191 g CO_2-C m^{-2} yr^{-1}) when compared to the next most variable intact site at Auchencorth Moss (−6 to −165 g CO_2-C m^{-2} yr−1; Helfter *et al.*, 2015). This large variation may be indicative of an ecosystem that is very much in transition in terms of vegetation composition, water retention capacity and GHG dynamics in general. While our study site experiences the same mild, oceanic climate and growing season length as the long-term monitoring sites at Glencar, south-west Ireland (McVeigh *et al.*, 2014), Forsinard (Levy & Gray, 2015) and Auchencorth Moss in Scotland (Helfter *et al.*, 2015), differences in annual NEE values are evident when values are compared across temporally similar periods (Fig. 8). Over the same monitoring period as this study (2009–2013), mean NEE values at the intact/semi-intact sites (where data were available) were all negative (i.e. CO_2 sinks). In contrast, Bellacorick was a strong CO_2 sink in 2009, 2011 and 2012, and a modest sink in 2013 and a small CO_2 source in 2010 (Fig. 8), which would indicate that site-specific differences account for the disparity in annual values between these sites.

The IAV in annual CO_2 exchange at our rewetted microsites was largely determined by annual R_{eco} (Fig. 6), which in turn, was driven by soil temperature and WT (S1 and Table S2). In contrast, Lund *et al.* (2010) in a study of 12 intact peatland sites across a number of climate zones observed that variability in R_{eco} was lower than GPP in those sites and partly attributed this to the presence of a high water table that suppressed decomposition. Similarly, Estop-Aragonés *et al.* (2016) found that GPP in moss species in a mesocosm study (peat from intact/semi-intact sites) was more sensitive to drought conditions than R_{eco}. At Bellacorick, the amplitude in annual R_{eco} values between years (Fig. 6) dictated the strength of the annual CO_2 source/sink function at this site. For 2010, the higher annual R_{eco} values were caused by a combination of a number of factors: lower-than-average rainfall (particularly during the early growing season), a subsequent drop in water table levels (Table 1), elevated soil temperatures (Fig. 2a) and the availability of labile organic matter. While inundation may act as a buffer during times of drought (Quinty & Rochefort, 2003; Money *et al.*, 2009), it was not sufficient to prevent increased CO_2 emissions at the rewetted microsites during the summers of 2010 and 2013 (Fig. 4).

Most CO_2 balance estimations on rewetted peat soils thus far have covered either a single growing season (e.g. Soini *et al.*, 2010; Strack *et al.*, 2014), multiple growing seasons (e.g. Tuittila *et al.*, 1999; Koebsch *et al.*, 2013; Urbanová *et al.*, 2013), a single year (e.g. Strack & Zuback, 2013; Knox *et al.*, 2015) or a 2-year period (e.g. Wilson *et al.*, 2007b; Beetz *et al.*, 2013; Günther *et al.*, 2015). As such, it is difficult to put the large IAV in NEE at this site in perspective, given the lack of comparable long-term rewetting studies. However, Dixon *et al.* (2014) measured CO_2 fluxes at eight rewetted and degraded blanket bog sites over a 5-year period (they did not estimate an annual CO_2 balance) and observed that while IAV was significant, the effect of site on CO_2 fluxes (both GPP and R_{eco}) was more important.

The upscaled annual CH_4 emissions at Bellacorick were slightly higher than emissions from comparable intact blanket bog sites (Laine *et al.*, 2007; Levy & Gray, 2015), but similar to annual emission factors derived for rewetted temperate nutrient-poor organic soils (IPCC, 2014, Wilson *et al.*, 2016). CH_4 emissions from peatlands in general have been shown to have high spatial variability (e.g. Wilson *et al.*, 2009; Couwenberg *et al.*, 2011; Moore *et al.*, 2011; Strack & Zuback, 2013; Koch *et al.*, 2014) and this was also evident in this study as annual emissions differed substantially between drained and rewetted sites, and between rewetted microsites (Table 3 and Fig. 5). The strong seasonal variation in CH_4 emissions at the rewetted site at Bellacorick has been highlighted by Wilson *et al.* (2013) and is in agreement with other studies on both intact (e.g. Shurpali & Verma, 1998; Olson *et al.*, 2013) and rewetted peatlands (Tuittila *et al.*, 2000a; Renou-Wilson *et al.*, 2016). Multiyear investigations of CH_4 fluxes have also provided evidence for strong IAV, in many cases driven by variations in temperature, the frequency and distribution of precipitation (Shurpali & Verma, 1998; Waddington & Day, 2007; Olson *et al.*, 2013) and changes in the position of the water table (Moore *et al.*, 2011). However, while IAV in CH_4 emissions is a strong feature in the JS and SE microsites in this study, it is less obvious in the EA microsite (Fig. 5), which is somewhat surprising given the water table drawdown evident at that microsite in 2010 and 2013 (Table 1). However, a statistically significant relationship between WT and CH_4 emissions in the EA microsite was not evident, and emissions were instead controlled to a large extent by GAI (Eq. 8, S1 and Table S2), which varied considerably between seasons but was broadly similar between years (Table 2).

We did not detect N_2O fluxes in either the rewetted or the drained sites and similar findings have been reported for rewetted organic soils in the temperate climate zone (IPCC, 2014, Wilson *et al.*, 2016). N_2O production is controlled by environmental factors, such as substrate, redox conditions and temperature (Müller & Sherlock, 2004; Butterbach-Bahl *et al.*, 2013). Under prolonged saturated conditions, the redox potential is lowered (Tauchnitz *et al.*, 2015) and this directly impacts on microbial processes within the soil and limits N_2O production (Rubol *et al.*, 2012). Drained organic soils used for peat extraction have been shown to be a small annual N_2O source (IPCC, 2014), which contrasts with the observations in this study. However, nitrogen deposition in the west of Ireland is very low (average ammonia concentration <0.5 μg m^{-3} (UCD 2015), and the residual peat at Bellacorick is inherently nutrient-poor (pH 3.8, C : N 58). N_2O production is likely to be constrained by nitrogen availability in the soil (Carter *et al.*, 2012), particularly as any excess nitrogen is likely to be taken up by the vigorous growth of vegetation communities. Nonetheless, given the sampling approach used for N_2O in this study (i.e. at monthly intervals), it may be possible that we missed 'hot moments' of emissions (McClain *et al.*, 2003; Butterbach-Bahl *et al.*, 2013), particularly when water table drawdown occurred (Tauchnitz *et al.*, 2015) in the summers of 2010 and 2013

(Wilson *et al.*, 2013), or that the sampling times (20–40 min) were insufficient to capture very low fluxes.

The use of GWP as a tool to assess the climatic impact of GHG emissions or removals over the lifetime of a peatland can be problematic (Frolking *et al.*, 2006; Bridgham *et al.*, 2014), although it has been widely utilized as a means of directly comparing the effects of management actions on GHG exchange (e.g. Maljanen *et al.*, 2010; Beetz *et al.*, 2013; Herbst *et al.*, 2013; Wilson *et al.*, 2013; Beyer *et al.*, 2015). Annual GWP values in peatlands typically display strong IAV as they are calculated from highly variable annual GHG fluxes. Long-term studies, however, provide the basis for a more accurate assessment of the climate impact of management actions, as the IAVs are averaged out to some extent over the duration of the study (Beetz *et al.*, 2013). At Bellacorick, the action of rewetting produced a significant reduction in the GWP in comparison with the drained sites (Table 4). When combined with recolonization, GWP values were further reduced (i.e. a smaller warming impact), particularly when the fluxes were integrated at the higher spatial level (Table 4), where the very high CO_2 sink values associated with the EA microsite compensated to some extent for the CH_4 emissions from the more widespread SE microsite. Although, the GWP values at the rewetted site remain somewhat removed from the values estimated for intact blanket bogs in the same climate region (Table 4), it should be noted that rewetting and recolonization have only taken place over the last decade or so at our site and further changes in vegetation composition and GHG exchange dynamics are highly likely in the decades ahead. Indeed, it may be possible that given suitable conditions the rewetted site could stabilize over time and become similar to intact peatlands in terms of GHG dynamics.

Long-term GHG monitoring of intact peatland sites provides robust baseline data that can be used to monitor the effects of climate change (Wu & Roulet, 2014). While a number of the long-term study sites in Fig. 7 appear relatively insensitive to moderate weather IAV (e.g. McVeigh *et al.*, 2014; Peichl *et al.*, 2014; Levy & Gray, 2015), other experimental studies have shown that under more 'extreme' conditions of climate warming, even intact sites may become net C sources (e.g. Welker *et al.*, 2004; Chivers *et al.*, 2009; Laine *et al.*, 2009a; Wu & Roulet, 2014). As such, given that rewetted peatlands may potentially be more sensitive to changes in interannual weather patterns and to long-term climate change than their intact counterparts, it is important that future research be also directed towards long-term monitoring of these new ecosystems to provide critical information for land managers, policymakers and other stakeholders.

Acknowledgements

The project was funded by the Environmental Protection Agency, Ireland, under the Climate Change Research Programme (Project codes: 2007-CCRP-1.6 and 2012-B-MS-9) and by Bord na Móna, Ireland. We thank Dr Peter Levy and Dr Mika Aurela for providing additional data for Forsinard and Lompolojänkkä, respectively, and Dr Carole Helfter for useful discussions. We particularly thank the two anonymous reviewers for their insightful comments and suggestions.

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Supporting Information

Additional Supporting Information may be found in the online version of this article:

Data S1. (a) Gross primary production (GPP), (b) ecosystem respiration (R_{eco}) and (c) methane (CH_4) models.
Table S2. Model parameters and goodness of fit (r^2).