Rewetting degraded peatlands for climate and biodiversity benefits: results from two raised bogs

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Abstract

Globally, peatlands are under threat from a range of land use related factors that have a significant impact on the provision of ecosystem services, such as biodiversity and carbon (C) sequestration/storage. In Ireland, approximately 84 % of raised bogs (a priority habitat listed in Annex I of the EU Habitats Directive) have been affected by peat extraction. While restoration implies the return of ecosystem services that were characteristic of the pre-disturbed ecosystem, achieving this goal is often a challenge in degraded peatlands as post-drainage conditions vary considerably between sites. Here, we present multi-year greenhouse gas (GHG) and vegetation dynamics data from two former raised bogs in Ireland that were drained and either industrially extracted (milled) or cut on the margins for domestic use and subsequently rewetted (with no further management). When upscaled to the ecosystem level, the rewetted nutrient poor domestic cutover peatland was a net sink of carbon dioxide (CO_2) (-49 ± 66 g C m^{-2} yr^{-1}) and a source of methane (CH_4) (19.7 ± 5 g C m^{-2} yr^{-1}), while the nutrient rich industrial cutaway was a net source of CO_2 (0.66 ± 168 g C m^{-2} yr^{-1}) and CH_4 (5.0 ± 2.2 g C m^{-2} yr^{-1}). The rewetted domestic cutover site exhibited the expected range of micro-habitats and species composition found in natural (non-degraded) counterparts. In contrast, despite successful rewetting, the industrially extracted peatland did not exhibit typical raised bog flora. This study demonstrated that environmental and management variables can influence species composition and, therefore, the regeneration of species typical of natural sites, and has highlighted the climate benefits from rewetting degraded peatlands in terms of reduced GHG emissions. However, rewetting of degraded peatlands is a major challenge and in some cases reintroduction of bryophytes typical of natural raised bogs may be more difficult than the achievement of proper GHG emission savings.

Keywords: Climate, greenhouse gases, restoration, species diversity

1. Introduction

Globally, peatlands and organic soils are under threat from a range of climatic and land use related factors that have a significant impact on the provision of ecosystem services, such as biodiversity (Lachance and Lavoie, 2004; Parish et al., 2008; Pasquet et al., 2015) and carbon (C) sequestration and storage (Beetz et al., 2013; Evans et al., 2014; Renou-Wilson et al., 2014; Tiemeyer et al., 2016).

In natural (i.e. not degraded) peatlands, permanently waterlogged conditions prevent the complete decomposition of dead plant material, leading to the accumulation of C rich peat (Page and Baird, 2016). Natural peatlands are generally net sinks for carbon dioxide (CO_2 uptake) and sources of methane (CH_4) emissions (Christensen et al., 2012; Roulet et al., 2007) and this interplay in gas exchange over millennia has played a major role in the regulation and maintenance of the global climate (Frolking et al., 2006). Furthermore, natural peatlands are an important global biodiversity resource as they provide niche habitats for a range of specialized fauna and flora (Chapman et al., 2003; Minayeva et al., 2017; Parish et al., 2008).

Global emissions from land use change (e.g. deforestation, afforestation) are estimated at 3.67 Gt CO_2 yr^{-1} (Le Quéré et al., 2016), however this value does not include emissions associated with the drainage of peatlands and organic soils, or from peat fires (Houghton et al., 2012), which could potentially release an additional 0.9-3 Gt of CO_2 to the atmosphere annually (Joosten et al., 2012; Smith et al., 2014). Furthermore, land use change has also led to the loss of 15 % of global peatland habitats over the last century (Barthelmes, 2016).

In Ireland, loss of peatland habitat through conversion to grassland, forestry and from peat extraction (industrial and domestic peat extraction) is estimated at 85 % of the national resource (Malone and O'Connell, 2009) and the most recent national monitoring survey showed that 84 % of raised bogs (a priority habitat listed in Annex I of the EU Habitats Directive (EU Directive on the Conservation of Habitats, Flora and Fauna 92/43/EEC), have been affected by peat extraction alone (NPWS, In press). Moreover, CO_2 emissions from degraded Irish peatlands and from related peat activities (combustion of peat for energy, horticulture) have been estimated at c. 11 to 12.4 Mt CO_2 each year (Barthelmes, 2016; Wilson et al., 2013b), which is similar to annual emissions from the Irish transport sector (Duffy et al., 2015). Critically, only 1,955 ha of raised bog in Ireland (~0.1 %) is considered ‘active’ i.e. capable of C sequestration and continued peat formation (Fernandez et al., 2014).

Rewetting of peat soils (i.e. by ditch blocking, creation of bunds etc.) has been suggested as an important climate change mitigation tool to reduce emissions, to create suitable conditions for C sequestration and to stimulate biodiversity (Bonn et al., 2014; Parish et al., 2008). However, a wide array of annual CO_2 (e.g. Knox et al., 2015; Renou-Wilson et al., 2016; Wilson et al., 2016b; Wilson et al., 2007b; Yli-Petäys et al., 2007), CH_4 (e.g. Günther et al., 2015; Strack and Zuback, 2013; Vanselow-Algan et al., 2015) and N_2O (e.g. Dinsmore et al., 2009; Minke et al., 2016; Ye and Horwath, 2016) fluxes have been reported for rewetted peatlands, largely driven by differences between climate zones (e.g. boreal, temperate and tropical) and nutrient status (Wilson et al., 2016a). Other factors such as time since rewetting, current vegetation composition and previous land use management are also likely to have a significant influence on GHG dynamics post-rewetting, although the extent of these effects may be highly site specific.

Rewetting and additional restoration measures have also the potential to re-instate the original ecosystem services, such as water conservation, erosion and fire prevention; and co-benefits, such as enhanced biodiversity value of peatlands (Parish et al., 2008; Tanneberger and Wichtmann, 2011), which themselves would contribute to climate change adaptation. For example, a specific Aichi Biodiversity Target (i.e. Target 15) to combat climate change and biodiversity loss by the restoration of 15 % of degraded ecosystems by 2020 has been proposed by the Convention for Biological Diversity (CBD, 2010) with rewetting and restoration of peatlands put forward as a specific instance of such a win-win outcome. While the biodiversity-climate change nexus is now well recognized (Parish et al., 2008), studies pertaining to the restoration/rewetting of degraded peatlands still focus mostly on one single objective (mainly due to the fact that these are in protected areas) and do not demonstrate this potential synergy (Andersen et al., 2016).

In this study, we appraised the climatic benefits from the rewetting of degraded peatlands in terms of reduced GHG emissions (from multi-year data sets) and the possible return of the C sequestration function, as well as the potential for increased biodiversity provision with the aim of identifying the most optimum land use category for future peatland restoration policies. We investigated two rewetted peatlands (originally raised bogs) within the same biogeographical region but under different land uses: industrial peat extraction (which accounts for 7 % of the peatland resource) vs drained/domestic peat extraction (38 % of the peatland resource). We assessed whether rewetting of these degraded peatlands can bring back the biological structure and ecosystem functions typical of a natural raised bog. We hypothesise that both sites are currently on dissimilar successional stages and will, therefore, exhibit contrasting biodiversity and climate benefits.

2. Material and methods

2.1 Study areas

Blackwater, Co. Offaly is located in the Irish Midlands (Supplemental Material (S1)) and is a former raised bog that was drained for milled peat extraction in the late 1950s. At the cessation of extraction in 1999, the drainage pumps were deactivated. Since then, the landscape has developed spontaneously as a composite of microsites; bare peat fields (some with active drainage systems), and re-colonized drained and rewetted areas (Fallon et al., 2012). The general hydrological and nutrient status of Blackwater is representative and typical of the outcome of industrial peat extraction in Ireland (Farrell, 2008), and due to the geographical location in the Irish midlands (a flat low-lying basin), rewetting of these sites usually leads to the formation of shallow lakes. The average thickness of the residual peat layer within the study site is 150 cm and the peat is composed mainly of highly humified *Phragmites* peat overlying limestone parent material (Table 1).

Moyarwood, Co. Galway is located in the west of the country (S1) and was ditched (similar drainage network to Blackwater) in 1983 but never developed for peat extraction. However, significant degradation (subsidence) due to the drainage and from domestic peat extraction around the margins of the bog and from burning events is evident. The topography is varied but relatively flat on the eastern side where the GHG monitoring plots were established. The drains were active until a rewetting programme commenced in 2012, which consisted of blocking drains with peat dams at regular intervals (generally at any point where there was a fall in a drain level of 10 cm). A drained vegetated area remains on the margins of the bog. The average thickness of the peat within the site is 440 cm and the peat is composed mainly of humified *Sphagnum* peat overlying limestone parent material (Table 1).

2.2 Climate

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 and moderately high annual rainfall. Although there is only 30 km distance between the sites (S1), the 30-year mean annual air temperature (1980-2010) is slightly lower at Blackwater (9.6 °C) strongly affected by lower winter temperatures in the area, while mean annual precipitation is 245 mm higher in Moyarwood (Table 1). During the study, potential evapotranspiration (PE; mm) ranged from 482 to 551 mm at Blackwater and from 502 to 517 mm at Moyarwood (Note: 30-year mean annual PE values are not available for the Met stations at Gurteen (Blackwater) and Athenry (Moyarwood)).

Table 1 Location, climate and physico-chemical characteristics* of the two rewetted sites. Air temperature and precipitation data (1981-2010) from Met Éireann, Gurteen station (Blackwater) and Athenry station (Moyarwood).

Site name	**Blackwater**	**Moyarwood**
Previous land use	Industrial peat extraction (30 years)	Drained/domestic peat extraction (30 years)
Time of rewetting	1999	2012
Latitude	53°17´48.9´´N	53° 20´49.2´´N
Longitude	7°57´56.3´´W	8°30´55.4´´W
Sub-region	Midlands	West
Mean annual air temperature (°C)	9.6	9.9
Mean precipitation ($mm\ yr^{-1}$)	948	1193
Current peat type	*Phragmites*	*Sphagnum*
von Post scale	H7	H6
Parent material	Limestone	Limestone
Acrotelm	Absent	Present but subsided
Peat depth (m)	1.5	4.4
pH	4.9	4.4
Electrical conductivity ($\mu s\ cm^{-1}$)	350	102
Bulk density ($g\ cm^{-3}$):		
- drained	0.19	0.13
- rewetted	0.14	0.08
C (%)	52.4	51.5
N (%)	2.14	1.32
C:N	24.5	39
Study period	1/5/2011: 30/4/2015	1/4/2013: 31/3/2015
Number of GHG plots:		
-drained	3	3
-rewetted	11	12

*See Renou-Wilson et al. (2018) and related technical report for details regarding soil sampling.

2.3 Field measurements

2.3.1 Biodiversity assessment

The hydro-ecological state of each site was assessed on the basis of a habitat assessment, survey of plant communities and indicator species (positive and negative). Four key abiotic and biotic components of biodiversity assessment were identified for the appraisal of the restored/rewetted status of each site, namely 1) hydrological integrity (see details in Renou-Wilson et al., 2018); 2) physico-chemical parameters of peat and drain water (pH, C:N, bulk density, electrical conductivity); 3); micro-habitat assessment (heterogeneity and condition); and 4) vegetation composition (species and abundance).

The relative presence of different micro-habitat types (hummocks, pools, hollows, lawns and flats) provide a measure for habitat heterogeneity in typical raised bogs (Schouten, 2002). In order to carry out the assessment of the biotic components, five habitat quadrats "HQ" (4 x 4 m plot) were established along a "W" shape transect through representative areas of each site, running perpendicular to the main drainage systems. In each HQ, the vegetation was described by identifying the percentage cover of the main plant functional types (PFT): woody vegetation; ericoid dwarf shrubs, total graminoids (grasses, sedges, forbs), bryophytes (Sphagnum mosses, other mosses, liverworts), lichens (demonstrating absence of burning events), litter and bare peat. Four peat samples and the nearest drain water were also sampled for analysis. Furthermore, the four corners of each HQ formed a vegetation quadrat "VQ" (1 x 1 m) and, therefore, 20 vegetation quadrats were also identified at each site in order to assess the fourth component of our survey method. All taxa of vascular plants, mosses and lichens and their cover values were recorded at each VQ. Algae were recorded but not identified. Cover of each species within a VQ was estimated using a revised Domin scale (Kent and Coker, 1992). The scale includes cover values from 0 to 4, whereby 0=absent, 1=rare (<5 %), 2=occasionally (5-20 %), 3=frequently (21-50 %) and 4=dominant (>50 %).

Species richness and the Shannon-Wiener Index were determined as indicators of vegetation diversity for each site and were both determined as mean values of the 20 VQs. Species richness was further determined in total species number, total vascular plant species number and total bryophyte species number per site. Dominant PFTs and species for each study site were determined on the basis of the respective cover value medians per VQ. Cover-weighted means of Ellenberg Indicator Values (EIV) were calculated from the VQ data for soil moisture, acidity and nitrogen (Ellenberg and Leuschner, 2010). Indicator values that were unknown or not considered in Ellenberg's publication were based on Hill et al. (1999) or Hill et al. (2007). Nomenclature for vascular plants follows Parnell and Curtis (2012), Atherton et al. (2010) for bryophytes, and Whelan (2011) for lichens.

2.3.2 Environmental monitoring

Weather stations (WatchDog Model 2400; Spectrum Technologies Inc., Aurora, IL, USA) were established at each 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 at each site to capture potential variation in soil temperatures between drained and rewetted microsites and recorded hourly soil temperatures (°C) at 5, 10 and 20 cm depths. Water table levels (WT) were manually measured from perforated dipwells (internal diameter 2 cm) that were inserted adjacent to each GHG measurement collar at fortnightly or monthly intervals and linearly interpolated between measurement days to provide continuous data (Alm et al., 2007; 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. At Blackwater, the boardwalks on the northern side of the collars were elevated (~1 m from the ground) to facilitate the installation of extension chambers over the *Phragmites* plants.

2.3.3 GHG monitoring

Blackwater: In 2011, 15 square stainless steel collars (60 x 60 cm) with a channel at the top were inserted to a depth of 30 cm into the peat in the following microsites: reeds (*Phragmites australis;* n=8), sedges (*Carex rostrata, Eriophorum angustifolium*; n=3) and drained bare peat (n=3). Although open water accounts for approximately one-third of the rewetted area, gas monitoring plots were not established there for logistical reasons.

Moyarwood: A total of 15 collars were established in early 2013 with 12 located in the rewetted area along a transect perpendicular to the ditches. One collar in the rewetted transect was suspended within a ditch, supported on wooden batons that extended across the ditch. The 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. As per IPCC guidance (IPCC, 2014), former ditches were not considered as being separate from the remainder of a rewetted site, and in this instance the collar located in the ditch was considered part of the transect. The vegetation in the collars was representative of the habitats found in the wider site, i.e. flats with some Sphagnum lawn and a few hummocks. Three monitoring plots were located in the drained area on the eastern margin, which was dominated by poorly growing heather (*Calluna vulgaris*) and lichens (*Cladonia* spp.). Collars were not established in drainage ditches in the drained areas at either site for logistical reasons; instead we use the IPCC default emission factors (IPCC, 2014) for ditches in drained peatlands where appropriate.

Greenhouse gas fluxes were measured at fortnightly or monthly intervals from May 2011 to April 2015 at Blackwater and from April 2013 to March 2015 at Moyarwood. Each measurement campaign consisted of 2-4 days. Net ecosystem exchange (NEE) was measured with a static transparent polycarbonate chamber (60 x 60 x 33 cm) equipped with two internal fans to ensure mixing of the headspace air. Vent holes on the chambers ensured that pressure artefacts and ebullition (for methane) were minimised during chamber placement. At Blackwater, extension chambers (Günther et al., 2015; Wilson et al., 2007b) equipped with fans were used for collars that contained *Phragmites australis*. 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., 2007), and PPFD was recorded from a sensor (PAR-1; PP Systems, King's Lynn, Norfolk, UK) located in the chamber. NEE was measured under a range of light levels (PPFD; µmol m^{-2} s^{-1}) 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 µmol m^{-2} s^{-1}). CO_2 measurements were carried out between 8 am and 6 pm in the summer and between 9 am and 3 pm in the winter to ensure that the maximum PPFD was reached at each measurement date. Ecosystem respiration (R_{eco}) was then measured by covering the chamber in an opaque cover and CO_2 exchange was measured as outlined 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 campaigns in summer) using an opaque, polycarbonate chamber (60 x 60 x 25 cm) equipped with a battery-operated fan that mixed the air within the chamber headspace. As for CO_2 sampling, extension chambers were used in the reed microsite at Blackwater. For a more detailed description of GHG sampling approach and laboratory analysis see Wilson et al. (2013a).

To incorporate the seasonal dynamics of the vegetation into CO_2 exchange models, a green area index (GAI) was estimated for each of the vegetated collars. In brief, this involved measuring the green photosynthetic area of all vascular plants (leaves and stems) within five sub-sample plots (8 cm x 8 cm) in the GHG collar 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 (see Wilson et al., 2007a)

2.3.4 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 for 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., 2007). 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 were evaluated using Akaike's Information Criterion for small sample sizes (AICc) and fluxes that exhibited low AICc values (representing lower variance and better model fitting) were accepted. In this study, we followed the sign convention whereby positive values indicate a 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., 2007) and the closest R_{eco} 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.

2.3.5 Modelling

Statistical and physiological response models were constructed and parameterized for each microsite within the study sites (see S2). 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). During model construction, the relationship between R_{eco}, GPP, CH_4 or N_2O 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 (S2a). 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 and Taylor, 1994) and are nonlinear models related to soil temperature and water table or volumetric moisture content (S2b). The CH_4 models are nonlinear models related to soil temperature and water table (S2c).

2.3.6 Annual GHG balances

The response functions estimated for R_{eco}, GPP and CH_4 were used for the reconstruction of the annual GHG balances for the drained and rewetted areas. Fluxes were reconstructed for each sample plot in combination with an hourly time series of (1) T_{5cm}, (2) WT levels linearly interpolated from weekly measurements, (3) PPFD values recorded by the weather station and (4) plot-specific modelled GAI that described the phenological development of the vegetation. Annual NEE was calculated as the sum of annual GPP (negative values) and annual R_{eco} (positive values). Annual balances (g C m^{-2} yr^{-1}) were calculated for each sample plot by integrating the hourly values over each 12-month period. Annual CO_2 balances from the drained areas in both sites were previously reported in Wilson et al. (2015) and here we provide an additional year of CO_2 data from both sites.

2.3.7 Upscaling

High resolution habitat maps were created for each site. A vegetation and habitat mapping survey was carried out 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 recorded. Once the field work was completed, the maps were digitised using Arc Map10 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. At Blackwater, the proportion of reeds, sedges and open water in the rewetted site were 13, 57, and 30 % respectively. At Moyarwood, the vegetation was relatively homogenous across the site and so the mean annual CO_2 and CH_4 balances from the collars were scaled up to an area of one hectare.

2.3.8 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 Wilcoxon signed-rank nonparametric tests were used. Uncertainty in reconstructed annual R_{eco} and NEE was calculated by summing up the maximum and minimum standard errors (Renou-Wilson et al., 2014).The species abundance data of rewetted plots in Blackwater and Moyarwood and intact raised bogs were analyzed with multivariate statistics using Canoco 4.5. The initial detrended correspondence (DCA) analysis revealed a gradient on the first axis that was longer than 5 SD units. Therefore, the unimodal DCA model of species distribution along gradients was applied.

2.3.9 Synergies

The outcome of rewetting / restoration activities are very site specific, which makes valuation of the ecosystem services provided by these sites a challenge (Glenk et al., 2014). Nevertheless, a robust valuation is needed if current land use management and alternative uses are to be compared for local, regional or national policy development, where it is critical to simplify the links between functional processes, such as the condition of the vegetation (composition and cover) and habitats, and the related ecosystem services. We assigned a ranking (1-5) to six metrics (measured in this study) under the categories of climate change mitigation and biodiversity provision. The criteria used to determine the rankings are defined in Table 2. Scaling within the rankings was assigned based on the range in values generally reported in the scientific literature for each metric, although we are cognisant that there is a level of subjectivity to the way that they are used here. For comparative purposes, a ranking of 5 is commensurate to a natural raised bog in most of the metrics. For CH_4, a ranking of 3 is more applicable for a temperate natural site (e.g. Goodrich et al., 2015; Green and Baird, 2017).

Table 2. Criteria used to determine rankings (1-5) for the decision matrix. Positive values for carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O) indicate net losses from the peatland to the atmosphere and negative values (< 0) indicate net uptake by the peatland.

Metric	**Units**	**Ranking**				
		1	**2**	**3**	**4**	**5**
Climate change						
CO_2	g C m^{-2} yr^{-1}	> 90	90- 61	60-31	30-1	≤ 0
CH_4	g C m^{-2} yr^{-1}	> 30	30-21	20-11	10-1	≤ 0
N_2O	g N m^{-2} yr^{-1}	> 1	0.9 to 0.61	0.6-0.31	0.3-0.1	≤ 0
Biodiversity						
Micro-habitat diversity	-	0 to 0.5	0.6 to 1	1.1 to 2	2.1 to 3	> 3
Bryophyte no.	-	<3	3 to 4	5 to 6	7 to 8	> 9
Shannon Wiener Index	-	0 to 0.99	1 to 1.99	2 to 2.99	3 to 3.99	>4

3. Results

3.1 Weather

A distinct seasonal pattern in air and soil temperatures was evident in both sites throughout the study (Fig. 1). At Blackwater, the lowest air temperature values (-7 °C) occurred during the winter of 2012/2013 (Fig. 1a), with highest air temperatures (29 °C) recorded in mid-summer 2013. Average annual air temperatures for each 12-month period were 9.8, 8.7, 10 and 9.7 with year 2 and 3 varying considerably from the long-term average (Table 1). At Moyarwood, the lowest air temperature (-4 °C) was recorded in April 2013 and the highest (29 °C) in mid-summer 2013 (Fig. 1b). Soil temperatures at both sites displayed the same seasonal patterns as air temperatures but were subject to less hourly variation in both the drained and rewetted areas.

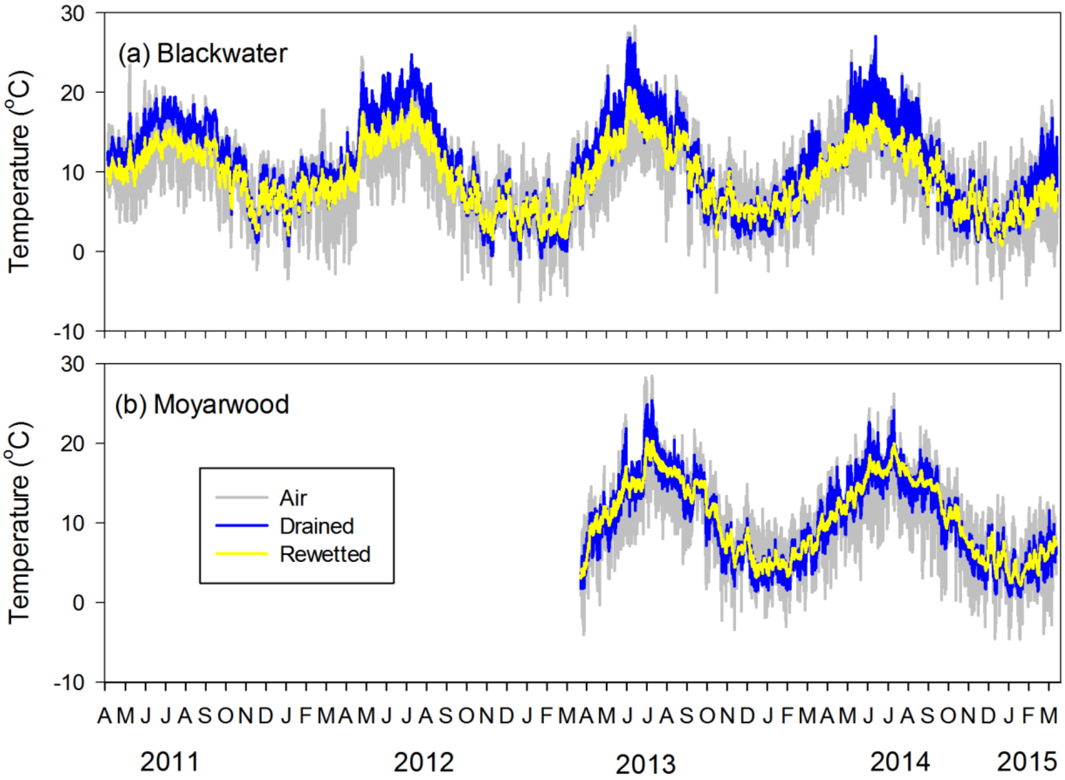

Fig. 1 Hourly air (grey line) and soil temperatures (°C) at (a) Blackwater and (b) Moyarwood for the duration of the study. Soil temperatures are from 5cm depth in the drained (blue line) and rewetted (yellow line) areas of the study sites.

At Blackwater, rainfall for each 12-month period; 949 mm, 1049 mm, 949 mm and 926 mm was similar to the long term average (Table 1), although rainfall distribution varied considerably between years with the summer of 2013 much drier than the others (Fig. 2). At Moyarwood, the rainfall in the first year of the study (1425 mm) was 19 % higher than the long-term average, driven by wetter summer and winter conditions, while the second year (1193 mm) was similar to the long-term average.

3.2 Hydrological and physico-chemical parameters

Both rewetted sites displayed a similar moisture profile with the water table at or just below the ground surface (Fig. 2). Water table levels displayed spatial and temporal variability over the monitoring periods (Fig. 2). All the rewetted plots displayed annual mean WT levels above -20 cm throughout the study and are confirmed as hydrologically restored. In 2013, the WT in the rewetted areas in Blackwater dropped below -20 cm for 3 months (Fig. 2a), although a similar response was not observed in Moyarwood at the same time (Fig. 2b). The WT in the drained areas of both sites were subject to considerable fluctuations throughout the study period with the deepest WT levels observed in Blackwater (-80 cm) in autumn 2013 (Fig. 2a).

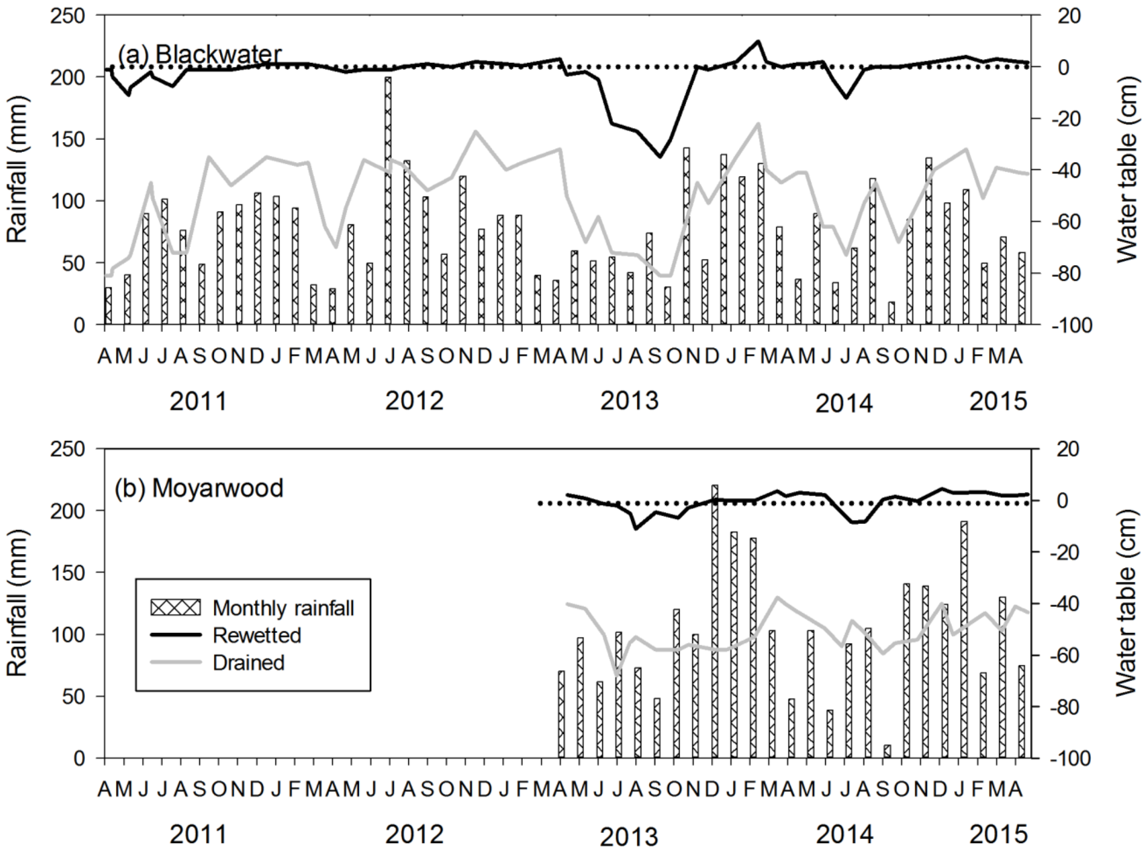

Fig. 2 Monthly rainfall (mm; hatched bars) and water table levels (cm) in the rewetted (black line) and drained (grey line) areas in (a) Blackwater (April 2011 to April 2015) and (b) Moyarwood (April 2013 to April 2015).

Blackwater had a pH of approximately 5 and a high von Post value associated with *Phragmites* peat (Table 1). This mesotrophic status is due to its basal fen peat now exposed to the surface after peat extraction and which is also more nutrient rich (C:N = 25). This was corroborated by the analysis of surface water that showed high pH (7.4) and electrical conductivity (316 µs cm^{-1}) values. In comparison, the surface layer of Moyarwood was more acidic (pH 4.4) and nutrient poor (C:N = 39) with oligotrophic surface water of low pH (4.3) and low electrical conductivity (59.8 µs cm^{-1}).

3.3 Habitat and vegetation profile

Blackwater lacked micro-habitat heterogeneity as no typical raised bog micro-habitats were recorded in the surveyed part of the site. Moyarwood displayed a range of typical micro-habitats (mean of 2.4 types of habitats, such as hummocks, hollows, lawns, and flats per habitat quadrat) but had no pools (Table 3). The proportion of vascular and bryophyte species differed between sites and was not necessarily reflected in the total species number (Table 3). Moyarwood had three times more bryophytes than Blackwater but the latter had the highest number of total species (27 versus 21). The species spread also varied as shown by the ratio of 'species number per plot: total species number': Blackwater displayed a low ratio (18 %) indicating that some species occurred only together. At Moyarwood, 50 % or more of all species occurred at all plots. The Shannon Wiener Index was higher at Moyarwood (2.26) than at Blackwater (1.34).

Table 3 Biodiversity variables, dominant Plant Functional Types (PFT), and dominant species recorded at both rewetted sites. Micro-habitat diversity: mean of different types present per quadrat; Ellenberg Index Values (EIV) for moisture, acidity and nitrogen are unit less.

Site name	**Blackwater**	**Moyarwood**
Micro-habitat diversity	0	2.4
Total species	27	21
Bryophyte species	3	9
Bryophyte/total species ratio	0.11	0.43
Shannon Wiener Index	1.34	2.26
Dominant PFTs in habitat quadrats	Sedges Litter	Sedges Ericoids/Sphagnum
Dominant species in vegetation quadrats	*Carex rostrata* *Phragmites australis*	*Calluna vulgaris* *Sphagnum capillifolium*
EIV moisture	8.4	8.0
EIV acidity	4.5	2.0
EIV nitrogen	3.5	1.5

Woody species, ferns and algae were absent in Moyarwood and only detected in very low numbers in Blackwater. The most dominant plant functional types (PFTs) at both sites were 'sedges' but with different combination of species occurring, namely *Eriophorum vaginatum and E. angustifolium* with *Carex panicea* in Moyarwood compared to the sedge combination of *Carex rostrata* and *E. angustifolium* in Blackwater. The second most dominant PFTs in Moyarwood were ericoids and *Sphagnum* mosses (in particular *Sphagnum capillifolium*) while plant litter was detected in high amounts at Blackwater. While very common moss species, such as *Polytrichum commune* and *Campylopus introflexus* regularly occurred in the more degraded Blackwater site, the fen-specialized bryophyte *Campylium stellatum* was also present. Although not recorded in the quadrats, *Sphagnum subnitens* was observed on very small patches and is known to colonize wet cutaway peatlands after sedges and rushes. At Moyarwood, hummocks were widespread but not very large and were associated with species such as *S. capillifolium, S. magellanicum, S. papillosum* and *Hypnum cupressiforme*. Another indicator species of healthy natural raised bogs is *S. fuscum,* which was found on one hummock. The average EIVs across all vegetation plots for Moyarwood indicated that the site was wet, quite acidic and infertile. In contrast, Blackwater had higher moisture, acidity and nitrogen EIVs demonstrating that while being very wet, the peat was more nutrient-rich, akin to a 'fen'.

At Moyarwood, species indicative of drier past conditions, for example an abundant cover of *Narthecium ossifragum*, *Trichophorum caespitosum* and *E. vaginatum*, while *C. panicea*, a negative indicator particularly in true "Midlands raised bogs" occurred on all vegetation quadrats. Of great significance was the absence of common alien (invasive) species at both sites. Invasive species are considered to be the main direct drivers of biodiversity loss in Europe and a threat to natural habitats in Ireland in particular (Caffrey et al., 2014). Lichens (mostly *Cladonia portentosa*), an indicator of the absence of fire events, were present at Moyarwood but not Blackwater. When comparing our results with typical natural raised bogs that were located in the same region of the studied sites (data extracted from Renou-Wilson et al., 2011), we can assess the success or failure of each site vis-à-vis ultimate objectives of rewetting/restoration. The radar graphics in Fig. 3 display the mean cover of PFTs for both sites compared with a typical natural site. Sphagnum mosses, ericoid and sedges formed a typical assemblage of PFTs in Moyarwood in similar proportion to a natural site, except for the higher cover of sedges and the presence of non-Sphagnum mosses. The PFT profile at Moyarwood had the closest resemblance with that of a natural raised bog. In contrast, Blackwater vegetation demonstrated a clear divergent pattern with 'sedges/rushes' as dominant PFT.

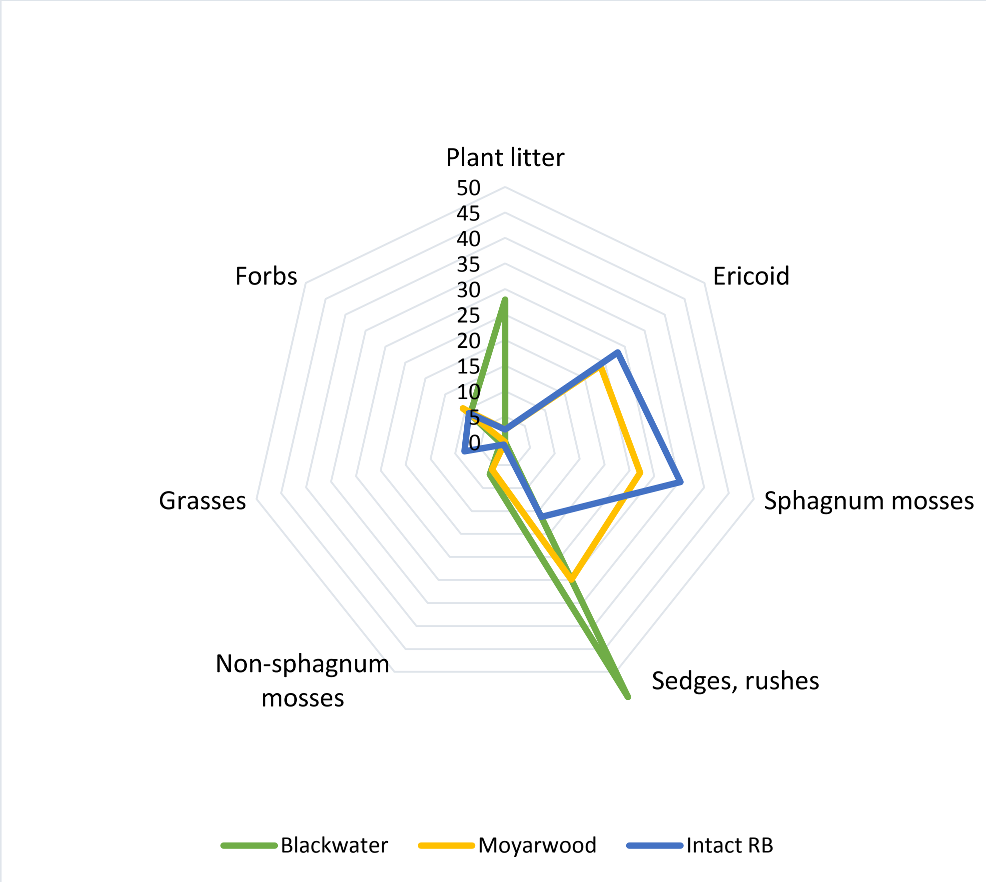

Fig. 3 Spider plot of Plant Functional Types percentage cover recorded at an intact Raised Bog (RB) and at Blackwater and Moyarwood.

The multivariate analysis of the species abundance data via DCA resulted in a sum of all eigenvalues of 4.752 (Axis 1: 0.903, Axis 2: 0.432). The gradient length of axis 1 was 5.315 and 3.327 for axis 2. The cumulative percentage variance of species data for the first four axis was: 19.0, 28.1, 33.3 and 36.9. In the DCA biplot of species and samples (Fig. 4), the high similarity and homogeneity of species composition between rewetted Moyarwood plots and intact raised bogs was clearly observable from the overlapping sample cloud and the dominance of *Sphagnum species*. The rewetted Blackwater plots showed a higher heterogeneity in species composition, which was in parts dominated by *P. australis*. The distance of the Blackwater samples to the next intact raised bog sample was greater than 2-5 SD units, showing medium to high dissimilarity of the species composition between them.

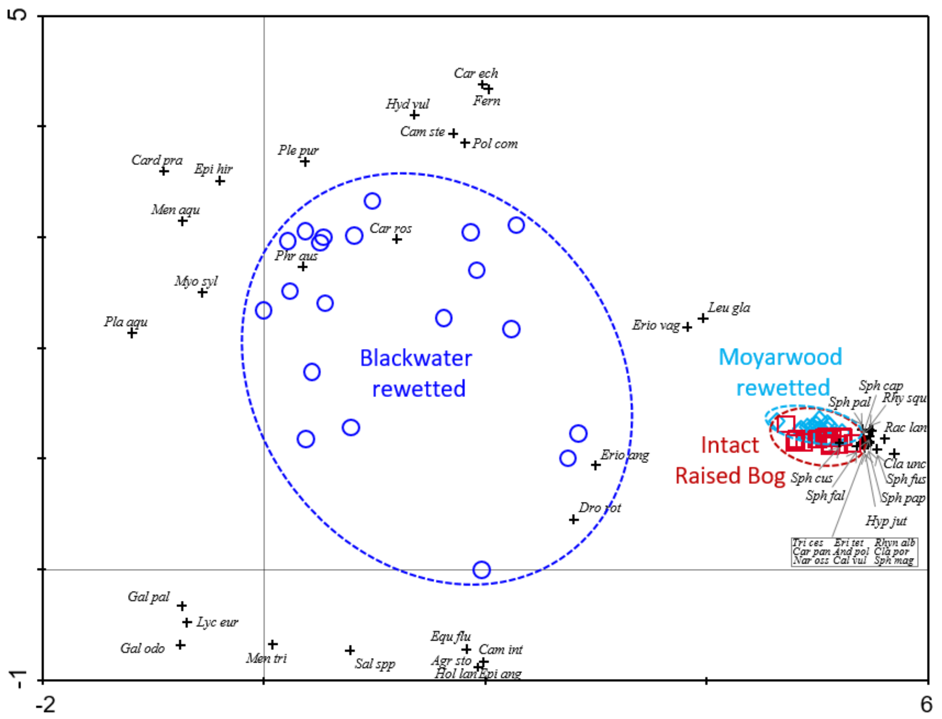

Fig. 4 Detrended correspondence analysis (DCA) biplot of species and sample plots at both rewetted sites and an intact raised bog. Sum of all eigenvalues of 4.752 (Axis 1: 0.903, Axis 2: 0.432).

3.4 Green area index (GAI)

At Blackwater, average (± standard deviation) GAI values in the GHG collars ranged from 0 (non-growing season) to 4.6 ± 1.1 $m^2 m^{-2}$ in the reeds, and from 0 to 0.8 ± 0.2 $m^2 m^{-2}$ in the sedge microsite (Fig. 5a). The drained area at Blackwater was unvegetated. There were noticeable differences between years in regard to the start and finish of the growing season, and the maximum GAI value observed in mid-summer (Fig. 5a). The amount of green plant surface available for photosynthesis over the growing season, as determined by the area under the GAI curves, varied considerably between years and followed the trend: 2011>2012>2013>2014. This trend was also evident in the sedge microsite. At Moyarwood, the shape of the GAI curves and average GAI values were similar in both years in the drained and rewetted areas (Fig. 5b), and ranged from 0.6 to 0.9 ± 0.3 $m^2 m^{-2}$ in the former and from 0.3 to 0.8 ± 0.4 $m^2 m^{-2}$ in the latter.

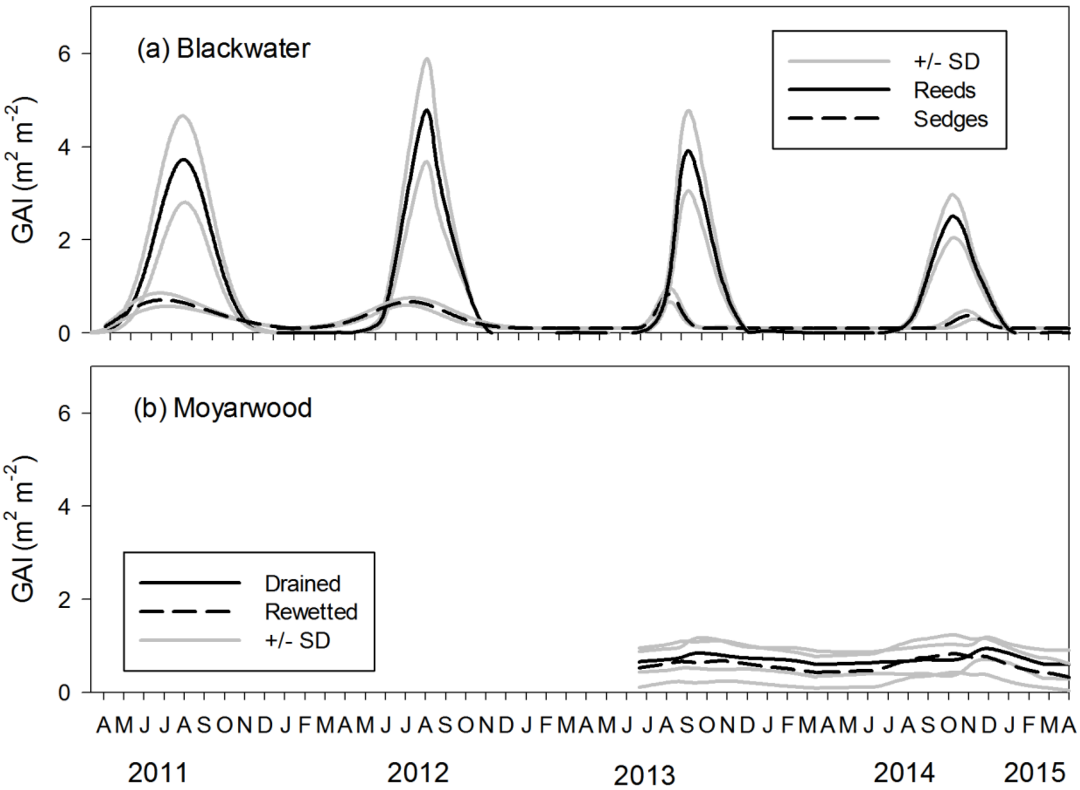

Fig. 5 Average modelled green area index (GAI; m^2 m^{-2}) in (a) Blackwater and (b) Moyarwood. Grey lines indicate ± standard deviation (SD) of the mean.

3.5 Annual GHG balances

3.5.1 Carbon dioxide fluxes

The drained areas in both sites were a net annual CO_2 source in all years of the study (Fig. 6a and 6d). At Blackwater, CO_2 emissions (in the absence of a vegetation component R_{eco} and NEE are analogous) were highest in year 3 (185 ± 8 g C m^{-2} yr^{-1}) and lowest in year 2 (111 ± 5 g C m^{-2} yr^{-1}). The 4-year average (± standard error) was 151 ± 11 g C m^{-2} yr^{-1}. At Moyarwood, R_{eco} was higher than GPP in both years in the drained area and resulted in net CO_2 losses of 115 ± 26 g and 159 ± 23 g C m^{-2} yr^{-1} in years 1 and 2 respectively. The 2-year average was 137 ± 24 g C m^{-2} yr^{-1}.

The reed microsite at Blackwater was a very strong CO_2 sink in the first two years of the study; -271 ± 92 and -255 ± 118 g C m^{-2} yr^{-1} (Fig. 6b). In year 3, R_{eco} increased by 45 % and GPP dropped by 24 % from the values estimated in the previous year. This produced a very strong CO_2 source of 248 ± 86 g C m^{-2} yr^{-1}. In year 4, both R_{eco} and GPP decreased and the microsite was a smaller CO_2 source of 129 ± 124 g C m^{-2} yr^{-1}. Average NEE (4-year) for the reed microsite was -37 ± 229 g C m^{-2} yr^{-1}. The sedge microsite followed the same trend as the reed microsite; a small CO_2 sink in years 1 and 2, and a strong CO_2 source in years 3 and 4 (Fig. 6c). Average NEE (4-year) for the sedge microsite was 90 ± 68 g C m^{-2} yr^{-1}. Average NEE for the rewetted area (excluding open water) was estimated at 66 ± 168 g C m^{-2} yr^{-1}.

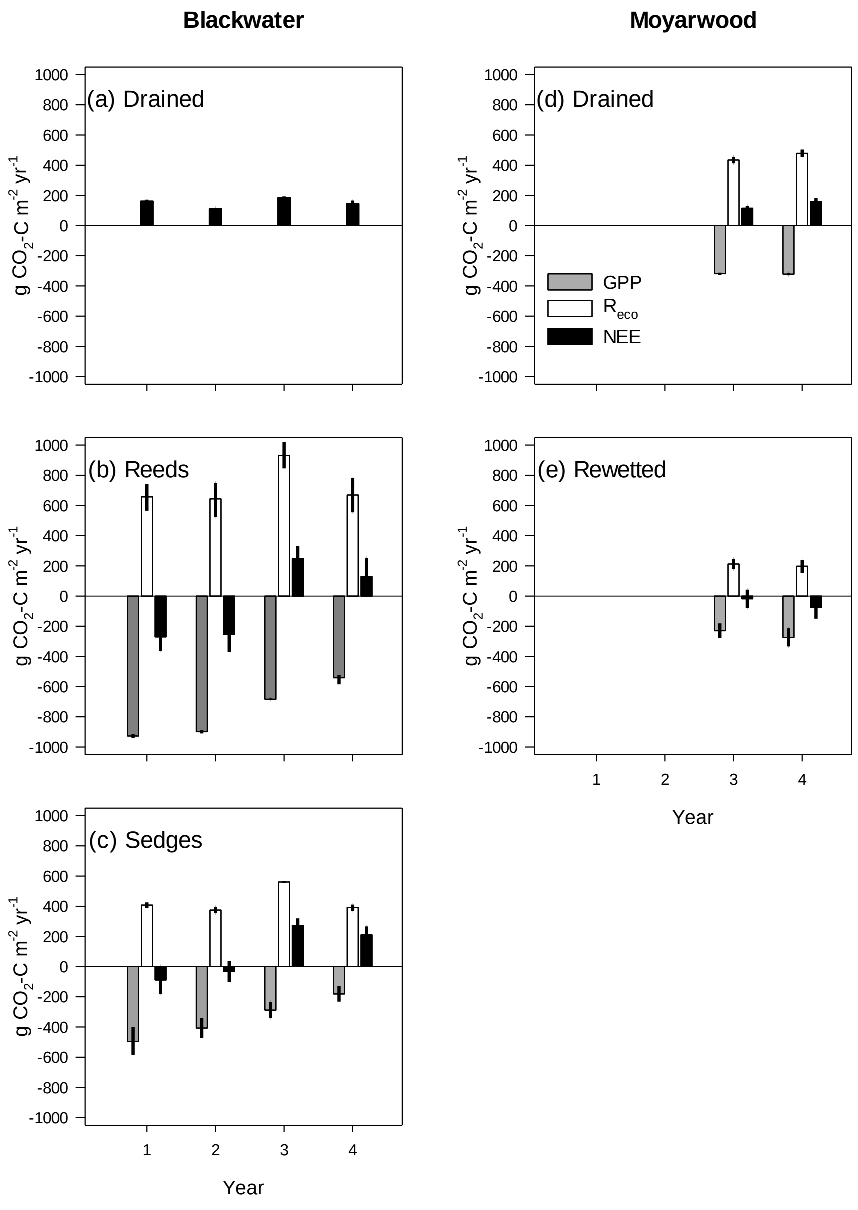

Fig. 6 Average annual modelled net ecosystem exchange (NEE), gross photosynthesis (GPP*)* and ecosystem respiration (R_{eco}) and (g C m^{-2} yr^{-1}) ± standard errors in Blackwater (left side panels) and Moyarwood (right side panels). Positive values indicate net loss of carbon dioxide (CO_2) from the peatland to the atmosphere and negative values indicate net CO_2 uptake by the peatland.

At Moyarwood, the rewetted area was a CO_2 sink in both years of the study (Fig. 6e). R_{eco} was similar between years, although GPP was 20 % higher in the second year. Average NEE (2-year) for the rewetted area was estimated at -49 ± 68 g C m^{-2} yr^{-1}.

3.5.2 Methane and nitrous oxide fluxes

Methane fluxes at the drained area in Blackwater were below the detection level of our equipment, so a value of zero was assigned (Fig. 7). Both rewetted microsites in Blackwater were annual CH_4 sources in all years of the study (Fig. 7). Emissions ranged from 6.7 to 9 g C m^{-2} yr^{-1} in the reed microsite and 4.2 to 4.5 g C m^{-2} yr^{-1} in the sedge microsite. Average CH_4 emissions for the upscaled rewetted area at Blackwater (excluding open water) were estimated at 5.0 ± 2.2 g C m^{-2} yr^{-1}. At Moyarwood, average annual CH_4 emissions (linearly interpolated between measurements) in the drained area were 0.77 g ± 0.49 g C m^{-2} yr^{-1} (Fig. 7). Average CH_4 emissions for the rewetted area were 19.7 ± 5 g C m^{-2} yr^{-1}. Nitrous oxide (N_2O) fluxes were below the detection level of the equipment throughout the study period and were therefore assigned a value of zero.

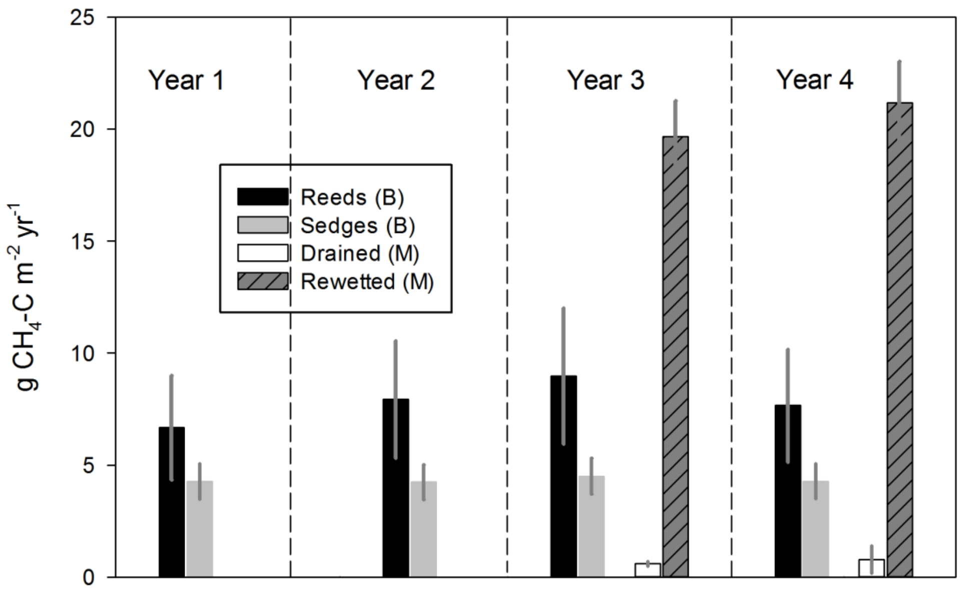

Fig. 7 Average annual modelled CH_4 fluxes (g C m^{-2} yr^{-1}) ± standard error in Blackwater (B) and Moyarwood (M). Methane fluxes at the drained area in Blackwater were below the detection level of our equipment, so a value of zero was assigned.

3.5.2 Synergies

In the decision matrix, climate change benefits from rewetting were slightly higher at Moyarwood where the site was ranked at 5 (maximum) for both CO_2 and N_2O (Table 4). For CH_4, it ranked lower than Blackwater reflecting the disparity in annual CH_4 emissions between the two sites. However, in term of biodiversity provision, Moyarwood was over twice as high as Blackwater, in particular for micro-habitat diversity and bryophyte numbers. Overall, Moyarwood ranked 1.5 times higher than Blackwater.

Table 4 Decision matrix for ecosystem services; climate change mitigation and biodiversity provision for the Blackwater and Moyarwood sites. See Table 2 for criteria used to determine rankings.

	Climate change mitigation				**Biodiversity**				
Site/Service	CO_2	CH_4	N_2O	Sub-total	Micro-habitat diversity	Bryophytes species	Shannon Wiener Index	Sub-total	**Overall total**
Blackwater	2	4	5	11	1	2	2	5	**16**
Moyarwood	5	3	5	13	4	5	3	12	**25**

4. Discussion

Restoration has been defined as “assisting the recovery of an ecosystem that has been degraded, damaged or destroyed” and can be viewed as a *process* and/or an *outcome* (McDonald et al., 2016); both with well-defined goals (Rochefort, 2000). In terms of the latter, the widest possible range of ecosystem services is re-established and the “new” ecosystem looks and functions similar to a historical pre-disturbed/degraded ecosystem (i.e. the target reference state), although a future driven focus has also been proposed (e.g. Martin-Ortega et al., 2017). However, in peatlands where much of the peat has been extracted (e.g. the raised bogs used for industrial peat extraction), this goal is impossible to achieve (at least in the short- and medium term) and the development of a fen ecosystem is a more feasible option (e.g. Cobbaert et al., 2004; Rochefort et al., 2016). Clearly, it is important to establish at the outset the criteria by which success will be evaluated and to thereby determine the position of a particular site on a restoration trajectory (Andersen et al., 2016). While cognisant that other key components of peatland functioning, such as microbial community structure (e.g. Urbanová et al., 2011), faunal composition (e.g. Hannigan et al., 2011), water quality (Martin-Ortega et al., 2017) and water borne carbon exports (e.g. Waddington et al., 2008), for example, were not quantified in this study, the two sites in this study, nonetheless, provide a valuable case study in regard to peatland restoration given that they were both raised bogs prior to disturbance (of varying intensity) but have plainly developed along fundamentally different restoration trajectories with dissimilar hydrological regimes, vegetation composition and GHG dynamics.

4.1 Hydrological regime

From a hydrological standpoint both sites have been successfully rewetted, as ascertained by the water table remaining at, just below or above the surface for the majority of the study period (Fig. 2), as well as by the high EIV moisture values associated with the current vegetation (Table 3). Species assemblages that form active raised bog require a mean water level near or above the surface of the bog lawns for most of the year (Robroek et al., 2007). In addition, seasonal fluctuations should not exceed 20 cm in amplitude and should only be 10 cm below the surface except for very short periods of time (NPWS, In press). Overall, the two sites have been successfully 're-plumbed' and both the "passive" rewetting at Blackwater (i.e. cessation of drain pumping) and the more active approach at Moyarwood (i.e. blocking of drains) have created a wet environment similar to the pre-disturbed ecosystem.

In terms of water chemistry, the electrical conductivity values varied between sites with Moyarwood displaying values similar to a nutrient poor raised bog (Kettridge et al., 2011; Van der Schaaf et al., 2002), while the electrical conductivity values at Blackwater are more consistent with nutrient rich peatlands (Van der Schaaf et al., 2002). The pH of both sites are comparable to natural sites, although the soil pH at Blackwater is at the lower range of the mesotrophic category (Jeglum, 1971), similar to natural fens (Doyle and O' Críodáin, 2003). This is to be expected as (a) Blackwater is fed by relatively alkaline water influenced by the limestone parent material beneath the peat and (b) the presence of residual fen peat following extensive extraction of the more acidic upper peat layers.

4.2 Species diversity

Species diversity is typical of the natural counterparts: typically low in the less degraded site at Moyarwood and with a higher number of species recorded in Blackwater due to the water influx. The presence of monodominant stands such as *P. australis* and the absence of a well-developed bryophyte layer at Blackwater signify a transition mire (from open water to fen). If the site remains wet, the development of a new ‘moderately rich’ ecosystem is the most likely trajectory with an increased species of mosses. This may not happen naturally, however, given the lack of seed banks in the locality and the difficulties in reintroducing fen bryophytes in particular (Rochefort et al., 2016). Following the Canadian experience on similar sites, intervention may be required with a combination of mechanized diaspore transfer and fertilization with phosphorus (the limiting nutrient in such sites and present in very low amounts in Blackwater (0.033 %) (F. Renou-Wilson, unpublished data). It may also be possible to transfer specific *Sphagnum* spp. since they are known to expand at higher pH levels amongst fen communities if precipitation remains high (Vicherová et al., 2017).

On the other hand, increased frequency of dry events, whereby the water table drops below -20 cm may be a further obstacle to bryophyte establishment as grasses and forbs will compete with the existing Cyperaceae and Juncaceae species; typical species of degraded bogs that are present in the surrounding areas. This continued increase in species is not desirable from a biodiversity perspective (diversity of biological information) since homogenization is at play (i.e. where most of the new species found in a habitat are also found in other ecosystems in the locality).

The presence of *Sphagnum capillifolium* at Moyarwood is of high significance as it indicates the presence of water table levels close to the surface and above. The presence of other *Sphagnum* species such as *S. papillosum* and *S. magellanicum* strongly indicates that vegetation succession at this site is on the correct trajectory. The type of *Sphagnum* species present is also critical for certain ecosystem processes, such as peat formation. For example, studies have shown that *Sphagnum cuspidatum* litter readily decomposes (Belyea, 1996; Hogg, 1993), particularly the young stems (Limpens and Berendse, 2003). The Moyarwood site has recovered a palette of *Sphagnum* species that has permitted the return of the C sequestration function as well as increased biodiversity. The latter was also supported by the presence of other raised-bog specialized species (e.g. the carnivorous *Drosera spp.*) confirming the return of wet and acidic environmental conditions but also demonstrates the increased conservation value of the site, as these species do not occur in any other ecosystems on the island.

4.3 GHG dynamics

The drained areas of both sites were annual CO_2 and CH_4 sources driven largely by soil temperatures (Figure 1, S3 in Supplemental Information). The annual values observed here are within the range reported by Wilson et al. (2015) for a large number of peat extraction sites in Ireland and the UK (that also used data from both the sites in this study), but lower than the derived emission factors for drained peat extraction sites in the temperate zone reported by Drösler et al. (2014). In the latter study, the CO_2 emission factor in particular was strongly affected by data from Canadian peatland sites, where the CO_2 emissions were much higher than extraction sites in Europe (see Wilson et al. 2015, for a more detailed discussion). While the CO_2 emissions reported for our sites are lower than other drained peatland land uses (e.g. cropland, grassland) they are still substantial, especially when compared to mineral soils (Abdalla et al., 2013). Therefore, if these emissions were replicated across the relatively large areas affected by peat extraction in Ireland (Malone and O'Connell, 2009), then these degraded ecosystems represent a substantial hotspot of GHG emissions at the national scale (Wilson et al., 2013b).

Rewetting has had a significant impact on GHG dynamics at both sites in this study. Although, the rewetted area at Blackwater was found to be a net CO_2 source, rewetting still represents a significant saving in terms of avoided emissions (and thereby has a direct climate benefit) in comparison to the drained area (Fig. 8). The 4-year average value reported here for the rewetted area at Blackwater (66 ± 168 g CO_2-C m^{-2} yr^{-1}) is slightly higher than the Tier 1 value (0.50) for nutrient rich temperate peatlands derived by Blain et al. (2014) for the IPCC Wetlands Supplement, although the very wide uncertainty range associated with our value is indicative of a site that fluctuates considerably from sink to source over a short time scale (i.e. 4 years). Nevertheless, it would suggest that the site has moved significantly along a restoration/rehabilitation trajectory. However, GHG emissions/removals from the large body of open water at the site were not quantified in this study and could potentially represent significant hotspots of CO_2 and CH_4 emissions. In the absence of truly comparable studies, values reported by Franz et al. (2016) for a eutrophic rewetted fen would suggest that this could be the case. Notwithstanding, there is potential for further improvements at this site, particularly if *Phragmites* reeds were to colonize the open water area. Our results indicate that *Phragmites* is an effective C sequestration species when water levels remain close to the surface, as has been reported in other post-rewetting peatland studies (Günther et al., 2015;

Minke et al., 2016), but can quickly become a large CO_2 source during drier years (Günther et al., 2015). Annual CH_4 emissions from *Phragmites* in this study were similar to those reported by Huth et al. (2013) for a rewetted fen for a year with high rainfall, but lower than values recorded in other comparable rewetted sites (Minke et al., 2016; van den Berg et al., 2016). However, some studies have shown that the measurement of CH_4 fluxes in *Phragmites* stands with opaque chambers (as in this study) could be underestimated as emissions in these plants are strongly controlled by light levels (Minke et al., 2014), although this effect may only be an issue during mid-summer (Günther et al., 2014). Methane emissions from the sedge microsite were low and comparable to natural peatland ecosystems (e.g. Laine et al., 2007).

Methane emissions at Moyarwood in the first two years post-rewetting were higher, particularly in comparison to natural sites (e.g. Laine et al., 2007; Levy and Gray, 2015). It may be possible that the high initial CH_4 emissions will decline over time as the litter inundated during rewetting activities is decomposed (Limpens et al., 2008); GHG monitoring is ongoing at the site and an extended dataset will provide valuable information. Over the 2-year study period, Moyarwood was a small net annual CO_2 sink (with high uncertainty associated with the estimate) and exhibited values similar to natural sites (e.g. McVeigh et al., 2014). Although rainfall was similar between the two sites for most of the summer months in 2013, the monthly value for July in Moyarwood (101.5 mm), however, was over twice as high as recorded for the Blackwater area (42 mm) and undoubtedly ensured that the former site acted as an annual CO_2 sink, even in that relatively “dry” year. Moreover, the abundance of *Sphagnum* species at Moyarwood may have (a) acted as a buffer to the drought-like conditions (particularly as the rainfall in July would have rehydrated the mosses) and may have helped prevent excessive water drawdown during that period similar to natural bogs (Waddington et al., 2011) and (b) enhanced CO_2 uptake simply because they decompose much more slowly than other species (Clymo, 1965). Nitrous oxide fluxes were not detected at either site, as also reported for some rewetted peatlands (e.g. Günther et al., 2015; Minke et al., 2016; Wilson et al., 2016b), although low N_2O fluxes have been reported for other rewetted studies (see Wilson et al., 2016a).

4.4 Synergies

Successful rewetting of degraded peatlands is a major challenge and, in some cases, may be a balancing act between biodiversity or climate benefits. In this study, the climate benefits from rewetting were slightly higher at Moyarwood where the strong CO_2 sink function offset the substantial CH_4 emissions (Table 4). Nonetheless, both sites have a warming impact on the climate and are clearly some distance away from functioning like natural sites (Fig. 8). Moreover, emissions from the open water at Blackwater (if quantified) could produce a situation where rewetting may actually be more deleterious for the climate than drainage, particularly if CO_2 emissions from drained areas were relatively low compared to other drained peatlands (Drösler et al. 2014). However, in regard to biodiversity provision, Moyarwood was clearly superior to Blackwater in all aspects (Table 4) and would indicate that rewetting of this land use category could provide the desired synergies between climate regulation and biodiversity provision.

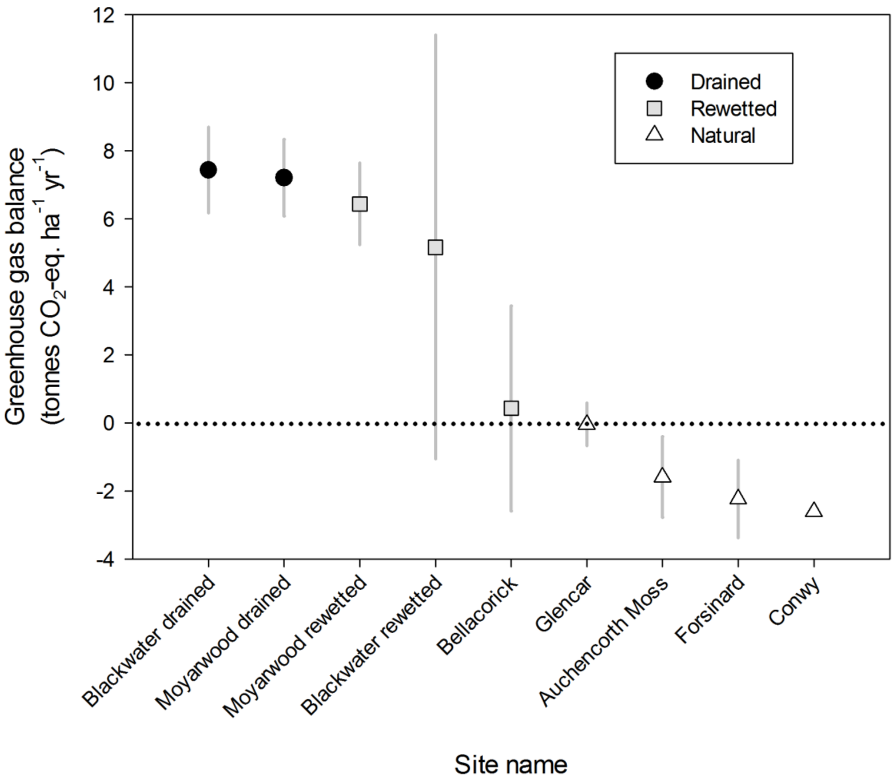

Fig. 8 Greenhouse gas balance (tonnes CO_2-eq. ha^{-1} yr^{-1}) at the drained and rewetted sites at Blackwater and Moyarwood. Dissolved organic carbon (DOC) and CH_4 emissions from drainage ditches were calculated using guidance provided by IPCC (IPCC, 2014). Methane and N_2O fluxes are converted to CO_2-equivalents (t CO_2-eq $ha^{-}1$ yr^{-1}) according to their global warming potential (GWP) on a 100-year timescale: CH_4 = 28 and N_2O = 265 (Myhre et al., 2013). Error bars indicate standard deviation (not available for Conwy). Positive values indicate a net warming impact on the climate and negative values indicate a net cooling impact. Ombrotrophic sites within the British Isles with long-term GHG datasets (>4 years) at Bellacorick, Ireland (Wilson et al., 2016b), Glencar, Ireland (Koehler et al., 2011;

McVeigh et al., 2014), Auchencorth Moss, Scotland (Dinsmore et al., 2010; Dinsmore et al., 2013; Helfter et al., 2015), Forsinard, Scotland (Levy and Gray, 2015) and Conwy, Wales (CEH, unpublished data) are shown for comparative purposes.

5. Conclusions

In Ireland, the current state of peatlands and the consequences of widespread degradation in terms of loss of various ecosystem services have been highlighted (e.g. Renou-Wilson et al., 2011; Wilson et al., 2013b), thereby establishing a framework for the development of the first National Peatlands Strategy (NPWS, 2015). Rewetting as an initial step towards restoration or towards new functioning ecosystems is called upon to meet not only the biodiversity targets but also climate change mitigation targets. However, from an economic point of view it is clearly impossible to rewet all the degraded peatland sites, assuming that they could be rewetted in the first instance (Andersen et al., 2016). Instead, it is more realistic to target specific sites that maximize the synergistic benefits of biodiversity provision and climate change mitigation and, therefore, provide effective cost benefits (Bonn et al., 2014).

The results from this study would indicate that raised bogs, which have been degraded through drainage and domestic extraction on the margins, could be prime candidates for future peatland restoration efforts in Ireland in that they provide suitable synergies between climate change mitigation and biodiversity provision. However, we accept that this prioritizing approach could mean that some industrial extraction sites will remain drained even after extraction has ceased. Nonetheless, given the large areas potentially associated with this land use category (38% of the national peatland resource versus 7% for industrial extraction) (Malone and O'Connell, 2009), actions (both financial and practical) should be directed at suitable sites that can be (a) successfully rewetted and (b) provide the biodiversity/climate benefits outlined above. Site-specific evaluations (micro-habitat and vegetation assessment), regardless of land use category, prior to the initiation of restoration actions, would also be valuable for the decision making process. In this regard, hydrological modelling (Mackin et al., 2017) and remote sensing techniques (e.g. Connolly and Holden, 2017) would also provide critical information to allow for targeted conservation, rewetting or restoration efforts but these approaches must be backed up by appropriate fiscal instruments either through direct funding from government or from carbon/biodiversity offset schemes, such as in operation in other jurisdictions (Tanneberger and Wichtmann, 2011).

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References

Abdalla, M., Saunders, M., Hastings, A., Williams, M., Smith, P., Osborne, B., Lanigan, G., Jones, M.B., 2013. Simulating the impacts of land use in Northwest Europe on Net Ecosystem Exchange (NEE): The role of arable ecosystems, grasslands and forest plantations in climate change mitigation. Science of the Total Environment http://dx.doi.org/10.1016/j.scitotenv.2012.1012.1030.

Alm, J., Shurpali, N.J., Tuittila, E.-S., Laurila, T., Maljanen, M., Saarnio, S., Minkkinen, K., 2007. Methods for determining emission factors for the use of peat and peatlands - flux measurements and modelling. Boreal Environment Research. 12, 85-100.

Andersen, R., Farrell, C., Graf, M., Muller, F., Calvar, E., Frankard, P., Caporn, S., Anderson, P., 2016. An overview of the progress and challenges of peatland restoration in Western Europe. Restoration Ecology. 25, 271-282.

Atherton, I., Bosanquet, S., Lawley, M.,2010. Mosses and Liverworts of Britain and Ireland: a field guide. British Bryological Society. UK. pp. 856.

Barthelmes, A., 2016. The global potential and perspectives for paludiculture. In: Wichtmann W., Schröder C., Joosten H., (Eds.), Paludiculture - productive use of wet peatlands Climate protection − biodiversity − regional economic benefits. Schweizerbart Science Publishers. Stuttgart, Germany. pp. 200-203.

Beetz, S., Liebersbach, H., Glatzel, S., Jurasinski, G., Buczko, U., Höper, H., 2013. Effects of land use intensity on the full greenhouse gas balance in an Atlantic peat bog. Biogeosciences. 10, 1067-1082.

Belyea, L.R., 1996. Separating the effects of litter quality and microenvironment on decomposition rates in a patterned peatland. Oikos. 77, 529-539.

Blain, D., Murdiyarso, D., Couwenberg, J., Nagata, O., Renou-Wilson, F., Sirin, A., Strack, M., Tuittila, E.S., Wilson, D., 2014. Chapter 3. Rewetted organic soils. In: Hiraishi T., Krug T., Tanabe K., Srivastava N., Baasansuren J., Fukuda M., Troxler T.G., (Eds.), 2013 Supplement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories: Wetlands. Intergovernmental Panel on Climate Change. Switzerland. pp.

Bonn, A., Reed, M.S., Evans, C.D., Joosten, H., Bain, C., Farmer, J., Emmer, I., Couwenberg, J., Moxey, A., Artz, R., Tanneberger, F., von Unger, M., Smyth, M.-A., Birnie, D., 2014. Investing in nature: Developing ecosystem service markets for peatland restoration. Ecosystem Services. 9, 54-64, DOI:10.1016/j.ecoser.2014.1006.1011.

Caffrey, J.M., Baars, J.-R., Barbour, J.H., Boets, P., Boon, P., Davenport, K., Dick, J., Early, J., Edsman, L., Gallagher, C., Gross, J., Heinimaa, P., Horrill, C., Hudin, S., Hulme, P.D., Hynes, S., MacIsaac, J., McLoone, P., Millane, M., Moen, T., Moore, N., Newman, J., O'Conhuir, R., O'Farrell, M., O'Flynn, C., Oidtmann, B., Renals, T., Ricciarid, A., Roy, H., Shaw, S.C., van Valkenbur, J., Weiyl, O., William, F., Lucy, F., 2014. Tackling invasive alien species in Europe: the top 20 issues. Management of biological invasions. 5, 1-20.

CBD. 2010. The Strategic Plan for Biodiversity 2011-2020 and the Aichi Biodiversity Targets. Convention on Biological Diversity. Nagoya, Japan, 18-29 October 2010. pp. 13.

Chapman, S., Buttler, A., Francez, A.-J., Laggoun - Défarge, F., Vasander, H., Schloter, M., Combe, J., Grosvernier, P., Harms, H., Epron, D., Gilbert, D., Mitchell, E., 2003. Exploitation of northern peatlands and biodiversity maintenance: a conflict between economy and ecology. Front Ecol Environ. 1, 525-532.

Christensen, T., R, Jackowicz-Korczyński, M., Aurela, M., Crill, P., Heliasz, M., Mastepanov, M., Friborg, T., 2012. Monitoring the multi-year carbon balance of a subarctic palsa mire with micrometeorological techniques. AMBIO. 41, 207-217.

Clymo, R.S., 1965. Experiments on breakdown of Sphagnum in two bogs. The Journal of Ecology. 53, 747-758.

Cobbaert, D., Rochefort, L., Price, J.S., 2004. Experimental restoration of a fen plant community after peat mining. Applied Vegetation Science. 7, 209-220.

Connolly, J., Holden, N.M., 2017. Detecting peatland drains with Object Based Image Analysis and Geoeye-1 imagery. Carbon Balance and Management. 12:7, 1-13.

Dinsmore, K.J., Billet, M.F., Skiba, U.M., Rees, R.M., Drewer, J., Helfter, C., 2010. Role of the aquatic pathway in the carbon and greenhouse gas budgets of a peatland catchment. Global Change Biology. 16, 2750-2762.

Dinsmore, K.J., Billett, M.F., Dyson, K.E., 2013. Temperature and precipitation drive temporal variability in aquatic carbon and GHG concentrations and fluxes in a peatland catchment. Global Change Biology. 19, 2133-2148.

Dinsmore, K.J., Skiba, U.M., Billett, M.F., Rees, R.M., Drewer, J., 2009. Spatial and temporal variability in CH_4 and N_2O fluxes from a Scottish ombrotrophic peatland: Implications for modelling and up-scaling. Soil Biology and Biochemistry. 41, 1315-1323.

Doyle, G., O' Críodáin, C., 2003. Peatlands - fens and bogs. In: Otte M., (Eds.), Wetlands of Ireland. University College Dublin Press. Dublin, Ireland. pp. pp. 256.

Drösler, M., Verchot, L., Freibauer, A., Pan, G., Evans, C.D., Bourbonniere, R.A., Alm, J.P., Page, S., Agus, F., Hergoualc'h, K., Couwenberg, J., Jauhiainen, J., Sabiham, S., Wang, C., Srivastava, N., Bourgeau-Chavez, L.L., Hooijer, A., Minkkinen, K., French, N., Strand, T., Sirin, A., Mickler, R., Tansey, K., Larkin, N., 2014. Chapter 2: Drained Inland Organic Soils. In: Hiraishi T., Krug T., Tanabe K., Srivastava N., Baasansuren J., Fukuda M., Troxler T.G., (Eds.), 2013 Supplement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories: Wetlands. Intergovernmental Panel on Climate Change. Switzerland. pp.

Duffy, P., Hanley, E., Black, K., O'Brien, P., Hyde, B., Ponzi, J., Alam, S., 2015. National Inventory Report 2015. Greenhouse gas emissions 1990-2013 reported to the United Nations Framework Convention on Climate Change. pp. 581.

Ellenberg, H., Leuschner, C., 2010. Chapter 27: Zeigerwerte der Pflanzen Mitteleuropas. Vegetation Mitteleuropas mit den Alpen. UTB. Stuttgart, Germany. pp. 109.

Evans, C.D., Page, S.E., Jones, T., Moore, S., Gauci, V., Laiho, R., Hruška, J., Allott, T.E.H., Billett, M.F., Tipping, E., Freeman, C., Garnett, M.H., 2014. Contrasting vulnerability of drained tropical and high-latitude peatlands to fluvial loss of stored carbon. Global Biogeochemical Cycles. 28, 2013GB004782.

Fallon, D., McCorry, M.J., Farrell, C., Moran, J., 2012. Ten years in rehab. What have we learnt from Mayo? In: Magnusson T., (Eds.), 14th International Peat Congress. Stockholm, Sweden. Extended abstract 201. 185.

Farrell, C.A., 2008. The biodiversity value and future management of degraded peatland habitats in Ireland. In: Feehan J., Farrell C.A., (Eds.), 13th International Peat Congress. Tullamore, Ireland. 686-689.

Fernandez, F., Connolly, K., Crowley, W., Denyer, J., Duff, K., Smith, G.D., 2014. Raised bog monitoring and assessment survey 2013. Irish Wildlife Manuals No 81: National Parks and Wildlife Service, Departments of Arts, Heritage and Gaeltacht, Dublin, Ireland. pp. 198.

Fossitt, J.A.,2000. A guide to habitats in Ireland. The Heritage Council. Dublin. pp. 120.

Franz, D., Koebsch, F., Larmanou, E., Augustin, J., Sachs, T., 2016. High net CO_2 and CH_4 release at a eutrophic shallow lake on a formerly drained fen. Biogeosciences. 13, 3051-3070.

Frolking, S., Roulet, N.T., Fuglestvedt, J., 2006. How northern peatlands influence the Earth's radiative budget: sustained methane emission versus sustained carbon sequestration. Journal of Geophysical Research. 111, G01008, doi:01010.01029/02005JG000091.

Glenk, K., Schaafsma, M., Moxey, A., Martin-Ortega, J., Hanley, N., 2014. A framework for valuing spatially targeted peatland restoration. Ecosystem Services. 9, 20-33.

Goodrich, J.P., Campbell, D.I., Roulet, N.T., Clearwater, M.J., Schipper, L.A., 2015. Overriding control of methane flux temporal variability by water table dynamics in a Southern Hemisphere, raised bog. Journal of Geophysical Research: Biogeosciences. 120, 819-831.

Green, S.M., Baird, A.J., 2017. Using ‘snapshot’ measurements of CH_4 fluxes from an ombrotrophic peatland to estimate annual budgets: interpolation versus modelling. Mires and Peat. 19, Article 09, 01–09, DOI: 10.19189/MaP.12016.OMB.19254.

Günther, A., Huth, V., Jurasinski, G., Glatzel, S., 2015. The effect of biomass harvesting on greenhouse gas emissions from a rewetted temperate fen. Global Change Biology: Bioenergy. 7, 1092-1106.

Günther, A.B., Huth, V., Jurasinski, G., Glatzel, S., 2014. Scale-dependent temporal variation in determining the methane balance of a temperate fen. Greenhouse Gas Measurement and Management DOI: 10.1080/20430779.20432013.20850395.

Hannigan, E., Mangan, R., Kelly-Quinn, M., 2011. Evaluation of the success of mountain blanket bog pool restoration in terms of aquatic macro invertebrates. Biology and Environment: Proceedings of the Royal Irish Academy. 111B, DOI: 10.3318/BIOE.2011.3111.3308.

Helfter, C., Campbell, C., Dinsmore, K.J., Drewer, J., Coyle, M., Anderson, M., Skiba, U., Nemitz, E., Billett, M.F., Sutton, M.A., 2015. Drivers of long-term variability in CO_2 net ecosystem exchange in a temperate peatland. Biogeosciences. 12, 1799-1811.

Hill, M.O., Mountford, J.O., Roy, D.B., Bunce, R., 1999. Ellenberg's indicator values for British plants: ECOFACT Research reoport series Volume 2, Technical Annex. Centre for Ecology and Hydrology. Abbots Ripton. pp.

Hill, M.O., Preston, C.D., Bosanquet, S., Roy, D.B., 2007. BRYOATT: Attributes of British and Irish mosses, liverworts and hornworts. Centre for Ecology and Hydrology. Huntingdon, Cambridgeshire, England. pp. 88.

Hogg, E.H., 1993. Decay potential of hummock and hollow sphagnum peats at different depths in a Swedish raised bog. Oikos. 66, 269-278.

Houghton, R.A., House, J.I., Pongratz, J., van der Werf, G.R., DeFries, R.S., Hansen, M.C., Le Quéré, C., Ramankutty, N., 2012. Carbon emissions from land use and land-cover change. Biogeosciences. 9, 5125-5142.

Huth, V., Günther, A., Jurasinski, G., Glatzel, S., 2013. The effect of an exceptionally wet summer on methane effluxes from a 15-year re-wetted fen in north-east Germany. Mires and Peat. 13, Article 02, 01–07. http://www.mires-and-peat.net/, ISSN 1819-1754X.

IPCC. 2006. IPCC Guidelines for National Greenhouse Gas Inventories. Prepared by the National Greenhouse Gas Inventories programme. Volume 4. Eggleston H.S., Buendia L., Miwa K., Ngara T., Tanabe K.: IGES. Japan. pp. 169.

IPCC.2014. 2013 Supplement to the 2006 Inter-Governmental Panel on Climate Change Guidelines for National Greenhouse Gas Inventories: Wetlands. IPCC, Switzerland. pp. 354.

Jeglum, J.K., 1971. Plant indicators of pH and water level in peatlands at Candle Lake, Saskatchewan. Canadian Journal of Botany. 49, 1661-1676.

Joosten, H., Tapio-Biström, M.-L., Tol, S., 2012. Peatlands - guidance for climate change mitigation through conservation, rehabilitation and sustainable use. Mitigation of Climate Change in Agriculture (MICCA) Programme. pp. 100.

Kent, M., Coker, P.,1992. Vegetation description and analysis: a practical approach. John Wiley & Sons Ltd. Chichester,England. pp. 363.

Kettridge, N., Binley, A., Green, S.M., Baird, A.J., 2011. Ebullition events monitored from northern peatlands using electrical imaging. Journal of Geophysical Research: Biogeosciences. 116, G04004, doi:04010.01029/02010JG001561.

Knox, S.H., Sturtevant, C., Matthes, J.H., Koteen, L., Verfaillie, J., Baldocchi, D., 2015. Agricultural peatland restoration: effects of land-use change on greenhouse gas (CO_2 and CH_4) fluxes in the Sacramento-San Joaquin Delta. Global Change Biology. 21, 750-765.

Koehler, A.-K., Sottocornola, M., Kiely, G., 2011. How strong is the current carbon sequestration of an Atlantic blanket bog? Global Change Biology. 17, 309-319.

Lachance, D., Lavoie, C., 2004. Vegetation of Sphagnum bogs in highly disturbed landscapes: relative influence of abiotic and anthropogenic factors. Applied Vegetation Science. 7, 183-192.

Laine, A., Wilson, D., Kiely, G., Byrne, K.A., 2007. Methane flux dynamics in an Irish lowland blanket bog. Plant and Soil. 299, 181-193.

Le Quéré, C., Andrew, R.M., Canadell, J.G., Sitch, S., Korsbakken, J.I., Peters, G.P., Manning, A.C., Boden, T.A., Tans, P.P., Houghton, R.A., Keeling, R.F., Alin, S., Andrews, O.D., Anthoni, P., Barbero, L., Bopp, L., Chevallier, F., Chini, L.P., Ciais, P., Currie, K., Delire, C., Doney, S.C., Friedlingstein, P., Gkritzalis, T., Harris, I., Hauck, J., Haverd, V., Hoppema, M., Klein Goldewijk, K., Jain, A.K., Kato, E., Körtzinger, A., Landschützer, P., Lefèvre, N., Lenton, A., Lienert, S., Lombardozzi, D., Melton, J.R., Metzl, N., Millero, F., Monteiro, P.M.S., Munro, D.R., Nabel, J.E.M.S., Nakaoka, S.I., O'Brien, K., Olsen, A., Omar, A.M., Ono, T., Pierrot, D., Poulter, B., Rödenbeck, C., Salisbury, J., Schuster, U., Schwinger, J., Séférian, R., Skjelvan, I., Stocker, B.D., Sutton, A.J., Takahashi, T., Tian, H., Tilbrook, B., van der Laan-Luijkx, I.T., van der Werf, G.R., Viovy, N., Walker, A.P., Wiltshire, A.J., Zaehle, S., 2016. Global Carbon Budget 2016. Earth Syst Sci Data. 8, 605-649.

Levy, P.E., Gray, A., 2015. Greenhouse gas balance of a semi-natural peatbog in northern Scotland. Environmental Research Letters. 10, 094019.

Limpens, J., Berendse, F., 2003. How litter quality affects mass loss and N loss from decomposing Sphagnum. Oikos. 103, 537-547.

Limpens, J., Berendse, F., Blodau, C., Canadell, J.G., Freeman, C., Holden, J., Roulet, N., Rydin, H., Schaepman-Strub, G., 2008. Peatlands and the carbon cycle: from local processes to global implications – a synthesis. Biogeosciences. 5, 1475-1491.

Lloyd, J., Taylor, J.A., 1994. On the temperature dependence of soil respiration. Functional Ecology. 8, 315-323.

Mackin, F., Flynn, R., Barr, A., Fernandez-Valverde, F., 2017. Use of geographical information system-based hydrological modelling for development of a raised bog conservation and restoration programme. Ecological Engineering. 106, Part A, 242-252.

Malone, S., O'Connell, C., 2009. Irelands Peatland Conservation Action Plan 2020 - Halting the loss of biodiversity. Irish Peatland Conservation Council. pp. 152.

Martin-Ortega, J., Glenk, K., Byg, A., 2017. How to make complexity look simple? Conveying ecosystems restoration complexity for socio-economic research and public engagement. PLoS ONE. 12, e0181686.

McDonald, T., Gann, G.D., Jonson, J., Dixon, K.W., 2016. International standards for the practice of ecological restoration - including principles and key concepts. Society for Ecological Restoration (SER). Washington D.C., USA. pp. 47.

McVeigh, P., Sottocornola, M., Foley, N., Leahy, P., Kiely, G., 2014. Meteorological and functional response partitioning to explain interannual variability of CO_2 exchange at an Irish Atlantic blanket bog. Agricultural and Forest Meteorology. 194, 8-19.

Minayeva, T., Bragg, O.M., Sirin, A.A., 2017. Towards ecosystem-based restoration of peatland biodiversity. Mires and Peat. 19, Article 01, 01-36, DOI: 10.19189/MaP.12013.OMB.19150.

Minke, M., Augustin, J., Burlo, A., Yarmashuk, T., Chuvashova, H., Thiele, A., Freibauer, A., Tikhonov, V., Hoffmann, M., 2016. Water level, vegetation composition, and plant productivity explain greenhouse gas fluxes in temperate cutover fens after inundation. Biogeosciences. 13, 3945-3970.

Minke, M., Augustin, J., Hagemann, U., Joosten, H., 2014. Similar methane fluxes measured by transparent and opaque chambers point at belowground connectivity of Phragmites australis beyond the chamber footprint. Aquatic Botany. 113, 63-71.

Myhre, G., Shindell, D., Bréon, F.-M., Collins, W., Fuglestvedt, J., Huang, J., Koch, D., Lamarque, J.-F., Lee, L., Mendoza, B., Nakajima, T., Robock, A., Stephens, G., Takemura, T., Zhang, H., 2013. Anthropogenic and Natural Radiative Forcing. In: Stocker T.F., Qin D., Plattner G.-K., Tignor M., Allen S.K., Boschung J., Nauels A., Xia Y., Bex V., Midgley P.M., (Eds.), Climate Change 2013: The Physical Science Basis Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press. Cambridge, United Kingdom and New York, NY, USA. pp. 659-740.

NPWS. 2015. National Peatlands Strategy. National Parks & Wildlife Service, Department of Arts, Heritage and the Gaeltacht. Dublin, Ireland. pp. 84.

NPWS. In press. National Raised Bog SAC Management Plan Draft. National Parks and Wildlife Service, Department of Arts, Heritage and the Gaeltacht. Dublin. pp. 122.

Page, S.E., Baird, A.J., 2016. Peatlands and Global Change: Response and Resilience. Annual Review of Environment and Resources. 41, 35-57.

Parish, F., Sirin, A., Charman, D., Joosten, H., Minayeva, T., Silvius, M., Stringer, I., 2008. Assessment on peatlands, biodiversity and climate change. Main report. Global Environment Centre, Kuala Lumpur and Wetlands International. Wageningen, Netherlands. pp. 179.

Parnell, J., Curtis, T.,2012. Webb's An Irish Flora (8th edition). Cork University Press. Cork, Ireland. pp. 560.

Pasquet, S., Pellerin, S., Poulin, M., 2015. Three decades of vegetation changes in peatlands isolated in an agricultural landscape. Applied Vegetation Science. 18, 220-229.

Renou-Wilson, F., Barry, C., Müller, C., Wilson, D., 2014. The impacts of drainage, nutrient status and management practice on the full carbon balance of grasslands on organic soils in a maritime temperate zone. Biogeosciences. 11, 4361-4379.

Renou-Wilson, F., Bolger, T., Bullock, C., Convery, F., Curry, J.P., Ward, S., Wilson, D., Müller, C., 2011. BOGLAND - Sustainable Management of Peatlands in Ireland. STRIVE Report No 75 prepared for the Environmental Protection Agency (EPA). Johnstown Castle, Co. Wexford, Ireland. pp. 157.

Renou-Wilson, F., Müller, C., Moser, G., Wilson, D., 2016. To graze or not to graze? Four years GHG balances and vegetation composition from a drained and a rewetted organic soil under grassland. Agriculture, Ecosystem and the Environment. 222, 156-170.

Renou-Wilson, F., Wilson, D., Rigney, C., Byrne, K., Farrell, C., Müller, C.,2018. Network Monitoring Rewetted and Restored Peatlands/Organic Soils for Climate and Biodiversity Benefits (NEROS project Synthesis Report). EPA Research Report No 236. Environmental Protection Agency. Wexford, Ireland. pp. 49.

Robroek, B.J.M., Limpens, J., Breeuwer, A., Schouten, M.G.C., 2007. Effects of water level and temperature on performance of four *Sphagnum* mosses. Plant Ecology. 190, 97-107.

Rochefort, L., 2000. Sphagnum: A Keystone Genus in Habitat Restoration. The Bryologist. 103, 503-508.

Rochefort, L., LeBlanc, M.-C., Bérubé, V., Hugron, S., Boudreau, S., Pouliot, R., 2016. Reintroduction of fen plant communities on a degraded minerotrophic peatland. Botany 1041-1051.

Roulet, N.T., Lafleur, P.M., Richard, P.J.H., Moore, T., Humphreys, E.R., Bubier, J., 2007. Contemporary carbon balance and late Holocene carbon accumulation in a northern peatland. Global Change Biology. 13, 397-411, doi:310.1111/j.1365-2486.2006.01292.

Schouten, M.G.C.,2002. Conservation and restoration of raised bogs: geological, hydrological and ecological studies. Department of the Environmental and Local Government. Staatsboshbeheer, The Netherlands. pp. 220.

Smith, G.F., O'Donoghue, P., O'Hora, K., Delaney, E., 2011. Best practice guidance for habitat survey and mapping. The Heritage Council, Ireland. pp. 132.

Smith, P., Bustamante, M., Ahammad, H., Clark, H., Dong, H., Elsiddig, E.A., Haberl, H., Harper, R., House, J., Jafari, M., Masera, O., Mbow, C., Ravindranath, N.H., Rice, C.W., Robledo Abad, C., Romanovskaya, A., Sperling, F., Tubiello, F., 2014. Agriculture, Forestry and Other Land Use (AFOLU). In: Edenhofer O., Pichs-Madruga R., Sokona Y., Farahani E., Kadner S., Seyboth K., Adler A., Baum I., Brunner S., Eickemeier P., Kriemann B., Savolainen J., Schlömer S., von Stechow C., Zwickel T., Minx J.C., (Eds.), Climate Change 2014: Mitigation of Climate Change Contribution of Working Group III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press. Cambridge, United Kingdom and New York, NY, USA. pp. 811-922.

Strack, M., Zuback, Y.C.A., 2013. Annual carbon balance of a peatland 10 yr following restoration. Biogeosciences. 10, 2885-2896.

Tanneberger, F., Wichtmann, W.,2011. Carbon credits from peatland rewetting: Climate - biodiversity - land use. Schweizerbart Science Publishers. Stuttgart, Germany. pp. 223.

Tiemeyer, B., Albiac Borraz, E., Augustin, J., Bechtold, M., Beetz, S., Beyer, C., Drösler, M., Ebli, M., Eickenscheidt, T., Fiedler, S., Förster, C., Freibauer, A., Giebels, M., Glatzel, S., Heinichen, J., Hoffmann, M., Höper, H., Jurasinski, G., Leiber-Sauheitl, K., Peichl-Brak, M., Roßkopf, N., Sommer, M., Zeitz, J., 2016. High emissions of greenhouse gases from grasslands on peat and other organic soils. Global Change Biology 10.1111/gcb.13303.

Tuittila, E.-S., Komulainen, V.-M., Vasander, H., Laine, J., 1999. Restored cut-away peatland as a sink for atmospheric CO_2. Oecologia. 120, 563 - 574.

Urbanová, Z., Picek, T., Bárta, J., 2011. Effect of peat re-wetting on carbon and nutrient fluxes, greenhouse gas production and diversity of methanogenic archaeal community. Ecological Engineering. 37, 1017-1026.

van den Berg, M., Ingwersen, J., Lamers, M., Streck, T., 2016. The role of Phragmites in the CH_4 and CO_2 fluxes in a minerotrophic peatland in southwest Germany. Biogeosciences. 13, 6107-6119.

Van der Schaaf, S., Streefkerk, J.G., Daly, D., Johnson, P., 2002. Regional hydrology. In: Schouten M.G.C., (Eds.), Conservation and Restoration of Raised Bogs: Geological, Hydrological and Ecological Studies. Department of the Environment and Local Government / Staatsbosbeheer. pp. 32-54.

Vanselow-Algan, M., Schmidt, S.R., Greven, M., Fiencke, C., Kutzbach, L., Pfeiffer, E.M., 2015. High methane emissions dominated annual greenhouse gas balances 30 years after bog rewetting. Biogeosciences. 12, 4361-4371.

Vicherová, E., Hájek, M., Šmilauer, P., Hájek, T., 2017. Sphagnum establishment in alkaline fens: Importance of weather and water chemistry. Science of The Total Environment. 580, 1429-1438.

Waddington, J.M., Lucchese, M.C., Duval, T.P., 2011. Sphagnum moss moisture retention following the re-vegetation of degraded peatlands. Ecohydrology. 4, 359-366.

Waddington, J.M., Tóth, K., Bourbonniere, R.A., 2008. Dissolved organic carbon export fom a cutover and restored peatland. Hydrological Processes. 22, 2215-2224.

Whelan, P.,2011. Lichens of Ireland: An Illustrated Introduction. Collins Press. Cork, Ireland. pp. 160.

Wilson, D., Alm, J., Riutta, T., Laine, J., Byrne, K.A., Farrell, E.P., Tuittila, E.-S., 2007a. A high resolution green area index for modelling the seasonal dynamics of CO_2 exchange in vascular plant peatland communities. Plant Ecology. 190, 37-51.

Wilson, D., Blain, D., Couwenberg, J., Evans, C.D., Murdiyarso, D., Page, S., Renou-Wilson, F., Rieley, J., Sirin, A., Strack, M., Tuittila, E.-S., 2016a. Greenhouse gas emission factors associated with rewetting of organic soils. Mires and Peat. 17, Article 04, 01–28.

Wilson, D., Dixon, S.D., Artz, R.R.E., Smith, T.E.L., Evans, C.D., Owen, H.J.F., Archer, E., Renou-Wilson, F., 2015. Derivation of greenhouse gas emission factors for peatlands managed for extraction in the Republic of Ireland and the United Kingdom. Biogeosciences. 12, 5291-5308.

Wilson, D., Farrell, C., A., Müller, C., Hepp, S., Renou-Wilson, F., 2013a. Rewetted industrial cutaway peatlands in western Ireland: prime location for climate change mitigation? Mires and Peat. 11, Article 01, 01-22

Wilson, D., Farrell, C., Fallon, D., Moser, G., Muller, C., Renou-Wilson, F., 2016b. Multi-year greenhouse gas balances at a rewetted temperate peatland. Global Change Biology. 22, 4080-4095, DOI: 4010.1111/gcb.13325.

Wilson, D., Müller, C., Renou-Wilson, F., 2013b. Carbon emissions and removals from Irish peatlands: current trends and future mitigation measures. Irish Geography. 46, 1-23.

Wilson, D., Tuittila, E.-S., Alm, J., Laine, J., Farrell, E.P., Byrne, K.A., 2007b. Carbon dioxide dynamics of a restored maritime peatland. Ecoscience. 14, 71-80.

Ye, R., Horwath, W.R., 2016. Nitrous oxide uptake in rewetted wetlands with contrasting soil organic carbon contents. Soil Biology and Biochemistry. 100, 110-117.

Yli-Petäys, M., Laine, J., Vasander, H., Tuittila, E.-S., 2007. Carbon gas exchange of a re-vegetated cut-away peatland five decades after abandonment. Boreal Environment Research. 12, 177-190.

Supplementary Material

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