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To graze or not to graze? Four years greenhouse gas balances and vegetation composition from a drained and a rewetted organic soil under grassland

F. Renou-Wilson[a,*], C. Müller[a,b], G. Moser[b], D. Wilson[c]

[a] School of Biology and Environmental Science, Science Centre West, University College Dublin, Belfield, Dublin 4, Ireland
[b] Institute for Plant Ecology, Justus Liebig University Giessen, Germany
[c] Earthy Matters Environmental Consultants, Glenvar, Letterkenny, Co. Donegal, Ireland

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ABSTRACT

Nutrient-poor organic soils under maritime grassland are often located in remote wet locations in the landscape. Leaving these soils without drainage maintenance often raise the water table but continuous management (grazing) means they could remain a source of carbon dioxide (CO_2) while also turning into a small source of methane (CH_4). Due to geographical and socio-economic reasons, removing these sites from agricultural production may be an option to mitigate greenhouse gas (GHG) emissions. To test this hypothesis we measured GHG fluxes over a four year period, at a drained and a rewetted organic soil under grassland, which were both grazed for the first two years and not grazed for the following two years. Statistical response functions estimated for gross primary production (GPP) and ecosystem respiration (R_{eco}) were used to reconstruct annual CO_2–C balances using site-specific models driven by soil temperature, solar radiation, soil water table (WT) and leaf area index (LAI). Annually, soil CO_2 emissions were comparable when grazed, although the rewetted site had a lower net ecosystem carbon balance (NECB) despite displaying higher CH_4 emissions. Both sites have lower CO_2 emissions than typical drained organic soils due to management practices: extensive grazing, no fertilisation and mean annual water tables above −25 cm. When grazing stopped, GPP and R_{eco} increased dramatically driven by vigorous growth of vegetation at both sites. The shallow drained site remained a source of CO_2 and small source of CH_4 while the rewetted site became either neutral or a small sink of CO_2 with decreased CH_4 emissions compared to the grazing period. Nitrous oxide (N_2O) emissions were negligible at either site. Removing grazing significantly reduced the NECB at both sites but in terms of global warming potential (GWP), the greatest GHG mitigation was in the rewetted site which exerted a cooling effect in the second year after the management shift.

1. Introduction

European soil carbon (C) stocks are estimated at 75 Gt of C with 20% located in organic soils in the northern part of Europe (Jones et al., 2004). This soil C pool is ten times the size of the next main terrestrial C pool that is European forests (Liski et al., 2003). The organic soil C pool is mainly affected by anthropogenic action rather than climate change (Smith et al., 2007) and it is of great significance in the context of climate change mitigation because the most effective option to manage soil C is to preserve existing stocks in soils (Schils et al., 2008; Powlson et al., 2011). Thus, the use of organic soils for agriculture is a contentious land use option in terms of atmospheric impacts (Joosten et al., 2012; Berninger et al., 2015; Regina et al., 2015). Land use change and related drainage of organic soils has influenced the global balance of the three main greenhouse gases (GHG)—carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O), and has resulted in the largest emissions of CO_2 from soils in Europe (Schils et al., 2008).

The drainage of organic soils for agriculture leads to an aerobic upper layer of the soil that (1) increases the decomposition of the soil organic matter, which releases the stored C as CO_2 to the atmosphere and, concomitantly (2) decreases CH_4 emissions through oxidation (albeit emissions may remain from the ditches) and (3) increases the mineralisation of organic nitrogen potentially causing elevated N_2O emissions to the atmosphere, a by-product of both nitrification and denitrification and a range of other processes (Butterbach-Bahl et al., 2013). Quantifying the spatio-temporal

* Corresponding author.
E-mail address: Florence.Renou@ucd.ie (F. Renou-Wilson).

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patterns of GHG emissions from organic soils that vary in terms of physico-chemistry, land use, management intensity and practice has become a critical issue in order to assess their true climate footprint (Elsgaard et al., 2012; Renou-Wilson et al., 2014; Eickenscheidt et al., 2015; Petrescu et al., 2015). However, research has mainly focused on how to reduce emissions from cultivated organic soils (Maljanen et al., 2010; Regina and Alakukku, 2010). Few studies of grasslands over peat have compared GHG fluxes from distinct sites under varying management activities (Beetz et al., 2013; Görres et al., 2014; Schrier-Uijl et al., 2014) including the cessation of all agricultural activities to create nature reserves (Hendriks et al., 2007; Maljanen et al., 2013; Pellegrino et al., 2015).

In Ireland, agriculture is the second largest producer of GHGs and grassland is the dominant land use category (60%). An estimated 300,000 ha of grassland is over organic soils while a mere 1235 ha of organic soils are cultivated (Donlan and Byrne, 2015). Nutrient rich, deep drained organic soils under grasslands have been identified as hotspots for both gaseous and fluvial C losses (Renou-Wilson et al., 2014). However, low intensity, nutrient poor organic soils under grassland are likely to emit much less CO_2 and even perhaps sequester C if the annual water table (WT) can be kept higher than −30 cm below the surface (Renou-Wilson et al., 2014; Barry et al., 2016). There is still a lack of quantitative data in relation to the influence of edaphic properties and management regime on CO_2 emissions from permanent grasslands, due to a lack of long-term studies. Consequently estimating the potential, costs and feasibility of mitigation measures associated with such organic soils has not been fully determined in national policy agenda (Regina et al., 2015).

Increasing the WT levels either by controlled drainage or full rewetting (successful rewetting is defined as maintaining the mean annual WT levels just below the surface) can play an important role in climate change mitigation by effectively reducing CO_2 and N_2O emissions. However, increased anaerobic conditions in previously drained organic soils are likely to lead to increased CH_4 emissions following rewetting (Hendriks et al., 2007; Beetz et al., 2013; Wilson et al., 2013), which are highly significant in the overall GHG balance. As with drained organic soils, the observed variability of annual GHG emissions reported from rewetted organic soils demonstrates that fluxes are dependent not only on peat physico-chemistry and WT dynamics (Koebsch et al., 2013; Lamers et al., 2015) but also on the intensity of management practices (e.g. grazing) that impact on vegetation and readily affect both CO_2 (Beetz et al., 2013) and CH_4 dynamics (Vanselow-Algan et al., 2015; Zak et al., 2015). Therefore the mitigation potential may be limited depending on which site is rewetted and which management practices are employed. Permanent grasslands on organic soils require intensive continuous drainage which, if not maintained, will allow the land to rewet. Taking such rewetted land out of agricultural production (e.g. fencing off cattle) could be an option where the land is considered to be of marginal economic viability, either due to rural de-population, ageing farmers, changing labour and input costs or because sites are remote, commonage land or particularly wet (Strijker, 2005). The impact of

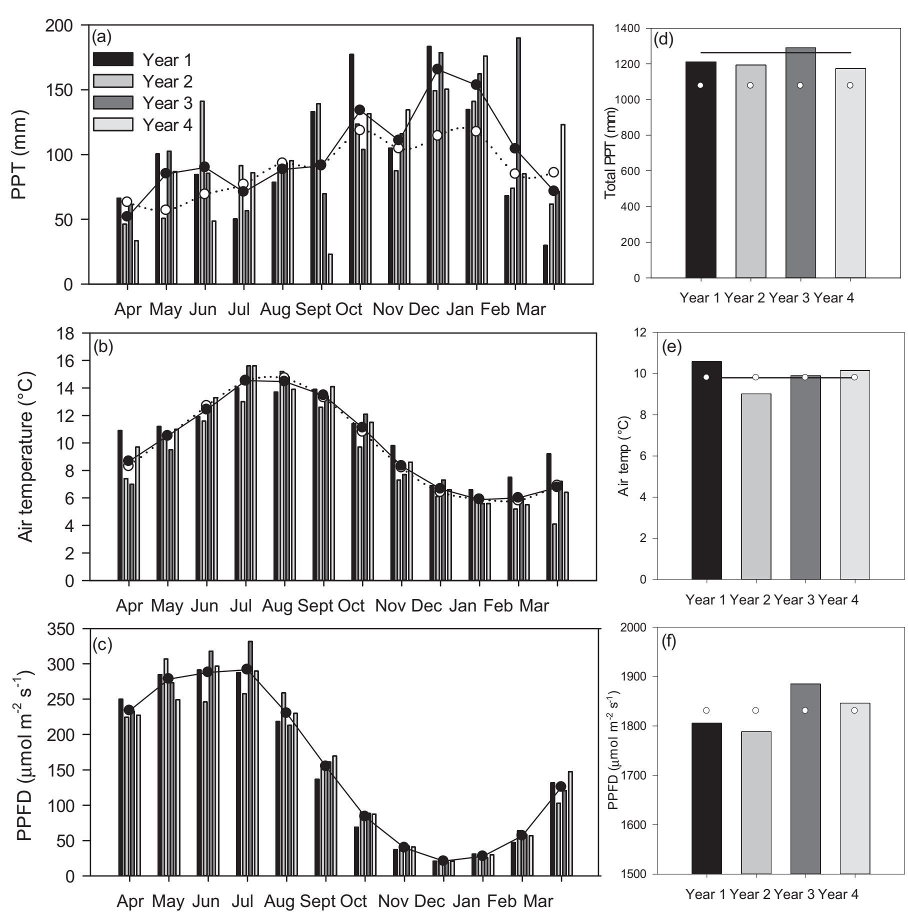

Fig. 1. Climatic data from Glenvar for each of the 4 years monitored in this study (Year 1 starting April 2011): (a) monthly averages or annual means of precipitation (PPT, mm), (b) air temperature (°C), (c) Photosynthetic Photon Flux Density (PPFD, μmol m^{-2} s^{-1}) and annual means of (d) total precipitation (mm), (e) air temperature (°C), (f) PPFD (μmol m^{-2} s^{-1}). White circles indicate 30-year average (1982–2010; www.met.ie; monthly long-term means not available for PPFD); dark circles and solid line indicate current 4-year average.w

a management shift, such as the complete removal of grazing pressures has not been tested for rewetted grasslands over organic soils.

Therefore, the objective of this study was to quantify the GHG emissions in adjacent shallow drained (SD) and rewetted (RW) nutrient poor organic soils under grassland with a view to analyse the role and relative importance of the WT levels, vegetation, weather conditions as well as the shift in management from 'grazed' to 'ungrazed'. To this end, both drained and rewetted fields were monitored for four years: the first two years under normal grazing regimes and the following two years under no grazing. We hypothesise that (1) the RW site has reduced CO_2 emissions but increased CH_4 emissions compared to the SD site and that (2) shifting to no grazing management decreases the global warming potential (GWP) in both cases; (3) GHG balances display a high degree of inter-annual variability being influenced by the interactive effects of both weather conditions and management actions. Finally we propose a worse and best case scenario of water table/vegetation management combination with a view to the development of a green low-carbon agri-environmental option for these marginal lands.

2. Materials and methods

2.1. Study sites and agricultural practices

The study area is located in the north-west of Ireland in Glenvar, Co. Donegal, c. 1 km from the coast (Latitude: 55°9′N, Longitude: 7°34′W; 40 m above sea level). The climate is humid (Atlantic) temperate and the 30-year mean annual air temperature is 9.8 °C (with mild winters and cool summers) and total annual precipitation is 1076 mm (Met Éireann, 1981–2010 average at Malin Head Meteorological Station located 30 km from the site, Fig. 1). The present study site was a permanent grassland managed for extensive beef suckler production (0.6 livestock units per ha), receiving only on-site organic fertilisation (i.e. manure excreted by animals only). Cattle are present only during the growing season, which can extend from mid-March to mid-October. While the site has been drained for over 60 years, the drainage network fell into disrepair due to age-related decline in farm activities and blocked drains have featured in parts of the farm for the last 15 years. This allowed an investigation of two distinct areas: a rewetted site (Site RW) where water saturation conditions were (unintentionally) re-established on previously drained soil, and a shallow drained site (Site SD) with the mean annual WT deeper than −30 cm below the soil surface (*cf.* IPCC, 2014). The soil is typical earthy peat, categorised as a terric histosol overlying bedrock of Precambrian schist and gneiss. The two sites are nutrient poor and demonstrate small differences in the physico-chemical characteristics (see Table 1 and additional information in Renou-Wilson et al. (2014)). Eleven permanent sample collars (60 cm × 60 cm) were established systematically along a transect with a minimum distance of 0.5 m between each plot (n = 5 for Site SD, n = 6 for Site RW) and were fenced off from cattle for the duration of the study.

Table 1
Soil and land use characteristics of the research sites.

	Shallow drained (SD)	Rewetted (RW)
pH	5.8	5.7
OM% (LOI)	36	39
C%	21	23
N%	0.78	0.86
C:N	26.9	26.7
P (mg/L)	0.41	0.44

C = total organic carbon; N = total nitrogen; P = available phosphorus; OM = Organic Matter; LOI (loss-on-ignition).

2.2. Vegetation management and measurements

Both sites displayed a low quality grass sward dominated by *Holcus lanatus* and *Agrostis* spp. with a varying proportion of wetland species (*Juncus* spp., *Equisetum* spp.). A vegetation survey was conducted at the beginning and the end of the 4-year study period, by dividing each collar (sample plot) into quadrants (30 × 30 cm) and estimating species cover within each quadrant (to the closest 1%). Individual species cover estimates were averaged for the entire plot and computed to estimate coverage of mosses, dryland and wetland species as well aerenchymous species.

For the first two years of the study, the study sites were managed so as to simulate local management activities and therefore the vegetation within and surrounding each collar was cut to a stubble height of 5 cm in order to mimic the cattle grazing regime. The vegetation was collected from each collar and oven dried to a constant mass at 75 °C. C and N contents were measured using a homogenised sample for each plot (CE440 Exeter Elemental Analyser). After two years, cutting of the vegetation ceased to simulate the removal of cattle grazing. Vegetation height (cm) was measured regularly and systematically throughout the study during GHG measurements and before 'cutting' events. During the first two years, leaf area index (LAI) was measured at each plot when photosynthetic photon flux density (PPFD) levels were sufficient (November–March excluded). The AccuPAR LP-80 (Ceptometer, Decagon Instruments, WA, USA) was used to measure light interception via 80 independent photo sensors. The PPFD transmission data was used to calculate gap fractions which were inverted to derive LAI estimates (Norman and Campbell, 1989). Polynomial regression constrained to pass through the origin was used to develop annual site-specific relationships between vegetation height and LAI (with a second-order polynomial term for vegetation height) for use in flux modelling (see below).

During the third and fourth year, the vegetation composition changed and the multi-layered canopy architecture meant that the growth dynamic of the predominant plant species could not be captured using the LAI method described above. Therefore the Green Area Index method as described by Wilson et al. (2007a) was used, which involved measuring the green photosynthetic area of all vascular plants within the sample plot using marked plants whose size and number of leaves were determined at monthly intervals. Moss cover was estimated twice during each year. Species-specific model curves were applied to describe the phenological dynamics of the vegetation of each plot to model the annual development of green area that was summed to produce LAI curves for each sample plot.

2.3. Greenhouse gas measurements

Each of the 11 permanent sample plots consisted of a stainless steel collar (60 × 60 cm) that was inserted to a depth of 20 cm into the soil prior to the start of the study. Perforated PVC pipes (internal diameter: 2 cm) were inserted adjacent to each sample plot to measure WT position. Wooden boardwalks were built around the sample plots to minimise damage to the vegetation and to avoid compression of the peat during gas sampling. Data loggers (Hobo External Data Loggers, Onset Computer Corporation, MA, USA) were established at each study site and recorded hourly soil temperatures (°C) at 5, 10 and 20 cm depths. A weather station (Watch Dog Model 2400, Spectrum Technologies Inc., IL, USA) was established at a location half way between SD and RW and recorded PPFD (μmol m^{-2} s^{-1}) and soil temperatures (°C) at 5 and 10 cm depths at 10 min intervals.

CO_2 fluxes were measured from 1st April 2011 to 31st March 2015 at fortnightly (summer) and monthly (winter) intervals and between sunrise and late afternoon (up to 5 measurements per collar per day), using the closed chamber method fully described in Renou-Wilson et al. (2014). Briefly, instantaneous net ecosystem exchange (NEE) was measured by fitting a transparent chamber ($60 \times 60 \times 33$ cm) into a water-filled groove in the collar. A constantly-running fan inside the chamber ensured well-mixed air and a pump circulated cold water into a heat exchanger thus maintaining the air temperature inside the chamber within 2 °C of ambient air temperature (on very rare occasions, freezer packs were also positioned inside the chamber on a ledge). This also prevented water vapour condensation on the walls of the chamber (preventing light attenuation) and reduced the dilution effect due to increased moisture in the chamber (Matsuura et al., 2011). CO_2 concentration within the chamber was measured at 15-s intervals over a period of 60–180s using a portable CO_2 analyser (EGM-4, PP Systems, UK). PPFD was measured by quantum sensor located at the top inside of the chamber (PAR-1 PP systems). Following each NEE measurement, the chamber was vented for a short time to ensure the gas concentration in the chamber was back to ambient levels. Ecosystem respiration (R_{eco}), i.e. the sum of heterotrophic and autotrophic respiration, was then measured by placing an opaque tarpaulin over the chamber. Concurrent to all gas measurements, soil temperature at 5, 10 and 30 cm were measured manually at each collar (soil probe; ELE International, UK) as well as WT position using an electric contact gauge. Transparent polycarbonate extension chambers were used to facilitate gas measurements in cases where the vegetation was higher than 33 cm (Wilson et al., 2007b). Each extension chamber also contained an internal fan to ensure mixing of the air inside the enclosure. Gross primary production (GPP) was calculated as the difference between NEE and R_{eco} values.

CH_4 and N_2O measurements were conducted during the same period and at the same intervals as CO_2 fluxes using an opaque polycarbonate chamber ($60 \times 60 \times 25$ cm) equipped with a battery-operated fan (one measurement per collar per day). Air samples (50 mL) were taken from the chamber at 10, 20, 30 and 40 min following chamber closure. The samples (approximately 30 mL) were injected into pre-evacuated Exetainers® vials (12 mL Soda Glass Vials, Labco, UK). Samples were analysed for CO_2, CH_4 and N_2O at Justus Liebig University Giessen, Germany within 2 months using a gas chromatograph (Bruker 450-GC) and concentrations were calculated using the Galaxie software (Varian Inc., 2006). Flux rates (mg $CO_2\,m^{-2}\,h^{-1}$) were calculated as the linear slope of the CO2 concentration 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. Acceptance of flux calculation and precision are as presented in Renou-Wilson et al. (2014).

2.4. Modelling of fluxes and annual balances

2.4.1. CO_2 flux modelling

Statistical and physiological response models (Alm et al., 2007) were constructed and parameterised for each for each of the two study sites: SD and RW (Table 2). Model evaluation was based on the following criteria; (a) statistically significant model parameters ($p < 0.05$), (b) lowest possible standard error of the model parameters, (c) highest possible coefficient of determination (adjusted r^2) and (d) normality of residuals and residuals evenly scattered around zero. The R_{eco} models, based upon the Arrhenius equation (Lloyd and Taylor, 1994), are non-linear models related to soil temperature. GPP was related to PPFD using the Michaelis–Menten type relationship that describes the saturating response of photosynthesis to light (Tuittila et al., 1999). LAI was also used as an explanatory variable in both GPP and R_{eco} models (e.g. Riutta et al., 2007). Given the homogenous dominant vegetation within each site, we combined data from all sample plots to construct a single model for GPP and R_{eco} for each site. Model coefficients were estimated using the Levenberg–Marquardt multiple non-linear regression technique (IBM SPSS Statistics for Windows, Version 21.0. Armonk, NY, USA). One-third of the data was randomly removed from all data sets prior to modelling and used to independently test the models. During model construction, the relationship between GPP or R_{eco} and the independent environmental variables recorded in conjunction with flux measurements was tested. Only variables that increased the explanatory power of the model were included. In the GPP models, we used PPFD and LAI as explanatory variables (Eq. (1)).

$$GPP = P_{max}\left(\frac{PPFD}{PPFD + k_{PPFD}}\right) * \left[\frac{LAI}{(LAI + a)}\right] \quad (1)$$

where P_{max} is maximum photosynthetic rates; PPFD is photosynthetic photon flux density; k_{PPFD} is the PPFD value at which GPP reaches half its maximum (half saturation constant); LAI is leaf area index and a is a model parameter

In the R_{eco} models, we used soil temperature at 5 cm depth (T_{5cm}) and WT level (Eq. (2)) for Site RW and included LAI (Eq. (3)) for Site SD.

$$R_{eco} = (a + (b * WT)) * \left[\exp\left(c * \left(\frac{1}{T_{REF-}T_0} - \frac{1}{T_{5cm} - T_0}\right)\right)\right] \quad (2)$$

$$R_{eco} = (a + (b * WT)) * \left[\exp\left(c * \left(\frac{1}{T_{REF-}T_0} - \frac{1}{T_{5cm} - T_0}\right)\right)\right] * (\ln(LAI) + d)) \quad (3)$$

where R_{eco} is ecosystem respiration; T_{REF} is reference temperature set at 283.15 K; parameter T_0 is a notional zero-respiration temperature, here set at 227.13 K (Lloyd and Taylor, 1994); T_{5cm} is soil temperature at 5 cm depth; WT is water table depth; LAI is leaf area index; Ln is log-normal and a, b, c and d are model parameters.

Table 2
Comparison of coefficient of determination (R^2) and residuals for the most suitable controls on gross primary production (GPP) and ecosystem respiration (R_{eco}) modelling for both shallow drained (SD) and rewetted (RW) sites. Standard error of the model parameters in parentheses. *not significant p values (i.e. >0.05) signifies that the residuals are normally distributed.

	Shallow-drained (SD)	Rewetted (RW)
	GPP	GPP
Equation #	1	1
N	253	258
Controls	PPFD, LAI	PPFD, LAI
R^2model	0.78	0.72
$R^2$with independent data	0.67	0.64
Residuals p = *	0.2	0.03
P_{max}	4769.70 (258.39)	4086.06 (254.41)
k_{PPFD}	649.50 (71.78)	564.68 (75.24)
a	0.08 (0.01)	0.07 (0.01)
	R_{eco}	R_{eco}
Equation #	3	2
N	180	186
Controls	WT, T_{5cm}, LAI	WT, T_{5cm}
R^2model	0.76	0.71
$R^2$with independent data	0.75	0.70
Residuals p = *	0.09	0.06
a	44.20 (11.17)	413.32 (19.34)
b	−0.63 (0.13)	−10.22 (1.01)
c	256.13 (27.11)	216.23 (23.39)
d	10.58 (1.82)	–

2.5. Reconstruction of annual CO_2–C balance

The sign convention used here consider fluxes relative to the atmospheric pool (i.e. R_{eco} fluxes are positive and GPP fluxes are negative). The response functions estimated for GPP and R_{eco} were used for the annual reconstruction of NEE. In combination with an hourly time series of (1) PPFD and T_{5cm}, recorded by the weather station and data loggers, (2) modelled LAI (see Section 2.2) and (3) WT depths linearly interpolated from weekly measurements, GPP and R_{eco} fluxes were reconstructed for each sample plot. NEE was then calculated on an hourly basis as the sum of GPP and R_{eco}. Negative NEE values indicated a net uptake of CO_2 from the atmosphere by the peatland and positive values indicated a net loss of CO_2 to the atmosphere. The annual CO_2–C balance ($g\,C\,m^{-2}\,yr^{-1}$) was calculated for each sample plot by integrating the hourly NEE values over each 12-month period (Year 1: April 1st 2011–March 31st 2012; Year 2: April 1st 2012–March 31st 2013; Year 3: April 1st 2013–March 31st 2014; Year 4: April 1st 2014–March 31st 2015).

2.6. Annual CH_4 and N_2O balances

Fluxes were calculated by linear regression (time enclosure being small) although this may still carry the risk of underestimating fluxes compared to non-linear functions (e.g. Pihlatie et al., 2013). CH_4 fluxes were transformed before statistical analysis but no significant relationship was found with any of the environmental parameters measured. Therefore, daily fluxes were calculated by linear interpolation between sampling measurement points for each collar and summed over the annual sampling period 1 April–30 March for the four years.

2.7. Calculation of terrestrial net ecosystem carbon balance and global warming potential

A terrestrial net ecosystem carbon balance (NECB) can be derived from adding gaseous CO_2 and CH_4 emissions (sources or sinks) to the C sources linked with biomass removal and livestock grazing (for the first two years). A nationally derived CH_4 emission rate for enteric fermentation for suckler cows was utilised in the calculation (see O'Mara et al., 2007): 74 kg $CH_4\,head^{-1}\,yr^{-1}$. Combining this figure with stocking rates yielded emissions of 3.6 $g\,C\,m^{-2}\,yr^{-1}$ for both sites. Given the very low stocking rates, we assume C import–export from on-site manure negligible. In order to fully examine the impact of management activities (rewetting and grazed/ungrazed) on the GHG dynamics of the study sites, CH_4 and N_2O fluxes were converted to CO_2 equivalent (t $CO_2eq\,ha^{-1}\,yr^{-1}$) according to their global warming potentials (GWP) over a 100-year horizon and including climate-carbon feedback: $CH_4 = 34$ and $N_2O = 298$ (Myhre et al., 2013). GWP was calculated for each soil ecosystem and therefore did not include CH_4 emissions from livestock and C-export from biomass.

2.8. Statistical and uncertainty analysis

After testing for normality, GLM-Anova, repeated measures analyses were performed on measured and modelled CO_2 and CH_4 fluxes as well as on environmental data to test for time (year and month) and site effects. Non-parametric tests (Friedman and Mann–Whitney U test) were used for non-normally distributed data and for larger environmental data sets at plot level. The significant threshold used in this study was $p \leq 0.05$ (IBM SPSS Statistics for Windows, Version 21.0. Armonk, NY. USA).

Standard errors of estimates are shown in parentheses for all fluxes Reco, GPP, NEE. One standard deviation is shown in parentheses for all other components, except for $T_{5\,cm}$ which was continuously recorded at site level and not plot level. Positive values indicate a loss of carbon from the site and negative values indicate an uptake of carbon by the site.

The statistical uncertainties of the models underlying reconstructed annual GPP and R_{eco} was estimated using the standard error of the estimation (e.g. Aurela et al., 2002) where the model's standard error (see Eq. (4)) is expressed as a percentage of the mean fluxes which is then applied to the annual balance.

$$E_r = \sqrt{\sum_{i=1}^{n} \frac{(F_{obs} - F_{mod})^2}{(n-1) * n}} \quad (4)$$

where E_r is the model standard error, F_{obs} is the measured flux and F_{mod} is the modelled flux and n the total number of measured fluxes.

This error estimate represents the combined effect of the random errors due to statistical uncertainties of measurements and the scatter in the model results. As NEE is not directly modelled, uncertainty in the annual 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}.

3. Results

3.1. Weather and site conditions

3.1.1. Seasonal and inter-annual weather variation

Annual precipitation between April 2011 and March 2015 (4 years) was higher than the 30-year average precipitation (1980–2010 Malin Head Meteorological Site) (Fig. 1a and d). While monthly precipitation was typical of a maritime temperate climate with a wetter period during the colder months, annual precipitation was *c.* 10% higher than the 30-year average, except for Year 3 when 20% more rain than the long-term average was recorded. In

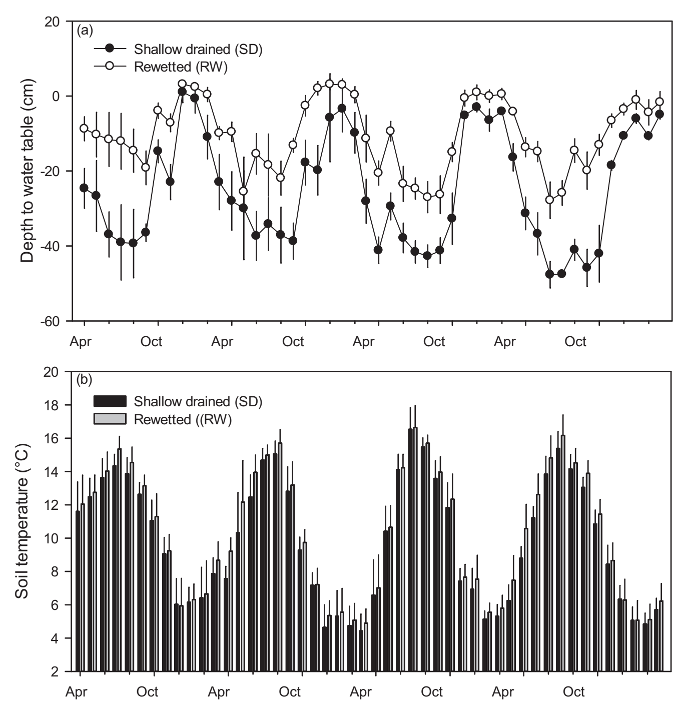

Fig. 2. Monthly means of (a) water table levels (cm) and (b) soil temperatures (°C) at 5 cm depth at the shallow drained (SD) and rewetted (RW) sites. Error bars are standard deviation of the monthly means.

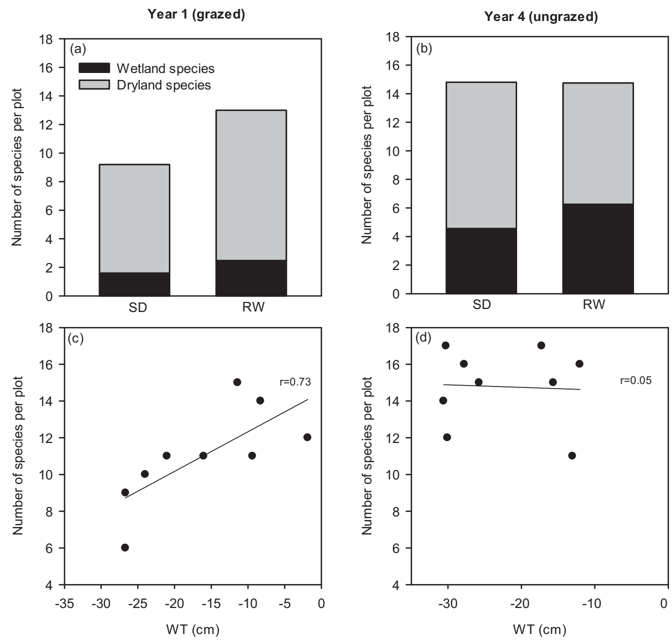

Fig. 3. Total number of species per plot in shallow drained (SD) and rewetted (RW) sites in (a) Year 1 (grazed) and (b) Year 4 (ungrazed); total bar height represents total number of species, black bar represent number of wetland species, grey bar represent number of dryland species. Correlation between total number of species per plot and water table (WT) levels (cm) during (c) Year 1 (grazed) and (d) Year 4 (ungrazed).

Year 3, the months of December, January and February received 67% more rain than 30-year seasonal average.

The seasonal pattern for air temperature was similar for all years and the 4-year monthly means closely following the long-term averages (Fig. 1b). Monthly mean PPFD typically peaked in June–July with the exception of Year 2 when it peaked in May and was followed by remarkably dim months of June and July (15% lower than the 4-year average) (Fig. 1c). Inter-annual variability was noticeable for both air temperature and PPFD with Year 2 being significantly colder than any other year (0.8 °C less than the 30-year average) (Anova repeated measures; $p < 0.001$) (Fig. 1e), and having the lowest PPFD (though not significantly different between years) (Fig. 1f).

3.1.2. Water table

WT levels reflected the seasonality in precipitation (Fig. 2a) with highest levels in winter (+5 cm in RW and +3 cm in SD) and lowest in summer time (−37 cm in RW and −50 cm in SD). Monthly means were significantly different ($p < 0.001$) between the two sites with 4-year averages of −25.9 cm and −11.3 cm for SD and RW respectively. The results of the Friedman Test indicated that there was a statistical difference between years at each site ($p < 0.001$). Inspection of the annual median values showed an annual increase in WT depth at both sites from Year 1 through to Year 4 as: −24, −28, −31, −34 cm in SD and −8, −10, −12, −14.5 cm in RW.

3.1.3. Soil temperature

Seasonal patterns of soil temperature at 5 cm depth were similar for both sites and peaked in July in all years, except Year 2 when it peaked in August (Fig. 2b). The coldest period was always observed in December/January. Anova repeated measures analysis concluded that soil temperatures were significantly different over the 4 years ($p < 0.05$) and between the two sites ($p < 0.001$) while the interaction was not significant ($p = 0.57$). Pair-wise comparison showed that the colder Year 2 was significantly different to Years 1 and 4. On a seasonal scale, the inter-annual variability was small

e-

Table 3
Summary of annual sums or averages of various variables from the shallow drained (SD) and rewetted (RW) sites. Standard errors of estimates are shown in parentheses for all fluxes R_{eco}, GPP, NEE and one standard deviation is shown in parentheses for all other components, except for T_{5cm} which was continuously recorded at site level and not plot level. Positive values indicate a loss of carbon from the site and negative values indicate an uptake of carbon by the site.

	Year 1		Year 2		Grazed		Year 3		Year 4		Ungrazed	
	SD	RW	SD	RW	SD	RW	SD	RW	SD	RW	SD	RW
T_{5cm}	10.4	10.8	9.1	9.8	9.7	10.3	10	10.4	9.8	10.5	9.9	10.4
WT	−22.8 (4)	−7.3 (3)	−24.9 (4.1)	−10.1 (2.8)	−23.6	−8.7	−27.1 (1.4)	-13.9 (1.9)	−28.9 (1.8)	−13.9 (1.8)	−28	−13.9
LAI	0.25 (0.01)	0.23 (0.01)	0.23 (0.01)	0.27 (0.01)	0.24	0.25	1.82 (0.8)	0.87 (0.3)	1.71 (0.8)	1.36 (0.5)	1.76	1.08
GPP	−1302 (20)	−1202 (19)	−1217 (19)	−1220 (19)	−1259	−1211	−1439 (23)	-1490 (24)	−1677 (27)	−1553 (25)	−1558	−1521
R_{eco}	1393 (22)	1286 (20)	1280 (20)	1306 (21)	1336	1297	1596 (26)	1491 (24)	1753 (28)	1473 (23)	1674	1482
NEE	90 (30)	85 (28)	64 (28)	86 (296)	77	85	87 (34)	0.80 (33)	76 (39)	-80 (34)	81	−40
Biom	174 (25)	137 (29)	137 (28)	113 (24)	155	125	-	-	-	-	-	-
CH_{4eco}	1.50 (1.1)	9.47 (6.5)	0.87 (1.0)	8.95 (5.6)	1.19	9.21	1.85 (2.5)	5.41 (4.0)	1.06 (1.0)	3.39 (2.0)	1.45	4.40
CH_{4live}	3.6	3.6	3.6	3.6	3.6	3.6	-	-	-	-	-	-
NECB	270 (59)	235 (82)	205 (32)	211 (83)	237	223	158 (92)	6 (97)	77 (52)	−76 (52)	118	−35

T_{5cm} = Soil temperature at 5 cm depth in °C. WT = water table depth in cm. LAI = Leaf Area Index in $m^2 m^{-2}$. NEE = net ecosystem exchange, GPP = gross photosynthesis production, R_{eco} = ecosystem respiration, Biom = biomass export, CH_{4eco} = CH_4 fluxes from ecosystem, CH_{4live} = CH_4 emissions from enteric fermentation of livestock, NECB = terrestrial net ecosystem carbon balance (without fluvial losses). All in g C m^{-2}. Grazed = mean of Years 1 and 2; Ungrazed = mean of Years 3 and 4.

xcept for spring in Year 3 when the soil temperatures were on average 3.8 °C (SD site) and 4.8 °C (RW site) lower than the 4-year average.

3.1.4. Vegetation composition

The composition of plant species was similar between the two sites in terms of dominant species but differed in terms of total number and over time. At the beginning of the study, the rewetted plots had on average 13.0 ± 1.5 species compared to 9.4 ± 1.8 in the SD plots (Fig. 3a). *Lolium perenne*,*Ranunculus bulbosus* and *Rumex acetosa* were found exclusively in SD while *Anthoxanthum odoratum*, *Bellis perennis* and *Trifolium pratense* were exclusively found in RW. *Juncus* spp. was present in all RW plots but only in two of the SD plots. At the end of the study period (after 2 years of no grazing), both sites displayed a similar increased average species number of 14.8 (Fig. 3b). This sharp increase in species numbers in SD was due to new herb species (*Bellis perennis*, *Taraxacum officinale*) but mainly wetland species (*J. bulbous, J. squarrosus, J. articulatus*) while shrubs (*Alnus glutinosa* and *Salix* spp.) appeared at both sites. During the grazing period, the total number of species decreased with WT depths (i.e. the drier the fewer species) (Fig. 3c) but this relationship disappeared once grazing was removed (Fig. 3d). Between Year 1 and Year 4 the wetland species cover increased from 16 to 48% in the SD plots and from 33 to 80% in the RW plots and this was predominantly due to rapid growth and spread of *J. effusus*. It is worth noting the appearance of two new moss species during the fourth year: *Polytrichum commune* and *Fissidens taxifolius* at both sites. While SD plots appeared more homogenous than RW plots, some SD plots distinguished themselves by retaining grassy species such as *Holcus lanatus* and *Agrostis stolonifera*.

3.1.5. Biomass

During the first two-year period when grazing was simulated by cutting, the cumulative production of biomass was superior at the SD site while the RW site was on average 20% less productive (Table 3). The colder temperatures and lowered PPFD values in Year 2 led to poor growth, particularly during the late summer months. This resulted in a decrease of 20% and 18% in annual productivity at the SD and RW sites respectively. LAI differed significantly between sites over the 4-year period. At both sites, the monthly LAI values did not differ between Years 1 and 2 when the biomass was regularly cut but significantly increased in Years 3 and 4 (Table 3). In the SD site, Years 3 and 4 did not differ significantly, in contrast with the RW site where the mean annual LAI was doubled in Year 4 compared to Year 3.

3.2. Models and CO_2 exchange patterns

At both sites, GPP was strongly dependent on PPFD but the addition of the LAI term further improved the explanatory power of the model (Eq. (1) SD: $r^2 = 0.78$ and RW: $r^2 = 0.72$, Table 2). The relationship between predicted and observed GPP fluxes was good (Fig. 4) with the good accuracy of predictions based on the independent test data (SD: $r^2 = 0.67$ and RW: $r^2 = 0.64$, Table 2).

R_{eco} was strongly dependent on T_{5cm} and WT ($r^2 = 0.71$ at both sites). At SD however, the residuals of the model were not normally distributed and the addition of the LAI term improved the explanatory power of the model (Eq. (3), $r^2 = 0.76$, Table 2). The independent test data demonstrated high prediction accuracy (SD: $r^2 = 0.75$ and RW: $r^2 = 0.70$) while the overall relationship between observed and modelled R_{eco} showed a slight overestimation of higher fluxes in the SD site (Fig. 4).

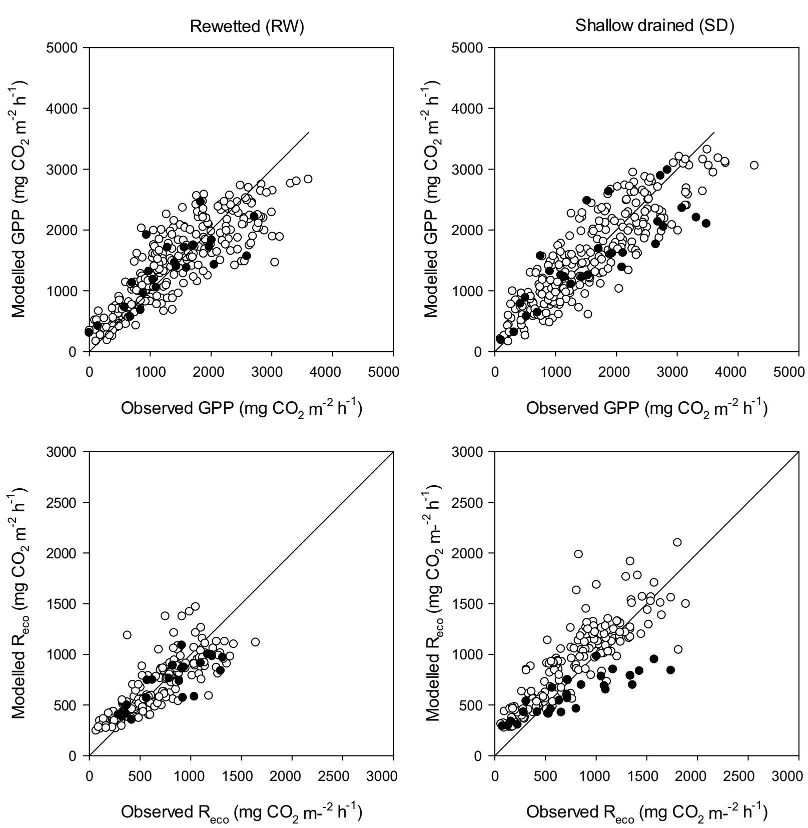

Fig. 4. Relationships between observed and modelled gross primary production (GPP) and ecosystem respiration (R_{eco}) fluxes (mg CO_2 m^{-2} h^{-1}) for the rewetted and shallow drained sites. White circles indicate data used in the construction of the models and dark circles indicate independent test data. Lines indicate 1:1 accordance between observed and modelled fluxes.

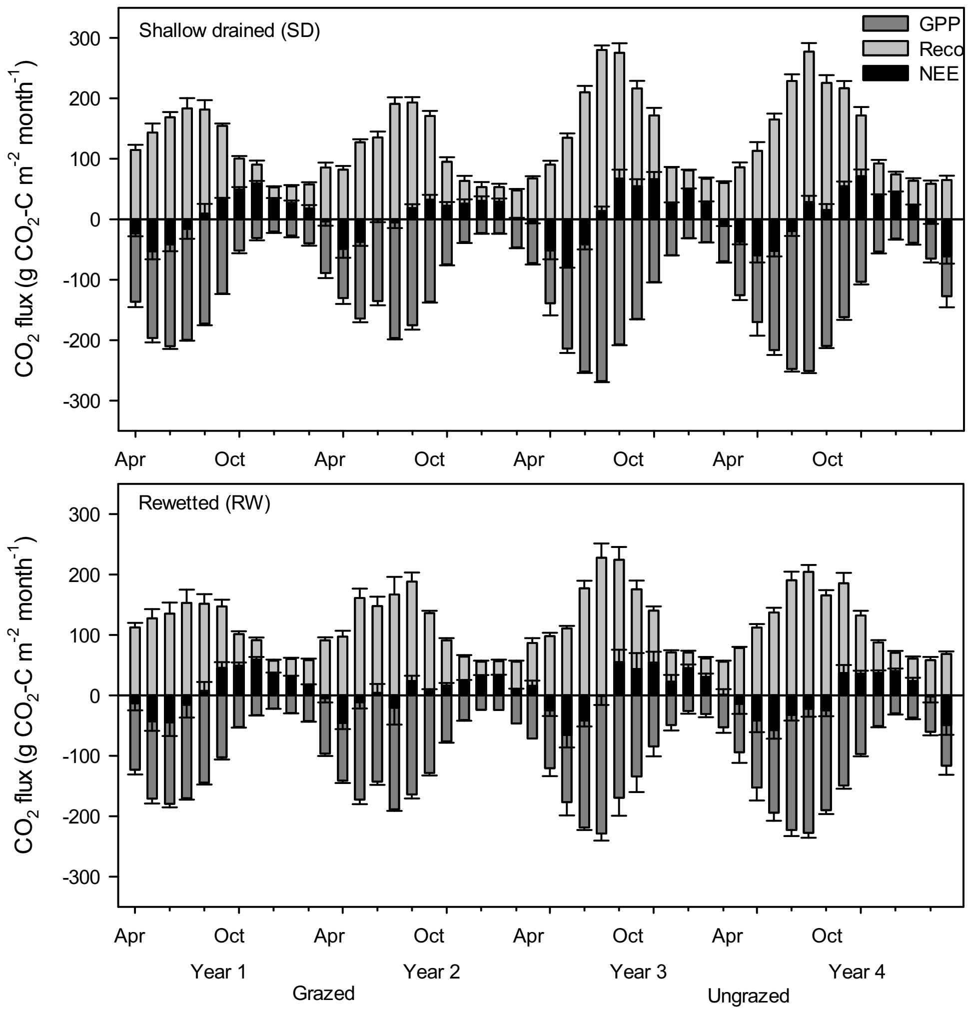

Fig. 5. Monthly mean gross primary production (GPP), ecosystem respiration (R_{eco}) and net ecosystem exchange (NEE) (g CO_2–C m^{-2} $month^{-1}$). Error bars represent standard deviation on the spatial variability; n = 5 for shallow drained and n = 6 for rewetted site. Negative NEE values indicate that the site was a net sink for CO_2–C for that month.

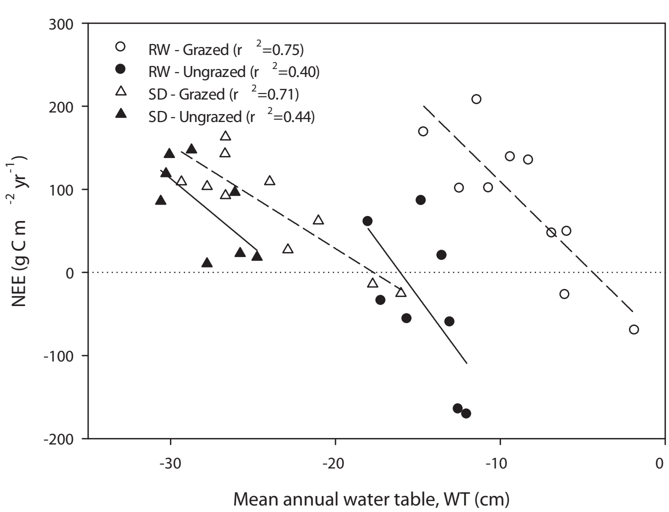

Fig. 6. Relationships between mean annual Net Ecosystem Exchange (NEE) (g CO_2–C m^{-2} yr^{-1}) and mean annual water table (WT, cm) at the rewetted (RW, n = 6) and shallow drained (SD, n = 5) sites. Linear regression lines: dashed = grazed and solid = ungrazed.

Modelled fluxes of GPP and R_{eco} demonstrated typical seasonality in all 4 years (Fig. 5). GPP peaked in July regardless of sites and year except in Year 2 in RW site when it peaked in August. There was no net CO_2 uptake at either site from August (when temperatures decreased) until February. On an individual plot basis, the largest monthly net CO_2 losses occurred at both sites (94 and 78 g C m^{-2} $month^{-1}$ at SD and RW respectively) in August Year 3 (following a dry and warm July). This dry spell led to the largest R_{eco} and GPP values over the 4 year period. The highest net CO_2 uptake (monthly and per plot basis) were recorded during May Year 3 at both sites.

The SD site displayed a wider range of fluxes with R_{eco} being on average c. 10% higher in SD than RW. At plot level, annual NEE in the SD site had similar range regardless of grazing regime: from −25 to +163 g C m^{-2} yr^{-1}. Annual NEE in the RW site under the grazing regime ranged from −70 (sink) to +170 (source) g C m^{-2} yr^{-1}. When ungrazed, annual NEE in the RW site ranged from −170 to +86 g C m^{-2} yr^{-1}. NEE showed a strong relationship with mean annual WT levels (Fig. 6) during the grazing period at both sites albeit the relationship was weaker during the ungrazed period, due to a narrower range in WT values.

While modelled R_{eco} and NEE fluxes were normally distributed, the two-way repeated measures ANOVA test showed significant effects between the factors 'sites' and 'grazing' but also a significant interaction between 'site' and 'grazing' for both sets

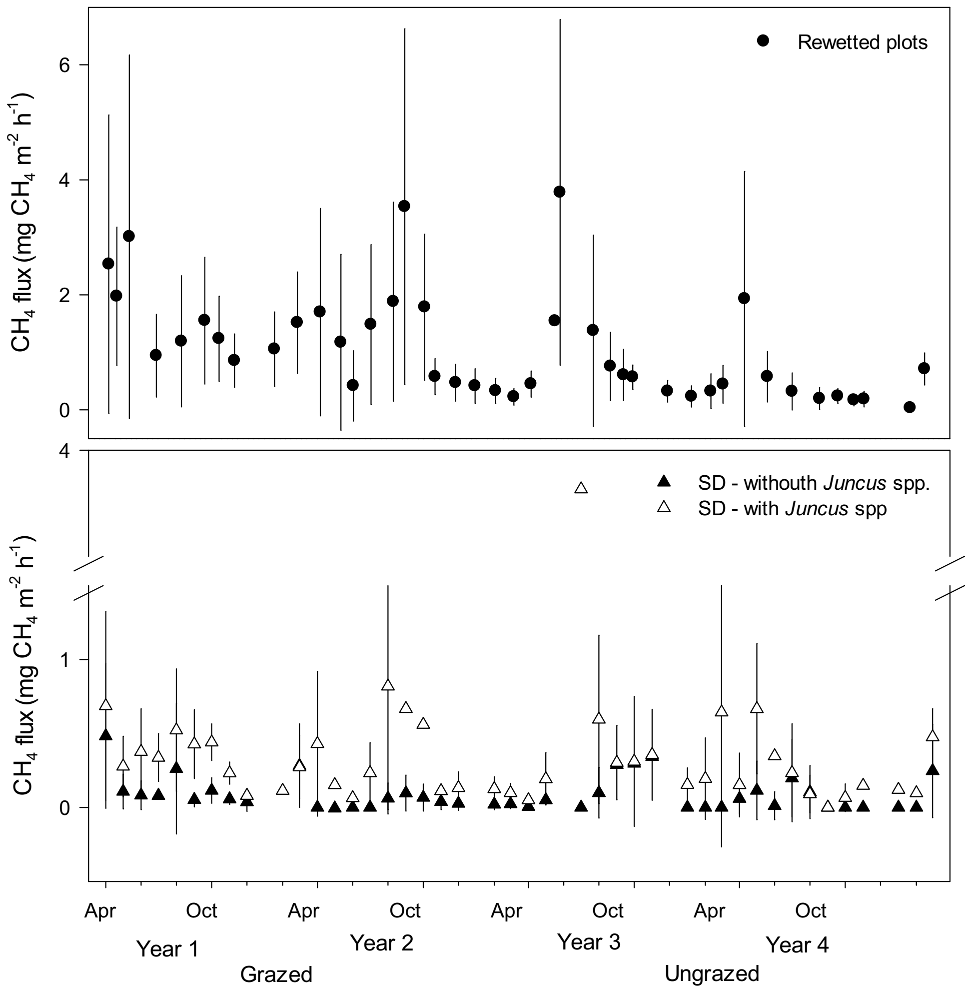

Fig. 7. Measured monthly means of methane (CH_4) fluxes (mg CH_4 m^{-2} h^{-1}) ± one standard deviation at the rewetted site (RW, $n = 6$) and shallow drained (SD, $n = 5$) site where collars were split according to presence/absence of *Juncus* spp. at the onset of the study; presence = open symbols, absence = closed symbols.

of fluxes. The data was therefore split between groups to test first for site effect. The sites did not differ during the 'grazed' period in terms of annual NEE and R_{eco} fluxes ($p = 0.608$ and $p = 0.189$ respectively). During the 'ungrazed' period, both NEE and R_{eco} were significantly different between sites ($p < 0.01$). Friedman test results demonstrated also that annual GPP fluxes were not significant between sites during the grazed period ($\chi^2(1, n = 10) = 1.6$, $p = 0.206$; M(SD) = −1259; M(RW) = −1209), but were significantly different during the 'ungrazed' period ($\chi^2(1, n = 8) = 4.5$, $p = 0.03$; M(SD) = −1675; M(RW) = −1521).

Grazing affected CO_2 fluxes, with both R_{eco} and GPP fluxes significantly different between grazed and ungrazed at both sites ($p < 0.01$). The cessation of grazing allowed GPP to increase by an average 24% at SD and by 26% at RW (Table 3), while R_{eco} increased by an average 25% in SD and 14% in RW. This did not affect NEE at the SD site to any great extent while NEE significantly decreased at the RW site during the 'ungrazed' period. Overall, both sites were a source of CO_2–C when grazed and modelled NEE values did not differ significantly ($p = 0.632$). Conversely, when grazing stopped the sites differed significantly: SD remained a C source (2-year

Table 4
Emission factors (EF) for CO_2, CH_4 and N_2O (units follow IPCC, 2014) and global warming potential (GWP) from the shallow drained (SD) and rewetted (RW) sites for each period: grazed and ungrazed. CO_2 EF includes NEE and C export in biomass. GWP is the sum of NEE, CH_4 and N_2O exchange (where CH_4 exchange includes contribution from the ecosystem only). Negative sign indicates uptake by the atmosphere and positive signs emissions to the atmosphere.

Site treatments				
Emission factors	SD- grazed	SD-ungrazed	RW-grazed	RW-ungrazed
CO_2EF (t C ha^{-1} yr^{-1})	2.32	0.81	2.1	−0.4
CH_4EF (kg CH_4ha^{-1} yr^{-1})	15.8	19.4	122.8	58.7
N_2O EF (kg N_2O–N ha^{-1} yr^{-1})	0	0	0	0
GWP (g CO_2eq m^{-2} yr^{-1})[a]	336	364	730	54

[a] IPCC standards were used with a radiative forcing factor of 34 for CH_4, 298 for N_2O and a time horizon of 100 years.

mean: 81.5 g C m^{-2} yr^{-1}) but RW was either neutral (Year 3) or a sink (Year 4) (2-year mean: −40 g C m^{-2} yr^{-1}).

3.3. CH_4 and N_2O fluxes

At the SD site, measured CH_4 fluxes were low during the whole 4 year period and did not significantly differ between the grazing and no grazing regimes ($p = 0.20$), averaging 0.206 mg CH_4 m^{-2} $h^{-1}$and ranging from −0.078 to +3.816 mg CH_4 m^{-2} h^{-1}. The later maximum fluxes were measured from plots containing *Juncus* spp (*J. effusus, J. articulatus* and/or *J. bulbosus*) which consistently emitted higher CH_4 amounts in comparison to plots without *Juncus* spp. (Fig. 7). Measured CH_4 fluxes in RW were on average 5 times higher than at the SD site and were significantly different over time (Chi 2 (3, $n = 5$) = 8.1, $p = 0.03$) with the highest fluxes (max: 9.42 mg CH_4 m^{-2} h^{-1}) recorded during the grazed summer period (Fig. 7). The two sites were significantly different throughout the monitoring period (Chi (1, $n = 20$) = 17 $p < 0.001$). While no significant regression correlation were found between the measured CH_4 fluxes and environmental parameters, measured CH_4 fluxes in RW increased with WT levels during the 'warm' season (defined arbitrarily as the period when soil temperatures at 5 cm depth were at or above 10 °C) (data not shown). Modelled CH_4 balances were not significantly different over time in the SD site (Table 3). At the RW site, annual CH_4 balances were significantly different between grazed and ungrazed (Chi (1, $n = 8$) = 4.5 $p < 0.05$). The average annual balance was 9.21 (SD 0.26) g CH_4–C m^{-2} yr^{-1} for grazed and 4.40 (SD 1.0) g CH_4–C m^{-2} yr^{-1} for ungrazed (Table 3).

Measured fluxes of N_2O were not distinguishable from zero at both sites during the four year monitoring period.

3.4. Calculation of NECB and GWP

The terrestrial NECB of SD was a relatively small source each year but recorded the highest annual NECB (237 (SD 45) g C m^{-2} yr^{-1}) (Table 3). However the NECB was halved during the 'ungrazed' period compared to the grazed period (Table 4). The terrestrial NECB of the RW site was similar to the SD site during the 'grazed' period but became a small sink −35 (SD 74) g C m^{-2} yr^{-1} during the 'ungrazed' period (Table 3).

GWP was estimated in the SD site at 336 g CO_2eq m^{-2} yr^{-1} during the grazed period and 364 g CO_2eq m^{-2} yr^{-1} during the ungrazed period. GWP at RW was the highest of all when 'grazed' at 730 g CO_2eq m^{-2} yr^{-1} but was reduced considerably to 248 g CO_2 eq m^{-2} yr^{-1} in the first year of no grazing. The site had a net climate cooling effect in the second year of no grazing at −139 g CO_2eq m^{-2} yr^{-1} (giving an average for the no grazing of 54 g CO_2eq m^{-2} yr^{-1}) (Table 4).

4. Discussion

4.1. Magnitude and significance of CO_2 fluxes

4.1.1. Impact of edaphic properties

During the two-year grazing period, the shallow drained and rewetted sites had similar annual GPP, R_{eco} and NEE values and were both a small source of CO_2. These results are comparable to NEE values reported for extensively grazed nutrient-poor moorland also in a maritime temperate climate (Beetz et al., 2013) but lower than other drained organic soils elsewhere in the temperate region where larger losses have been reported (Elsgaard et al., 2012; Renou-Wilson et al., 2014; Eickenscheidt et al., 2015). The edaphic properties (low fertility and annual mean WT above −25 cm) and management systems (low stocking density and low inputs) at the sites in this study resulted in much lower CO_2 emissions than has been reported for nutrient rich, deep drained sites (Drösler et al., 2013; Renou-Wilson et al., 2014). This study thereby confirmed that CO_2 emissions are likely to be relatively low if grassland production over organic soils is continued through extensive grazing, while keeping the mean annual WT above −25 cm (Renger et al., 2002; Regina et al., 2014).

Despite being grazed, annual NEE in the rewetted site is within the published values of typical rewetted sites (Blain et al., 2014). The significant inverse correlation between annual NEE and WT at the plot level (Fig. 6) demonstrates that the grazed rewetted plots were a CO_2 sink when the annual mean WT was above −5 cm, therefore confirming that the definition of successful rewetting in such a context should include the maintenance of the mean annual WT levels just below the surface over time (Blain et al., 2014). While R_{eco} values at our rewetted site are much lower than the drained site and indeed other drained organic soils under grasslands in Ireland (Renou-Wilson et al., 2014), they are still a third higher than a Dutch rewetted peat meadow for example (Hendriks et al., 2007). Such differences could be attributed to mild winter temperatures (mean air temperature above 5 °C) in the non-growing season and to the vegetation management regime carried out at our sites. The high WT induced a higher species diversity in the rewetted site (Fig. 3a) that consisted mainly of forbs and wetland species which, upon regular cutting, grow slower than typical grasses, a greater cover of which was found in our shallow drained site.

4.1.2. Impact of management shift

The management shift to 'no grazing' led to a significant increase in LAI, which in turn led to much larger GPP and R_{eco} fluxes than during the grazing period (Fig. 5). These fluxes are much higher than those recorded in natural or extensive wet grasslands (Beetz et al., 2013) and perhaps reflect a rapidly-changing state, with the vegetation allowed to develop fully throughout the growing season and die 'on-site'. In the shallow drained site, the increased LAI was a control on both GPP and R_{eco} and therefore had a neutral impact on NEE. In the rewetted site, however, the vegetation growth and succession pattern (very tall *Juncus effusus* specimens and increased shrubs and mosses) led to greater increases in GPP compared to R_{eco}, which was controlled by WT levels and soil temperatures. Overall, the vegetation (LAI) was the main control of NEE in the ungrazed rewetted site, which was either close to neutral or a sink (−80 g C m^{-2} yr^{-1}). This fits well with other rewetted study sites (Beetz et al., 2013; Herbst et al., 2013) and supports the hypothesis that such sites should behave similarly to natural organic soils (Sottocornola and Kiely, 2005; Blain et al., 2014). While mean annual WT was a good indicator of NEE at both sites when grazed (Fig. 6), the range of WT values was smaller during the 'no grazing' period thereby leading to a weaker relationship with NEE. However, the steep slope during the ungrazed period at the RW site, suggests a stronger sensitivity of NEE to WT. Thus, raising the WT higher (closer to the surface) while removing livestock could increase the climate mitigation benefit in terms of CO_2 fluxes.

4.1.3. Inter-active impact of management and inter-annual/seasonal weather variability

The annual weather pattern was relatively stable during the four year period with only two types of deviations from the long-term average: (1) colder and wetter periods and (2) dimmer periods. While cold winters or late snow has been shown to decrease summer NEE sinks (Herbst et al., 2013; Helfter et al., 2015), cold and wet weather during the growing season could impact CO_2 fluxes differently, primarily affecting R_{eco}. This was discernible in Year 3 in particular, when a cold and wet spring kept

R_{eco} very low without impacting GPP (Fig. 5). An earlier onset of cold wet weather in the autumn of Year 2 also reduced R_{eco} at both sites. However, this translated into a much reduced annual R_{eco} balance at the drained site only. This may be partially explained by the fact that soil temperature is more readily affected by air temperature when the WT level is deeper in the soil, as drier soils radiate heat more rapidly thus resulting in colder temperatures, while wetter soils can remain stable for longer (Collins and Cummins, 1996). While both soil temperature and WT explained R_{eco} flux variability at both sites, WT could be the critical driver at the rewetted site. WT level was also found to be a dominant driver of long-term variability in CO_2 fluxes in a Scottish peatland (Helfter et al., 2015).

Interestingly, CO_2 fluxes were not influenced by increased precipitation in the winter time (Year 3), a predicted climate change scenario for this region (McGrath and Lynch, 2008). During the cold, non-growing period, the WT is habitually close the surface and due to the gentle slope, flooding is not occurring. Therefore, weather variation did not control the range of fluxes during the winter period as seen in other maritime peatland ecosystems (Laine et al., 2009). This is in contrast with summer time when weather variability and management activities affecting the vegetation had an inter-active influence, making it a critical period for net CO_2 fluxes. Low summer GPP were recorded in response to low PPFD values, with examples of especially 'dim' periods in August Year 3 and in June and July in Year 2. In the latter, overall biomass production was significantly reduced by 20% and 17% in SD and RW respectively compared to Year 1. Thus such weather variation particularly affected annual emission factor (EF).

4.2. Uncertainties and CO_2 emission factors (EF_{CO2})

There is still a large uncertainty associated with CO_2 fluxes from managed organic soils (IPCC, 2014) and from rewetted soils in particular, given the wide range in previous land uses, source of incoming water and associated WT fluctuations, and changing management activities, sometimes on an annual basis (e.g. Herbst et al., 2013). Here we publish four years of continuous measurements and the statistical response models developed and published from the first 2 years data in the shallow drained site (Renou-Wilson et al., 2014) are therefore different to the model developed for this study. While the long-term, larger datasets improved slightly the statistical significance of the models (r^2 increased from 0.72 to 0.78 for GPP and from 0.75 to 0.76 for R_{eco}), the standard errors of parameters were reduced (data not shown). Moreover, the temporal resolution of our datasets was high enough to capture a wide range of explanatory values affecting CO_2 flux dynamics (Görres et al., 2014), thus reducing the overall uncertainties in the annual balances and thereby increase the accuracy of the derived EFs. EFs can be calculated from the two 'grazed' sites by adding NEE to the C losses from removed biomass (assuming instantaneous emission of the cut biomass) and averaged over the multi-annual measurement period. When grazed, the CO_2 EFs for the shallow drained and rewetted sites were 2.32 and 2.10 t CO_2–C ha^{-1} yr^{-1} respectively and fall below the lower 95% confidence interval of EFs for temperate drained nutrient-poor grassland sites in the IPCC (2014) guidance (2.8–12.2) t CO_2–C ha^{-1} yr^{-1}). Nevertheless, other studies have reported low EF for extensive grasslands over organic soils, and in Denmark, for example, an EF of 1.25 t CO_2–C ha^{-1} yr^{-1} (National Environmental Research Institute, 2013) is applied to non-fertilised permanent grassland. With no grazing, the CO_2 EFs for the SD and RW sites were 0.81 and −0.40 t CO_2–C ha^{-1} yr^{-1} respectively, with the latter comparable to the IPCC default EF for rewetted temperate, nutrient-poor organic soils (−0.23 t CO_2–C ha^{-1} yr^{-1}).

4.3. Methane fluxes

4.3.1. Inter-active effects of WT and vegetation on CH_4 fluxes

In general, despite large temporal and spatial variation, the estimated annual CH_4 balances were in accordance with other studies (Petersen et al., 2012; Beetz et al., 2013; Schrier-Uijl et al., 2014; Beyer and Höper, 2015). The management shift did not affect CH_4 emissions from the SD site with an annual CH_4 balance of 1.19 (SD 1.09) g C m^{-2} yr^{-1} for the grazed period compared to 1.45 g C m^{-2} yr^{-1} ± 1.2 during no grazing. Contradictory studies have been published on the effect of the cutting of vegetation on CH_4 emissions (Schafer et al., 2012; Noyce et al., 2014) and this may be attributed to the residual height of aerenchymous plants (Kelker and Chanton, 1997). While, in this study, the action of cutting the vegetation to simulate livestock grazing may not have reflected livestock grazing preferences (e.g. *Juncus* is unpalatable to cows and is left untouched if other food sources are available), the traditional management in this region is to top the field and gather and burn the cut vegetation. The end effect is therefore similar to our experimental plots with short stems of *Juncus* re-growing rapidly. As no grazing effect on CH_4 emissions was recorded at our shallow drained site, the decrease in CH_4 emissions at the rewetted site during the no grazing period are likely to be due to lower WT levels in years 3 and 4 (compared to years 1 and 2). At our rewetted site, WT inversely affected CH_4 emissions during the growing season over the 4 year period (Fig. 7) and our highest annual CH_4 balance (9.47 g CH_4 m^{-2} yr^{-1}) was recorded when the mean annual WT was the closest to the surface (−7 cm, Table 3). However this effect could have been compounded by (a) the vegetation composition and (b) the management shift. Vegetation composition has been identified as the most important driver for CH_4 fluxes in both drained (Couwenberg, 2009) and rewetted sites (Samaritani et al., 2011; Vanselow-Algan et al., 2015; Zak et al., 2015). In the shallow drained site, CH_4 emissions were low especially when aerenchymous plant species were absent. CH_4 emissions from *Juncus*-containing plots were higher than *Juncus*-free plots, as observed in similar ecosystems (Petersen et al., 2012; Schafer et al., 2012) and did not show distinctive seasonal trends but rather emitted regular peaks,which has been observed in a number of studies (Hendriks et al., 2010; Schafer et al., 2012; Henneberg et al., 2015). *Juncus* spp. are typically found in poorly drained soils in Ireland and the species is well-known for its capacity to move CH_4 directly from the anoxic rooting zone to the atmosphere (Garnet et al., 2005; Ström et al., 2005; Whalen, 2005). During the first year of the management shift to no grazing, very high CH_4 emissions were recorded in early summer when air temperatures and PPFD were higher-than-average and corresponded to the dramatic growth of *Juncus effusus* tussocks (height 140 cm) and the colonisation of new *Juncus* spp. in all the shallow drained plots in particular. This has been observed in similar passive restoration of wet/riparian grassland where the removal of cattle led to a switch in vegetation composition to large hydrophitic plants (Batchelor et al., 2015). CH_4 emissions in ungrazed sites could be influenced indirectly through the competitive interaction between species (Moran et al., 2008), as well as through the availability of fresh plant litter (as the plants are not cut but allowed to die off in-site) (Ström et al., 2015). Emissions decreased significantly at both sites in the second year of no-grazing (Year 4) when lower WT levels were recorded.

4.3.2. Uncertainty and emission factor for CH_4 (EF_{CH4})

While the measurement intervals and interpolation approach can induce some uncertainty in the annual budgets of CH_4 due to potential 'peaks' being missed, the rather low stocking level, the relatively stable weather patterns and the lack of management activities (no fertilisation) expectedly reduce these 'peaks' somewhat in this study. In addition, efforts were made to carry out additional measurements immediately after heavy rainfall events when large CH_4 emissions have been recorded at other sites due to rapid WT fluctuations (Olson et al., 2013; Günther et al., 2014). Together with a more robust statistical analysis, the 4-year dataset helped reduce bias in CH_4 balances that are typically estimated from one or two-years datasets (Günther et al., 2014). The calculated EF for the shallow drained site was 15.8 and 19.4 kg CH_4–C ha^{-1} yr^{-1} when grazed and ungrazed respectively. This is lower than the IPCC (2014) default EF of 29.2 kg CH_4–C ha^{-1} yr^{-1} for shallow-drained nutrient-rich organic soils (there is no EF for shallow drained nutrient-poor (IPCC, 2014)). For the rewetted site, the EF was higher at 122 kg CH_4–C ha^{-1} yr^{-1} when grazed, but was substantially reduced to 58 kg CH_4–C ha^{-1} yr^{-1} during the 'no grazing' period. This EF is lower but within the 95% range of the IPCC default EF (Blain et al., 2014) for rewetted temperate nutrient-poor wetlands (92 (3–445) kg CH_4–C ha^{-1} yr^{-1}).

4.4. Nitrous oxide fluxes

While N_2O fluxes were negligible, this result is consistent with a lack of artificial fertiliser use and supports other studies on extensive drained moorland (Skiba et al., 2013; Levy and Gray, 2015), abandoned peat meadow (Hendriks et al., 2007) and rewetted transition bog (Tauchnitz et al., 2015). N_2O emissions were probably limited by slow N transformation rates caused by limited nitrate availability from both relatively high WT levels and low nutrient availability (Martikainen et al., 1993) and possibly high N_2/N_2O ratios promoting complete denitrification. This is substantiated by the fact our study site is on the north-western fringe of Europe and records very low levels of N deposition (with average atmospheric ammonia concentration of less than 0.5 μg m^{-3}) (UCD, 2015). While not measured at our site, rewetting could have caused a shift in favour of dinitrogen (N_2) emissions; yet peaks of both N_2O and N_2 might have occurred during WT draw downs (Tauchnitz et al., 2015) and could have been missed by our sampling regime.

4.5. Net ecosystem carbon balance and global warming potential

At the shallow drained site, NECB was low when compared to other published results from grasslands on peat and reflects its nutrient-poor status. The lower NECB when ungrazed (117 g C m^{-2} yr^{-1} compared to 237 g C m^{-2} yr^{-1} when grazed) was solely attributed to the cessation of biomass export and the absence of livestock, while CO_2 emissions remained the same and were responsible for the majority of the GWP, especially during the no-grazing period. Despite reducing the climate footprint, abandonment of grassland ecosystems as a management action does not provide substantial net ecosystem C sink potential, at least at the early stages of succession, as seen in other abandoned grassland sites (Meyer et al., 2012; Berninger et al., 2015).

As expected the rewetted site had a lower NECB then the drained site and was either neutral or even negative when not grazed (−35 g C m^{-2} yr^{-1}). Therefore our findings confirm that rewetting organic soils under grassland leads to these ecosystems becoming neutral or small C sinks as per their natural counterparts (Blain et al., 2014). Fluvial C losses (dissolved organic carbon (DOC), particulate organic carbon (POC) and excess CO_2) were measured at the field level including both sites for Years 1 and 2 (Renou-Wilson et al., 2014). Thus, when a mean value of combined DOC, POC and excess CO_2 of 37.4 g C m^{-2} yr^{-1} is added the total NECB, the rewetted, ungrazed site would still remain neutral.

From a GWP point of view, the worst scenario ensues when the organic soils under grassland are rewetted and still grazed. CH_4 emissions are increased while GPP is suppressed by the continued grazing and therefore no positive climatic outcome was gained under this scenario. CH_4 emissions formed the major component of the GWP of the rewetted site and therefore rewetting should be accompanied by a change of management and sites should not be continuously grazed during the whole growing season.

Removing grazing altogether may however induce nefarious vegetation change and in the long term, species diversity may decline altogether or leading to a taxonomic homogenisation. Grazing can influence not only the vegetation and topsoil structure but indirectly, the competitive interaction between species (Moran et al., 2008). Higher species richness was recorded in the grazed rewetted site which contained higher forbs and wetlands species than the shallow drained sites. Species diversity increased with the moisture gradient but this relationship was annulled when grazing stopped. When left ungrazed for two consecutive years species diversity increased at all sites and the process of taxonomic homogenisation commenced (i.e., with similar species found in both sites), as seen in other abandoned wetlands (van Dijk et al., 2007; Koch and Jurasinski, 2015). Therefore, while the absence of management of wet grasslands may lead to the successful restoration of C sink function, long-term vegetation community changes may affect both biodiversity and the C sink function. In this study, species that dominated the ungrazed plots were typically tall, productive and competitive with the invasive presence of woody species, such as *Salix* already present in certain plots. However this feeds into recent concerns raised over the abandonment of wet grasslands at larger scales where reduced species richness was observed as competitors expand their coverage and woody plants encroach (Joyce, 2014). The future ecological trend of both sites in this study may depend on the WT level dynamics which, if they remain consistently high, may prevent the establishment of woody species and promote the development of wetland and moss species. Conversely, the site may not remain wet enough and woody species may start a process of drying both sites further with a subsequent negative impact on the C function of these grasslands. In other rewetted grasslands, both grazing or hay making were used to prevent the development of the meadows into woodland (Herbst et al., 2013). One cut a year or permitting livestock to graze for a short period during the summer months may be sufficient to promote species diversity while keeping CH_4 emissions low (Bhullar et al., 2014) and therefore remaining a climate friendly measure.

4.6. Climate mitigation and agri-environmental measures

Under the Kyoto Protocol process, parties are required to report on their actions to increase removals and decrease emissions of GHG from activities related to the Land Use and Land Use Change and Forestry (LULUCF) sectors. In addition, countries that elect to report under the "Wetland Drainage and Rewetting" activity (under Article 3.4 of the Kyoto Protocol) will also be able to claim C benefits from the rewetting of drained organic soils. Therefore, rewetting organic soils with a view to foster soil C sequestration should be encouraged whenever it is cheaper than the least expensive measure currently being used to meet climate policy targets.

In Ireland, the intensity of the management of organic soils has declined over time in marginal regions and a shift away from agricultural production and towards nature conservation may be possible in some cases. A high proportion of these permanent grasslands over organic soils are found in areas characterised by small farm size and difficult farming conditions. The 'wetter' grasslands are among the first to be neglected as farmers cease to use land associated with high costs, or associated with risk of spread of disease (e.g. Foot and Mouth). Currently, abandonment of these fields is not an option as single farm payments under the Common Agriculture Programme, requires that fields are maintained for agricultural production (e.g. cutting *Juncus* in 'wet' fields) in order to remain eligible for area-based subsidies. Paludiculture is an option but difficult to implement in rather hilly terrain found in this region but could be investigated in the larger, flat fields located in the Irish Midlands. Overall, synergetic benefits should accrue by greater targeting and tailoring the management practices through tying the climate mitigation measures into the more sophisticated management options supported by agri-environment schemes. Under Pillar 2 of the new CAP reform, agri-environment-climate measures may be an opportune instrument to use in this context as it aims to preserve and promote the necessary changes to agricultural practices that make a positive contribution to the environment and climate. While some studies have found that removing grazing on organic soils did not lead to effective restoration of some soil quality parameters, they may become more resilient to increased air temperatures (Pellegrino et al., 2015). Further long-term studies are needed to ascertain if a subsidy to maintain WT levels close to the surface in these grasslands would be efficient in contributing to both the environment (encompassing all ecosystem services) and climate agenda (Verhoeven and Setter, 2010).

5. Concluding remarks

Permanent grasslands represent the most common land use category for agricultural organic soils in Europe. Their potential to release stored C or indeed sequester C in the future can play an important role in climate change mitigation both in Europe and worldwide (Soussana et al., 2004; O'Mara, 2012). However, long-term measurement studies are essential to fully capture the influence of an ecosystem on the atmosphere (Beetz et al., 2013; Günther et al., 2014). This study was carried out to assess the full climatic footprint of shallow drained and rewetted grassland over organic soils and under various management options by providing total GHG balances and NECB budgets, as well as GWP for such land use category in the temperate zone. The 4-year study period intended to reduce the uncertainties associated with GHG balances of grassland on organic soils under various management options, focussing on vegetation composition and growth dynamics which are critical in rewetting situations (Blain et al., 2014). The main conclusions are: (1) CO_2 emissions are likely to be relatively low if grassland production over organic soils is continued through extensive grazing, while keeping the mean annual WT above −25 cm. (2) Removing grazing from shallow drained organic soils under grassland can reduce its overall climatic footprint by removing annual biomass exports and livestock emissions but NEE is not necessarily affected. (3) Rewetting and removing grazing in grassland over organic soils can be a climate mitigation tool and could even shift the ecosystem to a C sink (exerting a cooling effect on the climate) with careful management in the future. (4) The occurrence of aerenchymous plants, especially *Juncus effus*us and their management can strongly influence CH_4 emissions and the overall climatic impact of the ecosystem. (5) Finally shifting to a climate-smart land use could bring about synergetic benefits (ecosystem services) if integrated into agri-environmental schemes although long-term impacts are still unknown.

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