

1 **The impacts of drainage, nutrient status and management practice on the full**
2 **carbon balance of grasslands on organic soils in a maritime temperate zone**

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1 Abstract

2 Temperate grasslands on organic soils are diverse due to edaphic properties but also to
3 regional management practices and this heterogeneity is reflected in the wide range of
4 greenhouse gas (GHG) flux values reported in the literature. In Ireland, most grasslands
5 on organic soils were drained several decades ago and are managed as extensive
6 pastures with little or no fertilisation. This study describes a two-year study of the net
7 ecosystem carbon balance (NECB) of two such sites. We determined GHG fluxes and
8 waterborne carbon (C) emissions in a nutrient rich grassland and compared it with
9 values measured from two nutrient poor organic soils: a deep drained and a shallow
10 drained site. Carbon dioxide (CO₂), methane (CH₄) and nitrous oxide (N₂O) fluxes were
11 determined using the chamber technique, and fluvial C fluxes were estimated by
12 combining drainage water concentrations and flows.

13 The nutrient rich site was an annual source of CO₂ (233 g C m⁻² yr⁻¹), CH₄ neutral, and a
14 small source of N₂O (0.16 g N₂O-N m⁻² yr⁻¹). Net Ecosystem Exchange (NEE) at the
15 shallow drained nutrient poor site was -89 and -99 g C m⁻² yr⁻¹ in Years 1 and 2
16 respectively, and NEE at the deep drained nutrient poor site was 85 and -26 g C m⁻² yr⁻¹
17 respectively. Low CH₄ emissions (1.3 g C m⁻² yr⁻¹) were recorded at the shallow drained
18 nutrient poor site. Fluvial exports from the nutrient rich site totalled 69.8 g C m⁻² yr⁻¹ with
19 54% as dissolved organic C (DOC). Waterborne C losses from the nutrient poor site
20 reflected differences in annual runoff totalling 44 g C m⁻² yr⁻¹ in Year 1 and 30.8 g C m⁻²
21 yr⁻¹ in Year 2.

22 The NECB of the nutrient rich grassland was 663 g C m⁻² yr⁻¹ with biomass exports being
23 the major component accounting for 53%. The NECB of the nutrient poor deep drained
24 site was less than half of the nutrient rich site (2 year mean 267 g C m⁻² yr⁻¹). Although
25 NEE at the nutrient poor shallow drained site was negative in both years, high biomass
26 export meant it was a net C source (2 year mean NECB 103 g C m⁻² yr⁻¹). While the
27 impacts of the nutrient and drainage status on NEE, biomass exports and fluvial C
28 losses were confirmed, inter-regional differences in management practice and climate
29 were also significant factors which impacted on the overall NECB of these ecosystems.
30 Contrary to expectation, the NECB of nutrient poor drained organic soils under
31 grasslands is not necessarily a large C source and this has implications for Ireland's
32 choice of national GHG inventory reporting methodologies. This study can also aid the
33 development of strategies to deliver reduced emissions tailored to local grassland types.

34 Keywords: carbon balance, GHG, waterborne carbon, DOC, net ecosystem exchange,
35 methane, NECB, methane, nitrous oxide, peat soils

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1 Introduction

Organic soils are characterised by a high content of partially decomposed organic matter and are an important component of terrestrial carbon (C) storage (Gorham, 1991; Garnett et al., 2001). Drained peatlands and peat fires are responsible for almost one-quarter of C emissions from the land use sector with at least 2 Gigatonnes (Gt) of carbon dioxide (CO₂) emitted to the atmosphere globally each year (Parish et al., 2008; FAO, 2013a). In Europe, 20% of all CO₂ emissions between 1990 and 2010 originated from drained organic soils (FAO, 2013b).

In Ireland, organic soils contain an estimated 1-1.5 Gt of C, which represents between 62% and 75% of the total soil C pool (Tomlinson, 2005; Eaton et al., 2008; Renou-Wilson et al., 2011). However, this C store is under threat as centuries of peatland exploitation (peat extraction, agriculture and forestry), have left only c. 15% of peat soils in a natural state (Renou-Wilson et al., 2011). Carbon emissions (CO₂ and methane (CH₄)) from Irish peat soils and related activities are estimated to account for c. 3 Mt C per year (Wilson et al., 2013b), equivalent to the emissions reported from the transport sector (Duffy et al., 2013). Therefore, managing organic soils as a means of mitigating greenhouse gas emissions (GHG) from agricultural systems can be an effective strategy for lowering national emissions (Dawson and Smith, 2007; Smith et al., 2007). Of note, a new activity called 'rewetting and drainage' of organic soils has been included in the next post-Kyoto Protocol commitment period (2013-2017). New guidelines, together with default emission factors, have been developed for the 2013 Wetland Supplement of the Intergovernmental Panel on Climate Change (IPCC, 2014).

The use of organic soils for agriculture is a contentious land-use option when considering atmospheric impacts. In Ireland, agriculture is the largest contributor to overall national emissions and accounted for 32.1% in 2012 (Duffy et al., 2013), which is the greatest proportion among EU Member States (Eurostat, 2013). Grassland is the predominant land use in the temperate zone and covers 60% of the land area of Ireland (EPA, 2008). The reclamation of raw peat soils or cutover peat soils for grassland was a direct result of population pressures in the 19th century which intensified in the 20th century due to national drainage acts and agricultural schemes (Feehan et al., 2008). Presently, some 300,000 ha of organic soils is under grassland (with negligible area cultivated for crops), on par with organic soils drained for forestry and, together with industrial cutaway peatlands, managed peatlands represent nearly half of the total peat soil area or 10% of the total land area (Wilson et al., 2013b).

Drainage and grassland management practices (e.g. fertilisation) of organic soils stimulate the aerobic oxidation of previously accumulated organic matter, promoting CO₂ and nitrous oxide (N₂O) emissions while reducing methanogenesis (Byrne et al., 2004; Freibauer et al., 2004; Nieveen et al., 2005; Jacobs et al., 2007; Elsgaard et al., 2012; Drösler et al., 2013; Schrier-Uijl et al., 2013). However, inter-regional spatio-temporal variability of CO₂ flux is high, particularly in Western Europe. The implication is that physical and biogeochemical soil conditions (nutrient content, organic matter quality and water table levels) as well as local management practices and subtle climatic variations exert strong influences on C loss (Drösler et al., 2008). In addition, drainage of

1 temperate organic soils typically leads to higher DOC fluxes (Wallage et al., 2006;
2 Urbanová et al., 2011), so that management practices may alter both contemporary
3 photosynthetic uptake rates, but also the stability of older soil stores of sequestered C.
4 Further, although fluvial C exports are increasingly recognised as potentially significant
5 components of terrestrial C balances, full accounting of all fluvial C forms remains the
6 exception rather than the rule, notably for dissolved inorganic and particulate organic C.

7 The annual GHG emissions from organic soils under grassland are currently evaluated
8 in Ireland using the generic Tier 1 reporting method (IPCC, 2006). These emission
9 factors typically rely on assessing and averaging available data and so can be biased
10 towards locations with high research activities, and therefore individual regional climatic
11 and edaphic conditions where such research is conducted. For example, grasslands on
12 organic soils in Germany, where much research on grassland GHG balances is
13 conducted, are typically fertilised, deep drained and nutrient rich (Drösler et al., 2008;
14 Drösler et al., 2013), whereas in Ireland, extensively grazed permanent pasture with
15 partial drainage is common. The primary objective of this study is to estimate annual net
16 CO₂, CH₄ and N₂O fluxes as well as fluvial C fluxes from two typical Irish grasslands
17 over organic soil. We further combine estimates of terrestrial net ecosystem exchange
18 (NEE), fluvial C export, and CH₄ fluxes associated with grazing livestock, as well as
19 biomass C export, to calculate the net ecosystem C balance (NECB). By comparing two
20 grasslands with contrasting drainage, soil nutrient status and management practices, we
21 assess the role of these attributes in respect to biomass productivity, GHG and
22 waterborne C fluxes, and appraise the hypothesis that specific site characteristics (low
23 soil fertility and higher water table levels), management systems (low stocking density
24 and low inputs) and regional climate can affect the overall C balance of these
25 ecosystems. Finally, this study provides data to support a progression towards the IPCC
26 Tier 2 reporting level in Ireland by producing emission factors (EFs) for CO₂, CH₄, N₂O
27 and DOC for typical organic soils under grassland.

28

1 2 Material and methods

2 2.1 Study sites

3 The study sites were located in a maritime temperate climate zone in two distinct Irish
4 ecoregions, both characterised as landscapes rich in organic soils. Site A at Glenvar,
5 Co. Donegal (Latitude: 55° 9' N, Longitude: 7° 34' W) is situated at 40 m elevation in the
6 north-west of the country, less than 1km from the sea shore and the climate is typical of
7 western maritime Ireland with warm winters (very few ground frosts) but cool summers.
8 Site B in Lanesborough, Co. Longford (Latitude: 53° 39' N, Longitude: 7° 56' W) is also
9 low-lying, situated at 38m elevation but is typical of the midlands landscape, with late
10 spring frosts occurring as late as the beginning of June. Despite comparable long term
11 annual mean temperatures, lower minimum and higher maximum temperatures are
12 typically recorded at Site B. A minor gradient in precipitation is also present with higher
13 mean rainfall in Site A (Table 1). Potential evapo-transpiration in that region is estimated
14 at 32%-36% of annual precipitation (estimated long-term annual runoff of c. 700mm),
15 compared to 44% (annual runoff, c. 586mm) for Site B (Mills, 2000).

16 Land use in both regions is predominantly grassland and the two sites are managed for
17 low intensity beef suckler production. Site A has a lower stocking density of 0.6
18 Livestock Units (L.U.) per ha (all suckler cows) compared to Site B with 1.2 L.U. (sheep
19 in the winter months and suckler cows and calves in the summer months). While Site A
20 only receives on-site organic fertilisation from manure directly deposited by cattle in the
21 field, artificial fertiliser is usually applied in spring to Site B at a rate of 25kg N/ha.
22 However, no fertilisation took place at Site B during the first monitoring year. No cows
23 are present on-site during the winter months and no concentrates or additional feeding
24 occur on site.

25 Both sites A and B have been drained for over 60 years and the soils are categorised as
26 terric and limnic histosols respectively (FAO, 1998). With an organic matter content
27 higher than 30% (Table 2) over a depth greater than 40cm, they satisfy the definition of
28 peat soils in both the Irish soil classification (Hammond, 1981) and the FAO key to soil
29 types (FAO, 1998), which form the basis of definitions used in the IPCC guidelines
30 (IPCC, 2006, 2014). They are typical 'earthy peat', i.e. reclaimed drained peat soils
31 characterised by a well aerated structure and a relatively firm surface horizon containing
32 few or no recognisable plant remains. The peat in Site A overlies bedrock of
33 Precambrian schist and gneiss and contains on average 20% C (Table 2). The base
34 geology of Site B is of limestone origin and the overlying peat has a higher pH (5.5) and
35 almost double the C content (40%) and is situated over a thick layer of marl (pH 7.2).
36 Despite similar C:N ratios (16-21), sites A and B differ in nutrient status and, based on N,
37 phosphorus (P) and potassium (K) content, Site A is nutrient poor while Site B is nutrient
38 rich (Table 2).

39 Site B has a uniform hydrological soil profile throughout the experimental area. In
40 contrast, Site A can be divided into two distinct sub-sites: a deep drained area "Site A_d"
41 (defined in the 2006 IPCC guidelines as the mean annual water table depth of 30 cm

1 and deeper below the surface) and a shallow drained area “Site A_s” (defined as the
2 mean annual water table depth of less than 30 cm below the surface).

3

4 2.2 Vegetation

5 Both sites displayed a moderate to low quality sward dominated by *Holcus lanatus* and
6 *Agrostis spp.* Wetland species, such as *Equisetum palustre*, *Juncus effusus* and *Juncus*
7 *articulatus* were present in the shallow drained Site A_s while Site B was characterised by
8 a greater cover of *Rumex acetosa* and *Cirsium arvensis*. While the moss layer was
9 almost non-existent in Site B, Site A_d and A_s had an average ground cover of 20% and
10 60% respectively, composed solely of *Rhytidiadelphus squarrosus*. Experimental plots
11 were fenced off from cattle for the duration of the study. To simulate local activities, the
12 vegetation within and surrounding each collar for GHG measurements was cut to a
13 stubble height of 5 cm in order to mimic the cow grazing regime, so that the height of the
14 vegetation was always comparable to that in the grazed fields. The interval between
15 cuttings (considered as grazing events) was not strict but rather followed the
16 concomitant status of the vegetation in the fields. The number of cuttings also varied
17 each year reflecting the biomass production. The vegetation was collected from each
18 plot and oven-dried to a constant mass at 75 °C. C and N contents were measured
19 (CE440 Exeter Elemental Analyser) using a pooled, homogenised sample of all biomass
20 cut from each plot from the first growing season. Vegetation height (cm) was measured
21 regularly throughout the year and systematically during GHG measurements and before
22 cutting events. Conjunctly, leaf area index (LAI) was measured at each plot when
23 Photosynthetic Photon Flux Density (PPFD) levels were sufficient (November-March
24 excluded). The AccuPAR LP-80 (Ceptometer, Decagon Instruments, WA, USA) was
25 used to measure light interception via 80 independent photo sensors. The PPFD
26 transmission data was used to calculate gap fractions which were inverted to derive LAI
27 estimates (Norman and Campbell, 1989). Polynomial regression constrained to pass
28 through the origin was used to develop site-specific relationships between vegetation
29 height and LAI (Fig. 1) for use in Eq. 1 (see below).

30 2.3 Greenhouse gas sampling

31 Twelve permanent sample plots were established systematically within Site A (n=7 for
32 Site A_d, n=5 for Site A_s) and nine sample plots at Site B (at least 15 m from field border
33 or river). Each sample plot consisted of a stainless steel collar (60 x 60 cm) that was
34 inserted to a depth of 20 cm into the soil prior to the start of the study. Perforated PVC
35 pipes (internal diameter: 2 cm) were inserted adjacent to each sample plot to measure
36 water table position (WT). Wooden boardwalks were built around the sample plots to
37 minimise damage to the vegetation and to avoid compression of the peat during gas
38 sampling. Data loggers (Hobo External Data Loggers, Onset Computer Corporation, MA,
39 USA) were established at each study site and recorded hourly soil temperatures (°C) at
40 5, 10 and 20 cm depths. A weather station (Watch Dog Model 2400, Spectrum
41 Technologies Inc., IL, USA) was established at each site and recorded PPFD ($\mu\text{mol m}^{-2}$

1 s⁻¹), soil temperatures (°C) at 5 and 10 cm depths, hourly rainfall (mm) and volumetric
2 soil moisture at 5 cm (%; Site B only).

3 Site A was monitored starting April 2011 (first two years are reported here) while Site B
4 was only monitored for one year (April 2011 to March 2012) with no fertiliser application
5 taking place during that time. Additional N₂O measurements were taken at Site B in
6 April-June 2012 during an intensive campaign after a typical fertilisation event of 25kg
7 N/ha.

8

9 *CO₂ flux measurements*

10 CO₂ fluxes were measured from April 2011 to March 2013 (Site A) and from April 2011
11 to March 2012 (Site B) at fortnightly (summer) and monthly (winter) intervals using the
12 static chamber method (Alm et al., 2007). Measurements were done between 8 a.m. and
13 6 p.m. (2 to 9 measurements per collar per day). Instantaneous NEE was measured
14 over a range of PPFD using a transparent polycarbonate chamber (60 x 60 x 33 cm)
15 equipped with internal fans to ensure mixing of the air and a cooling system to maintain
16 the temperature within the chamber close to the ambient air temperature (see Alm et al.
17 2007). For each measurement, the chamber was placed in a water-filled channel at the
18 top of the collar and CO₂ concentration (ppmv) in the chamber headspace was
19 measured at 15-second intervals over a period of 60-180 seconds using a portable CO₂
20 analyser (EGM-4) (PP Systems, UK). PPFD was measured by a quantum sensor
21 located at the top of the chamber (PAR-1. PP Systems). Concurrently, air temperature
22 (°C) within the chamber and soil temperatures at 5, 10 and 20 cm depths were recorded
23 at each collar (soil temperature probe; ELE International, UK). The WT position relative
24 to the soil surface was manually measured with a water level probe (Eijkelkamp
25 Agrisearch Equipment, The Netherlands). Following each NEE measurement, the
26 chamber was vented for a short time by removing it from the collar. This was carried out
27 in order to ensure equilibration of the gas concentration. The chamber was then
28 replaced in the collar and covered with an opaque material in order to provide an
29 estimate of ecosystem respiration (R_{eco}), which is the sum of heterotrophic and
30 autotrophic respiration enclosed by the chamber. Flux rates (mg CO₂ m⁻² h⁻¹) were
31 calculated as the linear slope of the CO₂ concentration in the chamber headspace over
32 time, with respect to the chamber volume, collar area and air temperature. A flux was
33 accepted if the coefficient of determination (r^2) was at least 0.90. An exception was
34 made in cases where the flux was close to zero (mainly in winter time where soil / plant
35 processes are typically slower) and the r^2 is always low (Alm et al., 2007; Görres et al.,
36 2014). In these cases (~2 % of total fluxes) the flux data were examined graphically and
37 fluxes with obvious non-linearity (due to chamber leakage, fan malfunction etc.) were
38 discarded. The remainder were accepted provided that some of the environmental
39 variables measured at the same time (e.g. soil temperature, PPFD, LAI) were sufficiently
40 low to account for the low flux values (e.g. Wilson et al., 2013a). Gross primary
41 production (GPP) was calculated as the sum of NEE and R_{eco} values (Alm et al., 2007).

42

1 *CH₄ and N₂O flux measurements*

2 CH₄ and N₂O fluxes were measured at fortnightly/monthly intervals using the static
3 chamber method, which consisted of an opaque, polycarbonate chamber (60 x 60 x 25
4 cm) equipped with a battery-operated fan, which mixed the air within the chamber
5 headspace. Four 50 ml samples were withdrawn into 60 ml polypropylene syringes from
6 the chamber headspace at 10-minute intervals over a 40 minute period (first sample
7 taken 10 minutes after chamber closure) and then injected into pre-evacuated
8 Exetainers® vials (12ml Soda Glass Vials, Labco, UK). During each measurement, air
9 temperature inside the chamber, soil temperature (at 5, 10 and 20 cm depths) and WT
10 were recorded at each collar. Gas samples were sent to Justus-Liebig Universitat
11 Giessen, Germany for analysis. The samples were analysed for CO₂, CH₄ and N₂O
12 concentrations with a gas chromatograph (Bruker Greenhouse gas Analyzer 450-GC)
13 equipped with a Thermal Conductivity Detector (TCD), a Flame Ionisation Detector (FID)
14 and an Electron Capture Detector (ECD). Detector temperatures were 200 °C (TCD),
15 300 °C (FID) and 300 °C (ECD). The CO₂ (304, 402, 1509, 4000 and 20,005 ppm), CH₄
16 (1.02, 1.81, 5.02, 20.9 and 101.9 ppm) and N₂O standards (0.25, 0.32, 2.01, 15.1 and
17 100.1 ppm) standards were supplied by Deuste Steininger GmbH. Gas peaks were
18 integrated using Galaxie software (Varian Inc. 2006). Fluxes (mg CH₄ m⁻² h⁻¹ and µg
19 N₂O m⁻² h⁻¹) were calculated from the linear change in gas concentration as a function of
20 time, chamber volume, collar area and air temperature. A flux was accepted if the
21 coefficient of determination (r^2) was at least 0.90. The same criteria applied to CO₂ in
22 regard to low fluxes were also applied to CH₄ and N₂O fluxes. Approximately 32% of
23 CH₄ fluxes and 15% of N₂O fluxes were discarded for obvious non-linear changes over
24 the measurement period. Positive flux values indicated a loss of CH₄ and N₂O to the
25 atmosphere and negative flux values indicated CH₄ and N₂O uptake.

26

27 *CO₂ flux modelling*

28 Statistical and physiological response models (Alm et al., 2007) were constructed and
29 parameterised for each study site. Model evaluation was based on the following criteria;
30 (a) statistically significant model parameters ($p < 0.05$), (b) lowest possible standard error
31 of the model parameters and (c) highest possible coefficient of determination (adjusted
32 r^2) (see Laine et al., 2009). Given the homogenous dominant vegetation within each
33 study site, we combined data from all sample plots ($n=12$ at Site A and $n=9$ at Site B) to
34 construct a single model for gross photosynthesis (GPP) and ecosystem respiration
35 (R_{eco}) for each site. Model coefficients were estimated using the Levenberg-Marquardt
36 multiple non-linear regression technique (IBM SPSS Statistics for Windows, Version
37 21.0. Armonk, NY, USA). One-third of the data was randomly removed from all data
38 sets prior to modelling and used to independently test the models. During model
39 construction, the relationship between GPP or R_{eco} and a range of independent
40 environmental variables (recorded in conjunction with flux measurements) was tested.
41 Only variables that increased the explanatory power of the model were included. The
42 models were accepted if the residuals were evenly scattered around zero. In the GPP
43 models, we used PPFD and LAI as explanatory variables (Eq.1).

1

2

$$3 \quad GPP = P_{max} \left(\frac{PPFD}{PPFD + k_{PPFD}} \right) * \left[\frac{LAI}{(LAI + a)} \right] \quad \text{Eq. 1}$$

4

5 where P_{max} is maximum photosynthetic rates; $PPFD$ is photosynthetic photon flux
6 density; k_{PPFD} is the $PPFD$ value at which GPP reaches half its maximum (half saturation
7 constant); LAI is Leaf Area Index (see Section 2.2) and a is a model parameter

8

9 In the R_{eco} models, we used soil temperature at 5 cm depth (T_{5cm}) and either WT level
10 (Eq. 2) for Site A or VMC (Eq. 3) for Site B.

11

$$12 \quad R_{eco} = (a + (b * WT)) * \left[\exp(c * \left(\frac{1}{T_{REF} - T_0} - \frac{1}{T_{5cm} - T_0} \right)) \right] \quad \text{Eq. 2}$$

13

$$14 \quad R_{eco} = \frac{a}{(1 + \exp(-(VMC - b)/c))} * T_{5cm} \quad \text{Eq. 3}$$

15

16 where R_{eco} is ecosystem respiration; T_{REF} is reference temperature set at 283.15 K;
17 parameter T_0 is the (minimum) temperature at which respiration reaches zero and set
18 here at 227.13 K; VMC is volumetric moisture content; WT is water table depth; and a , b
19 and c are model parameters.

20 *Reconstruction of annual CO₂-C balance*

21 The response functions estimated for GPP and R_{eco} were used for the annual
22 reconstruction of NEE. In combination with an hourly time series of (1) $PPFD$, VMC (Site
23 B) and T_{5cm} , recorded by the weather station and data loggers, (2) modelled LAI and (3)
24 WT depths linearly interpolated from weekly measurements, GPP and R_{eco} fluxes were
25 reconstructed for each sample plot. NEE was then calculated on an hourly basis as
26 follows: $NEE = GPP - R_{eco}$ (Alm et al., 1997). Negative NEE values indicated a net
27 uptake of CO_2 from the atmosphere by the peatland and positive values indicated a net
28 loss of CO_2 to the atmosphere. The annual CO_2 -C balance ($g \text{ C m}^{-2} \text{ yr}^{-1}$) was calculated
29 for each sample plot by integrating the hourly NEE values over each 12-month period
30 (Year 1: April 1st 2011 to March 31st 2012 and Year 2: April 1st 2012 to March 31st 2013).
31 Uncertainty in the reconstructed annual GPP and R_{eco} was calculated by summing up
32 the maximum and minimum standard errors associated with each of the model
33 parameters (e.g. Drösler, 2005; Elsgaard et al., 2012). As NEE is not directly modelled,
34 uncertainty in the annual NEE estimate was calculated following the law of error
35 propagation as the square root of the sum of the squared standard errors of GPP and

1 R_{eco} . Modelled CO₂ fluxes, GPP and R_{eco} , as well environmental parameters were tested
2 using repeated measures GLM to show any effects of sites and year and combination of
3 site and year (IBM SPSS Statistics for Windows, Version 21.0. Armonk, NY. USA).

5 2.4 Waterborne C exports

6 Fluxes of DOC, particulate organic C (POC), and dissolved inorganic C (DIC) were
7 determined for each site for the hydrological year beginning October 2011. For Site A, a
8 second year of monitoring covered the hydrological year commencing October 2012.
9 Discharge measurement, laboratory analyses and C flux estimation methods employed
10 are briefly described below and detailed in Barry et al. (2014).

11 In order to accurately estimate waterborne C fluxes representative of the area on which
12 the terrestrial GHG fluxes were conducted, runoff measurements were confined to the
13 smallest catchment areas encompassing the GHG study sites. Field drains adjacent to
14 each of the study sites were selected and these drained relatively small areas of
15 between 3 and 5 ha. Critically, soils and land use within these catchments was uniform
16 and consistent with that of the GHG study sites. Although areas outside of the study
17 sites yielded water, this approach has advantages in that sampling is close to source,
18 limiting the residence time between exports from the soil environment and thereby
19 lessening the impact of processes that can alter aqueous fluxes (e.g. remineralisation
20 and atmospheric evasion of CO₂). Continuous level recorders (Orpheus mini, OTT
21 Germany) and flow gauging instrumentation were installed at each site. C concentrations
22 were determined at fortnightly to monthly intervals throughout each monitoring period.
23 DOC concentration was measured by high temperature Pt-catalysed combustion on
24 glass fibre filtered samples (Whatman GF/C; 1.7µm), POC concentrations by loss on
25 ignition of samples filtered onto glass fibre filters, and DIC was calculated according to
26 Stumm and Morgan (1996), from alkalinity (gran titration), and *in situ* measurements of
27 pH, temperature and specific conductance. Annual C loads were calculated as the
28 product of annual or seasonal flow-weighted mean concentrations and runoff volumes,
29 corresponding to 'method 5' given by Walling and Webb (1985), and DOC rainwater
30 inputs were subtracted based on concentrations determined for rainwater during the
31 study. Fluxes are expressed as areal rates according to catchment area. 95%
32 confidence limits for flux estimates were calculated according to Hope et al. (1997)

33 As DIC reflects both CO₂ derived from soil respiration and C derived from the
34 atmosphere and rock weathering, total DIC exports were apportioned to reflect terrestrial
35 organic matter dynamics alone (Stumm and Morgan, 1996; Telmer and Veizer, 1999).
36 All CO₂ above atmospheric equilibrium was considered derived from respiration of
37 organic matter. The origin of C derived from rock weathering present as dissolved
38 carbonates varies with respect to the C content of the source geology, such that 50% of
39 the C present as bicarbonate from the dissolution of carbonate rocks originates from the
40 rock, and 50% from carbonic acid (with soil respiratory / atmospheric origin). In contrast,
41 all C present as bicarbonate from the dissolution of feldspars and silicate rocks
42 originates from carbonic acid. Thus for Site B, which overlies limestone geology, half of
43 the bicarbonate export was attributed to soil respiratory C, whereas for site A, which

1 overlies Precambrian Gneiss and Schist, all bicarbonate was attributed to soil respiratory
2 C.

3 2.5 Calculation of NECB

4 A full net ecosystem carbon balance (NECB) can be derived from the terrestrial GHG
5 (CO₂ and CH₄) sources and sinks by adding fluvial C fluxes and C emissions associated
6 with biomass removal and the livestock grazing (enteric fermentation and on-site manure
7 deposits). The following nationally derived emission rates for CH₄ from enteric
8 fermentation were employed as detailed by O'Mara et al. (2007): 74 kg CH₄ head⁻¹ yr⁻¹
9 for suckler cows; 22.4 kg CH₄ head⁻¹ yr⁻¹ for calves; 8 kg CH₄ head⁻¹ yr⁻¹ for sheep.

10

11 3 Results

12 3.1 Weather conditions

13 In Year 1, Site A displayed higher precipitation (1211 mm) as well as a higher annual
14 mean air temperature (10.2°C) compared to Site B (1003 mm; 9.6°C) (Table 1). Rainfall
15 and temperature patterns were similar with both sites having higher mean annual
16 precipitation (7-12%) and air temperature (5-10%) compared to the 30-year average
17 values (Table 1). Both sites also had cooler summer but warmer autumn and winter
18 temperatures compared to the 30-year monthly averages. There was no significant
19 difference between monthly mean soil temperatures between all the sites ($p=0.801$) (Fig.
20 2). In Year 2, Site A again received higher precipitation than the long-term average (1193
21 mm) driven by high values during June and July, September and October, and
22 December and January. However, both the mean annual and monthly air temperatures
23 were consistently below the long-term averages (by 8-9%).

24 There was a typical seasonal trend in mean monthly PPFD values at both locations (Fig.
25 2) associated with day length and seasonal cloud cover, with the highest values
26 observed in summer (June to August) and lowest in the winter (December to February).
27 Annual and mean monthly PPFD values were significantly higher ($p<0.001$) at Site B
28 than Site A during Year 1. There was no significant difference between Year 1 and Year
29 2 at Site A but PPFD values were consistently higher in Year 2 during the period June-
30 December, except for July.

31 Seasonality of precipitation was reflected in WT depth at Site A and soil moisture content
32 at Site B (Fig 3). Despite rising slightly above -40 cm in winter, the monthly WT site
33 averages at Site A_d were relatively stable and similar annual means were recorded in
34 both years (-47.8 and -47.4 cm). Monthly means were significantly different between
35 Sites A_d and A_s ($p<0.001$), the latter being characterised by seasonal fluctuations with
36 monthly WT below -30 cm during late summer/early autumn periods, rising quickly to
37 levels above -10 cm during winter periods. Despite this, the annual WT mean at Site A_s
38 was similar between years at -22.9 cm in Year 1 and -24.2 cm in Year 2. At Site B, VMC
39 averaged 47.5% and ranged from 20 to 62% with two significantly drier periods in May
40 and August in Year 1.

1

2 3.2 CO₂ fluxes

3 The relationships between GPP and the environmental variables differed between the
4 study sites as demonstrated by the different model coefficients derived for equations (1),
5 (2) and (3), and given in Table 3. GPP is strongly dependent on irradiance (PPFD) and
6 is commonly described by a rectangular hyperbola. Seasonal variation in the
7 photosynthetic capacity of the vegetation, described by LAI, was incorporated into the
8 model in a manner similar to that described by Wilson et al. (2007b). A close relationship
9 between GPP and PPFD was observed at all sites ($r^2=0.53-0.59$) and the addition of the
10 LAI term further improved the explanatory power of the model for both Site A ($r^2 = 0.72$)
11 and Site B ($r^2 = 0.85$). Validation of the model showed a strong agreement between
12 modelled versus observed GPP fluxes (Fig. 4) and independent test data ($r^2 = 0.86$ at
13 both Sites A and B). However, higher variation was evident in the predicted GPP fluxes
14 at Site B, particularly at low to medium flux rates (1000 - 3000 mg CO₂ m⁻² h⁻¹). At Site
15 A, R_{eco} was driven by T_{5cm} and WT ($r^2 = 0.75$) and at Site B, R_{eco} was controlled by VMC
16 and T_{5cm}, and this combination explained 67% of the variation. As with GPP, the
17 relationship between observed and modelled R_{eco} was generally strong (Fig. 4). At Site
18 B, there was a tendency for the model to overestimate low flux rates and underestimate
19 higher flux rates ($r^2 = 0.63$ at Site A and $r^2 = 0.54$ at Site B using independent test data).
20 As expected, monthly estimates of GPP demonstrated clear seasonal variation with
21 maximum rates occurring during the month of July regardless of sites and year. In Year
22 1, monthly GPP values were consistently higher in Site B (Fig. 5) than Site A. This was
23 also reflected in the higher P_{max} parameter values in the GPP model (Table 3). Within
24 Site A, monthly GPP was higher at Site A_d than Site A_s in both years. Monthly R_{eco}
25 followed the same seasonal trend as GPP, which exceeded R_{eco} between April and
26 August in Year 1 at Site A_s and similarly at Site A_d except for the month of July. NEE
27 was positive (i.e. CO₂-C source) throughout the winter periods as a result of a greater
28 reduction of GPP compared to R_{eco}. Monthly CO₂ emissions (positive NEE values) were
29 much lower in Year 2 for 4 months (Oct-Jan) due to lower winter temperatures (Figs 2
30 and 5). In contrast R_{eco} exceeded GPP for 8 months of the year at site B, with CO₂
31 uptake only occurring in May through July. However, the highest monthly CO₂ uptake for
32 any plots (-128 g C m⁻² month⁻¹) were observed during these months (Fig. 5).

33 In Year 1, annual GPP was significantly greater ($p<0.001$) at Site B (2089 g CO₂-C m⁻²
34 yr⁻¹) compared to Sites A_d (1651 g CO₂-C m⁻² yr⁻¹) and A_s (1349 g CO₂-C m⁻² yr⁻¹) with
35 the latter being 35% lower than Site B (Table 4). In Year 2, annual GPP was 10% lower
36 at Site A_s and 5% lower at Site A_d presumably in response to lower temperatures and
37 irradiance in the spring and autumn. In Year 1, annual R_{eco} at Site B (2322 g C m⁻² yr⁻¹)
38 was almost twice that at Site A_s (1260 g C m⁻² yr⁻¹) with Site A_d intermediate (1736 g C
39 m⁻² yr⁻¹). In Year 2, similar trends to GPP were observed with a decrease in annual R_{eco}
40 of 12% at both Sites A_d and A_s (Table 4).

41 Annual NEE varied between sites and years (Table 4). In Year 1, Site A_d was a CO₂-C
42 source of 85 g CO₂-C m⁻² yr⁻¹ and Site A_s was a net sink (-89 g CO₂-C m⁻² yr⁻¹). However,
43 both sites were CO₂-C sinks in Year 2 (-26 and -99 g CO₂-C m⁻² yr⁻¹ respectively). In

1 contrast, Site B was a CO₂-C source of 233 g CO₂-C m⁻² yr⁻¹. The site effect was highly
2 significant (p<0.001) between all sites in Year 1. Similar analysis performed between
3 Site A_d and A_s for both years generated a significant site x year interaction (p<0.001)
4 demonstrating that CO₂ fluxes were significantly different between the sites and between
5 years in Site A.

6 3.3 Biomass production/export

7 Contrasting annual GPP values between sites and between years are associated with
8 biomass productivity. In Year 1, the cumulative production was superior at Site B (351 g
9 C m⁻²), with Site A_d and A_s 35% and 50% less productive respectively (Table 4).
10 Biomass production in Site B started to increase sooner than Site A and was
11 consistently higher, peaking during the warmest months (July-August) (data not shown).
12 Biomass production also lasted for a longer period at Site B allowing for biomass
13 removal to take place as late as October (8 cutting events in total). Productivity was
14 lower throughout Year 2 with poor growth during late summer months a particular
15 feature. Lower temperatures throughout the second growing season was likely the driver
16 behind the decrease in productivity with biomass production 18% lower at Site A_d (from
17 225 to 185 g C m⁻²) and 21% at Site A_s (from 174 to 137 g C m⁻²) (Table 4). Since
18 vegetation density (stems m⁻²) was similar between the shallow and deep drained plots,
19 an analysis of growth rates demonstrated that for both years, mean growth rates (over
20 periods between grazing events) were significantly lower in Site A_s compared to A_d
21 (p<0.001), except for the first two months of the study (Fig 7). The contrasting
22 productivity between all sites was also reflected in LAI values, which peaked before
23 cutting events during the summer with higher values at ~ 2 m² m⁻² for Site B compared
24 to ~1.5 m² m⁻² for Site A_d and ~ 0.8 m² m⁻² for A_s (data not shown). In Year 2, maximum
25 LAI at Site A_d was also consistently higher than Site A_s but the difference was reduced in
26 comparison to Year 1. A significant inverse relationship was found between mean
27 annual water table and mean LAI (growing season) during both years (Fig. 6).

28 The nitrogen content of the vegetation was similar at both Site A_d and A_s (2%) but higher
29 at Site B (3.3%). Nitrogen export in biomass followed C export trends with the nutrient
30 rich Site B losing the highest amount at 265 (27) kg N ha⁻¹ yr⁻¹ compared to 107 (45) and
31 80 (12) kg N ha⁻¹ yr⁻¹ in Site A_d and Site A_s (data are mean ± standard deviation with n = 7
32 for Site A_d and n = 5 for site A_s). Compared to Year 1, N biomass exports were 17 to 19%
33 lower in Year 2 in Site A_d and A_s respectively.

34 3.4 Methane fluxes

35 Fluxes of CH₄ were not detectable at the deep drained sites, namely Site A_d and Site B
36 and were relatively low at Site A_s (Fig. 8). A strong seasonal effect could be observed in
37 both years at Site A_s (Fig. 8). CH₄ fluxes followed a general trend where the highest
38 values were observed in late summer and the lowest in winter time. However, high flux
39 values for this site (0.9 and 1.4 mg CH₄ m⁻² h⁻¹) were also observed in April and
40 November. Spatial variation in emissions at Site A_s was very evident (Fig. 8) with the
41 higher emissions associated with plots that contained *Juncus effusus*. No statistical
42 relationship was observed between CH₄ fluxes and any of the environmental

1 parameters. Therefore, in order to calculate an annual CH₄-C balance, linear
2 interpolation between the observed fluxes was performed and the values integrated over
3 a 12 month period (Beetz et al., 2013). Annual CH₄-C emissions were not significantly
4 different between years with 1.3 (1.09) g CH₄-C m⁻² yr⁻¹ in Year 1 and 1.4 (1.1) g CH₄-C
5 m⁻² yr⁻¹ in Year 2 (Table 4).

6 7 3.5 Nitrous oxide fluxes

8 Fluxes of N₂O were not distinguishable from zero at Site A during the monitoring period.
9 At Site B, a seasonal trend was observable during Year 1 with the highest values (153
10 µg N₂O m⁻² h⁻¹) observed in October (Fig. 9). N₂O uptake (~14 µg N₂O m⁻² h⁻¹) was
11 observed during May. Following a fertilisation event at Site B in Year 2, N₂O emissions
12 increased considerably and reached an observed maximum of 484 µg N₂O m⁻² h⁻¹. As
13 with CH₄ fluxes, no statistical relationship was observed between N₂O fluxes and any of
14 the environmental parameters. Instead, linear interpolation between the observed fluxes
15 was performed and the values integrated over a 12 month period (Note: The integration
16 period was from April 1st 2011 to March 31st 2012 and, therefore, did not cover the
17 fertilisation experiment). Annual N₂O-N emissions for this period were estimated at 0.16
18 ± 0.34 g N₂O-N m⁻² yr⁻¹.

19 20 3.6 Waterborne C losses

21 Fluvial exports were measured at the field scale at Site A and so encompass both sites
22 A_d and A_s. In Years 1 and 2 the fluvial fluxes totalled 44 g C m⁻² yr⁻¹ and 30.8 g C m⁻² yr⁻¹
23 respectively. DOC export rates were 18.1 g C m⁻² yr⁻¹ in Year 1 and 4.9 g C m⁻² yr⁻¹ in
24 Year 2, representing 41% and 16% of the total C flux in each year respectively (Table 4).
25 The mean drainage water pCO₂ in Years 1 and 2 respectively was 102 and 146 times in
26 excess of atmospheric equilibrium concentrations and consequently the export of excess
27 CO₂ (above equilibrium) was substantial at 13.4 and 18.5 g C m⁻² yr⁻¹ respectively.
28 Excess CO₂ represented 30 and 60% of the waterborne C flux each year and dominated
29 the export of DIC (excess CO₂ + HCO₃⁻) which overall comprised 44% and 78% of the
30 total fluvial C flux in each year. As with DOC, POC fluxes were also lower in Year 2,
31 declining from 6.4 g C m⁻² yr⁻¹ to 2.0 g C m⁻² yr⁻¹.

32 The fluvial C flux was greater at Site B at 68.9 g C m⁻² yr⁻¹, comprising 54% DOC, 35%
33 DIC and 11% POC. By comparison to Site A, this waterborne C flux largely reflected a
34 greater export of DOC, particularly relative to the second year at site A. Despite
35 differences in base geology between sites, the flux of DIC (estimated as derived from
36 soil respiratory processes) was comparable to both years at site A, but differed in that it
37 was primarily in the form of bicarbonate (29% of total flux). The implication is that soil
38 respiratory CO₂ at this site yielded greater carbonate dissolution rather than
39 accumulating in soil pore waters to high pCO₂.

40

1 3.7 NECB

2 Combining the nationally-derived CH₄ emission rates from enteric fermentation with
3 contemporary stocking rates at the sites yielded further emissions of 3.6 g C m⁻² yr⁻¹ for
4 Site A and 9.5 g C m⁻² yr⁻¹ for Site B. Given the very low intensity farming systems at
5 both sites, we assume the balance of C import/export from on-site manure deposits
6 negligible (there was no fertilisation at Site B in the first year). The NECB of the shallow
7 drained nutrient poor grassland site was a small source in both years (134 (75) and 73
8 (56) g C m⁻² yr⁻¹) while the deep drained nutrient poor site was double with 358 (111)
9 and 193 (64) g C m⁻² yr⁻¹. The NECB of the nutrient rich drained site was 663 (42) g C m⁻²
10 yr⁻¹ (Table 4).

11

12 4. Discussion

13 The large amount of C stored in organic soils means that careful management of these
14 soils should be a priority to reduce C losses. This should start with more accurate
15 estimates of GHG fluxes and waterborne C losses that reflect regional variability
16 (climatic and edaphic) but also local management practices. In this study, we have
17 investigated contrasting permanent grasslands over organic soils in a maritime
18 temperate climate to evaluate site effects and management practices on the GHG fluxes
19 and waterborne C losses.

20 4.1 Contrasting NEE and driving parameters

21 Annual NEE differed significantly among site with the nutrient poor shallow drained site
22 (A_s) a net annual sink of CO₂-C in both years of the study and the nutrient poor deep
23 drained site (A_d) a net sink of CO₂-C in the second year. In contrast, the nutrient rich
24 drained Site B was a net annual source of CO₂-C. Given that there was a tendency for
25 the R_{eco} model to underestimate the higher fluxes at this site (Fig. 4), this NEE value
26 may be a conservative estimate as it is at the lower range of data reviewed from
27 temperate and boreal studies (e.g. Maljanen et al., 2010; Elsgaard et al., 2012; Leiber-
28 Sauheiti et al., 2013). Annual losses of CO₂ from our nutrient rich drained site B were
29 more in line with those determined for low-intensity Dutch permanent grasslands on
30 organic soils (Jacobs et al., 2007; Veenendaal et al., 2007) and for extensive wet
31 nutrient poor grasslands in Germany (Drösler et al., 2013). This corroborates the fact
32 that Site B is a low-intensity grassland (no cultivation and low inputs). The importance of
33 a “grassland type” appears even more apparent in the case of nutrient poor sites as
34 investigated in this study. Net annual uptake of CO₂-C in both the deep drained and
35 shallow drained nutrient poor sites under investigation here contrast significantly with
36 studies from similar site types in temperate climatic regions where a net annual loss of
37 CO₂-C has been reported (Drösler et al., 2013; Leiber-Sauheiti et al., 2013). The NEE
38 values for the nutrient poor shallow drained sites in this study represent similar sinks
39 (small source in some years) to those reported by Skiba et al. (2013) for grazed drained
40 nutrient poor organic soils (moorland) in a maritime temperate climate. Our results would
41 therefore point to the significance of edaphic conditions as well as the low intensity
42 management system encountered at our sites. An indication of such influences can be

1 found in the contrasting GPP and R_{eco} parameters found at all sites and which are
2 discussed below.

3 *Productivity drivers*

4 The nutrient rich grassland was the most productive with an annual GPP value in the
5 range of those reported from Irish grasslands (Byrne et al., 2005; Gilmanov et al., 2007)
6 and comparable to Danish grassland over organic soils (2100 to 2500 g C m⁻² yr⁻¹)
7 (Elsgaard et al., 2012) and to other maritime intensive grasslands over peat (2239 –
8 2403 g C m⁻² yr⁻¹) in north Germany (Beetz et al., 2013). Despite GPP being significantly
9 lower in the nutrient poor sites, the values were remarkably high and more productive
10 than similar grasslands over peat in Germany (Beetz et al., 2013; Drösler et al., 2013;
11 Leiber-Sauheitl et al., 2013). This may be a reflection of the longer growing season due
12 to mild winter temperatures experienced in a maritime temperate climate like Ireland
13 (Wilson et al., 2007b) which is influenced by the North Atlantic Drift. Increased soil
14 temperatures over the last three decades also indicate that the growing season is
15 getting longer in Ireland (Hurtado-Uria et al., 2014). Moreover, this temperate humid
16 climate is not associated with very high temperatures that limit productivity during the
17 growing season via water shortage as experienced in other temperate regions. GPP was
18 closely linked to PPFD at all studied sites and the significantly higher PPFD received by
19 the plants at Site B (located further south than Site A) may explain some of the higher
20 productivity there, especially given that irradiance was highest at the end of the growing
21 season when it is one of the most dominant growth-limiting factors as soil temperature is
22 still high (Fig. 2). However, it cannot explain the contrasting GPP between Sites A_d and
23 A_s where plants received the same PPFD. Gross photosynthesis depends on vegetation
24 status, commonly estimated by proxies, such as LAI, which captures well the variations
25 in vegetation growth that occur both seasonally and as a result of grazing events (e.g.
26 Fig. 1). Our CO₂ measurement campaigns covered the full range of LAI and vegetation
27 heights, which were always highest in the nutrient rich site. In comparing the two
28 nutrient-poor sites, a significant inverse relationship was found between mean annual
29 water table and mean LAI (growing season) during both years (Fig. 6). Higher water
30 table levels most likely reduced biomass productivity by slowing the rate of gas diffusion
31 through the vegetation (Lohila, 2008), effectively inhibiting photosynthesis (decreased
32 GPP). The lower growth rates observed in Site A_s could also be explained by the
33 presence of wetland plant species in some of the plots, such as *Juncus spp.* and
34 *Equisetum spp.* which have cylindrical leaves and thus affect leaf area estimations
35 (Chen and Black, 1992; Wilson et al., 2007a).

36 As such, local climate and the nutrient and drainage status have been identified as the
37 most important factors explaining the contrasting biomass productivity and consequent
38 GPP at our sites and this was also reflected in the biomass exports. While the number of
39 'grazing' events were similar at both Sites A_d and A_s (5 in Year 1 and 4 in Year 2), the
40 higher productivity in the deep drained site was reflected in larger C biomass exports
41 (30% more in Year 1 and 35% more in Year 2 compared to site A_s). N biomass exports
42 were also on average 36% higher in Site A_d over the two years. By comparison, eight
43 'grazing' events were possible in the nutrient rich Site B. Management practice in the

1 form of grazing regime would therefore have a greater impact on GPP than at the other
2 sites.

3 *Respiration drivers*

4 Modelled annual R_{eco} was significantly different among all sites and was highest at the
5 nutrient rich site (B) with an estimated mean value of $2322 \text{ g C m}^{-2} \text{ yr}^{-1}$, which is similar
6 to values measured for grasslands over organic soils in Europe (Gilmanov et al., 2007;
7 Elsgaard et al., 2012; Beetz et al., 2013). R_{eco} significantly decreased with increasing soil
8 moisture and decreasing soil temperatures in accordance with other studies on
9 managed organic soils (Teh et al., 2011; Elsgaard et al., 2012). Soil temperatures were
10 not significantly different among all sites and therefore did not explain the differences in
11 R_{eco} . VMC varied widely at the nutrient rich site and wet-dry cycles can enhance peat
12 degradation and therefore increase soil respiration (Aerts and Ludwig, 1997; Chow et
13 al., 2006) as was also reflected in the higher DOC fluxes measured at that site.
14 Furthermore, as high GPP would also result in high autotrophic respiration rates, the
15 sustained high respiration rates at Site B, especially at the end of the growing season,
16 might be the result of enhanced levels of heterotrophic soil respiration in the
17 rhizosphere, known as the 'priming effect' (Kuzyakov, 2006) driven by the addition of
18 labile C in the form of root exudates and litter.

19 The two well-drained organic soils (nutrient poor Site A_d and nutrient rich Site B)
20 contrasted in that, while Site A_d was less productive (20% lower) due mainly to a lower
21 nutrient content, it also displayed reduced R_{eco} sustained through the year (30% lower
22 on average). Lower rates of autotrophic respiration due to lower productivity could only
23 explain the difference during the growing season. Given that they display similar soil
24 temperature profiles, other edaphic characteristics may be at play. When peat soils are
25 drained, organic matter can be rapidly oxidised by aerobic metabolic processes.
26 However, rates of aerobic decay vary widely across drained organic soil types even
27 under conditions of similar climate, water table and management (Byrne et al., 2004;
28 Couwenberg, 2011). The lower organic matter (OM) content (30% and 70% at Site A_d
29 and B respectively) may have influenced respiration rates although Leiber-Sauheitl et al.
30 (2013) found that CO_2 emissions were not affected by lower soil organic carbon content
31 in the surface horizon of cultivated histic soils. Leifeld et al. (2012) demonstrated that the
32 quality of the OM at the surface of organic soils that have been drained for a long time is
33 poor (e.g. lower polysaccharides content) and is less sensitive to C loss via
34 heterotrophic respiration due to preferential metabolism of labile organic matter fractions.
35 While OM quality was not specifically examined in this study, the soil C:N ratios of the
36 deep drained sites were very similar (Table 1) suggesting similar OM quality. However,
37 the depth of OM already exposed to oxygen would have been greater over time at Site
38 A_d as this site has been drained for >100 years so that the recalcitrant OM fraction may
39 be greater compared to Site B. Heterotrophic metabolism can also be limited by nutrient
40 availability, so that despite the similar site characteristics and soil C:N, the greater soil
41 nutrients and higher pH at Site B may explain the greater respiration at this site (Table
42 1). Indeed the elevated respiration rates are consistent with the greater soil nitrate
43 concentrations at this site, which may partly reflect elevated nitrification rates but also
44 that this site receives low inorganic fertiliser inputs. It should be noted that while marl is

1 present under the peat at Site B, the contribution of the abiotic CaCO_3 -derived CO_2
2 effluxes to the annual soil CO_2 fluxes, is deemed marginal (Kuzyakov, 2006).

3 In this study, respiration rates contrasted between the two nutrient poor sites where
4 higher water table levels led to decreased respiration rates, which have been observed
5 elsewhere (Leiber-Sauheitl et al., 2013; Skiba et al., 2013). The linear response of R_{eco}
6 to water table depth as modelled by Eq. (2) and the good fit between modelled and
7 observed data for the derived models (Fig. 4) indicated that R_{eco} was not restricted by
8 very dry hydrological conditions. This is due to the relatively constant high moisture
9 content (Irish soils are rarely subject to moisture deficit) linked to a high precipitation
10 regime with the number of wet days well distributed throughout the year (Xu et al., 2004;
11 Lawton et al., 2006). Soil temperature also exerted a strong influence on R_{eco} , as has
12 been found in other studies and this explained the inter-annual variation of annual R_{eco}
13 at this site. Respiration rates were reduced by 12% at both sites A_s and A_d in Year 2
14 which had similar water table levels but colder soil temperatures (at 5 cm, Fig. 2) than
15 Year 1. Interestingly, productivity was less affected by the cooler temperatures,
16 suggesting soil metabolism was more affected by this climatic feature than
17 photosynthetic activity. Similarly, higher water table levels affected R_{eco} more strongly
18 than GPP (through LAI) with 27% lower annual R_{eco} in the shallow drained site in both
19 years compared to the deep drained site. Taken together our results demonstrate that
20 low respiration rates controlled by higher water table levels and colder soil temperatures
21 were the main drivers of NEE at the nutrient poor sites. Thereby, while the CO_2 -C sink
22 potential of these sites may be threatened by climate change, it may be safeguarded by
23 ensuring a shallow drainage status (water table remaining above -25 cm).

24

25 4.2 Magnitude of CH_4 and N_2O fluxes

26 Uncertainty associated with both CH_4 and N_2O fluxes stems from source variation
27 (temporal and spatial) as well as limitations in measurement and methodology. As CH_4
28 production is expected to occur mainly within the anoxic soil horizons, i.e. below the
29 groundwater table, the absence of CH_4 emissions at Sites B and A_d are consistent firstly
30 with annual mean water table depths (below -30 cm) and secondly with the absence of
31 aerenchymous plant species that can transport CH_4 from the soil to the atmosphere
32 (Couwenberg, 2009). Fluxes close to zero were also recorded from drained grasslands
33 over peat in Germany (Drösler et al., 2013; Leiber-Sauheitl et al., 2013) and Denmark
34 (Petersen et al., 2012).

35 The estimated mean annual CH_4 fluxes for the shallow drained nutrient poor Site A_s (18
36 $\pm 15 \text{ kg CH}_4 \text{ ha}^{-1} \text{ yr}^{-1}$ over 2 years) are at the lower range of values reported for wet
37 grasslands in Germany (Leiber-Sauheitl et al., 2013) and are only 10% of values
38 reported for managed fen meadow and extensively managed grassland on peat in the
39 Netherlands (Kroon et al., 2010; Schrier-Uijl et al., 2010). Our sampling frequency may
40 have under-estimated annual emissions as some fluxes might have been missed during
41 wetter periods. However, measurements for CH_4 were carried out at all sites over a wide
42 range of water table levels or VMC. The absence of CH_4 fluxes recorded during the

1 wetter periods in Site B in particular may be due to several reasons. Since net CH₄
2 fluxes are measured, CH₄ oxidation and consumption may be taking place within the
3 uppermost well-aerated horizon. Suppressed methanogenesis may also occur in the
4 deeper layers due to the abundance of electron acceptors other than oxygen, such as
5 nitrate (Table 2). The low or even absence of CH₄ fluxes may also be due to the
6 relatively shallow peat depth (<1 m) as suggested by Levy et al. (2012) and the fact that
7 these sites have been drained for a very long time (Flasse et al., 1998). Poor quality and
8 supply of substrates may also restrict methanogenesis (Couwenberg, 2009). Given that
9 a recent meta-analysis has demonstrated that vegetation is the strongest explanatory
10 parameter for CH₄ flux variability (Levy et al., 2012), it is likely that the higher CH₄ fluxes
11 measured from the shallow-drained nutrient poor sites were also a product of the
12 presence of *Juncus effusus* (aerenchymous species) as seen in other studies (Petersen
13 et al., 2012; Herbst et al., 2013). Overall, our results demonstrate that CH₄ fluxes were
14 negligible, if not absent components of the C balance of extensively drained grasslands
15 over organic soils in a maritime temperate climate and that CH₄ was emitted only in very
16 small amounts when the mean annual water table was around – 23 cm.

17 The absence of N₂O emissions from the nutrient poor sites was consistent with the lack
18 of artificial fertiliser use for several decades at these sites and similar results have been
19 reported from an extensive drained moorland in Scotland (Skiba et al., 2013). In nutrient
20 rich sites, the potential for N₂O emissions is much higher, and is additionally forced by
21 fertiliser applications to these more productive systems. Estimated annual N₂O
22 emissions from Site B are nonetheless still low by comparison with most grasslands
23 situated over peat (closer to the IPCC Tier 1 default values for shallow drained nutrient
24 organic soils). This study provides further evidence that N₂O fluxes are predominantly
25 affected by the grassland management type, which is not represented in the categories
26 with the IPCC guidelines (2014). Most investigations are carried out in intensively
27 managed grasslands. While being nutrient rich, Site B is still very much 'extensive' given
28 the low artificial N input. Our values are, however, comparable to Dutch figures reported
29 by Langeveld et al. (1997) and van Beek et al. (2011) where grasslands had typically
30 less fertiliser and manure inputs over time. Overall, our results are not sufficient to
31 conclude firmly on the insignificance of N₂O fluxes in extensive grassland over organic
32 soils and further investigations in this area are warranted to ascertain the potential for
33 large denitrification losses. Since N₂O uptakes were detected on occasion (Fig. 9), N₂O
34 reductase activity could be high and therefore large dinitrogen (N₂) emissions may be
35 possible. This study also highlights the need for further research on intensive
36 measurement of N₂O based on C:N ratios, which may not reflect soil organic C
37 degradation as seen in the contrasting ecosystem respiration measurements from our
38 two sites with similar C:N ratios.

39 4.3 Fluvial C losses

40 Studies have shown that fluvial C losses can be large components of terrestrial C
41 balances at field and landscape scales, particularly where organic soils predominate
42 (Worrall et al., 2009; Dinsmore et al., 2010; Buffam et al., 2011). Furthermore, the
43 importance of this component can be magnified by management practices given that
44 large increases in DOC export (51 to 118%) can occur following drainage of such soils,

1 related to greater DOC production under aerobic conditions and soil redox conditions
2 during wet and dry periods (Wallage et al., 2006; Clark et al., 2009; Urbanová et al.,
3 2011). In this study, DOC fluxes were greatest for the drained nutrient rich site but lower
4 than values reported for drained peatland catchments in the same climatic region
5 (Gibson et al., 2009; Rowson et al., 2010). More significantly, the 2-year mean DOC flux
6 from the nutrient poor site ($11.5 \text{ g C m}^{-2} \text{ yr}^{-1}$) is at the low end of values reported for
7 temperate semi-natural peatland catchments in Ireland and the UK (Dawson et al., 2004;
8 Dinsmore et al., 2010; Koehler et al., 2011; Kiely et al., 2014). This suggests that DOC
9 fluxes from nutrient poor organic soils under extensive grasslands in Ireland may be
10 more similar to semi-natural ecosystems. Annual DOC fluxes from the nutrient poor site
11 were almost 4 times lower in Year 2 reaching a low value of $4.9 \text{ g C m}^{-2} \text{ yr}^{-1}$. Inter-annual
12 variability in DOC flux at this site is likely due to reduced precipitation and greater evapo-
13 transpiration during the second hydrological year (runoff 897 mm and 745 mm in Years 1
14 and 2 respectively). However, the decline in DOC export from Year 1 to 2 was partially
15 offset by a greater flux of aqueous CO_2 export, implying that drier conditions were more
16 conducive to soil OM and soil pore water OM mineralisation. In both years, the flux of
17 DIC (derived from soil respiration) exceeded the flux of DOC, and was predominantly in
18 the form of CO_2 . The high $p\text{CO}_2$ of samples implies that accurately accounting for
19 aqueous CO_2 fluxes requires that samples are taken close to source to account for C
20 that is otherwise rapidly evaded to the atmosphere, particularly with waters with low
21 alkalinity. While the literature on aqueous inorganic C fluxes from agricultural
22 catchments is not extensive, our observations are consistent with findings for small
23 boreal streams where exports of CO_2 were also substantial and atmospheric evasion of
24 CO_2 from streams was equivalent to approximately half of the total stream C flux (Wallin
25 et al., 2013). Our findings also provide support for arguments for greater incorporation of
26 stream C fluxes to improve regional and global C accounting (e.g. Downing et al., 2012).
27 At Site B the DIC flux was also considerable, but comprised a lower fraction of the total.
28 Notably this DIC export was predominately in the form of bicarbonate so that the soil
29 respiratory origins of much of this fraction are indicated to behave more conservatively
30 during downstream transport but nevertheless remains an important term in the
31 terrestrial C balance. These results support an argument for the requirement to report
32 other fluvial C components in GHG inventories, notably DIC and POC (e.g. Hope et al.,
33 2001; Dinsmore et al., 2010).

34

35 4.4 Complete NECB

36 Concurrent GHG and fluvial C balance studies on organic soils have been undertaken
37 on only a few sites worldwide and this is the first full NECB of a temperate drained
38 organic soil under grassland. At the nutrient rich site, NECB was relatively high ($663 \pm$
39 $42 \text{ g C m}^{-2} \text{ yr}^{-1}$) considering that the grassland is extensive and this value is greater than
40 those reported for intensive grassland on peat in both Germany (Beetz et al., 2013) and
41 the Netherlands (Veenendaal et al., 2007). However, the value falls within the range of
42 annual emission rates of $410\text{-}760 \text{ g C m}^{-2} \text{ yr}^{-1}$ reported by Couwenberg (2011), who
43 reviewed several temperate drained peat soils under grassland. It should be noted,
44 however, that the values above do not include fluvial C losses, which in this study

1 accounted for 11% of the total NECB for the nutrient rich site. While annual biomass
2 exports were at the low end of literature values e.g. 317-515 g C m⁻² yr⁻¹ (Veenendaal et
3 al., 2007; Beetz et al., 2013), they accounted for 53% of the NECB (Table 4). The two-
4 year mean NECB of the deep and shallow drained nutrient poor sites were considerably
5 smaller than the nutrient rich site (40% and 16% respectively). Biomass exports were
6 also the biggest components of the NECB at these sites (Table 4) and were sufficient to
7 influence the source-sink balance of the shallow drained site in both years and at the
8 deep drained site in Year 2 (i.e. shift from a NEE sink to a net source of C). Given the
9 impact of the biomass removal on NECB, controls on grazing regimes could therefore
10 significantly influence the C sink capacity of these ecosystems, as documented for an
11 abandoned former pasture over peat in the Netherlands (Hendricks et al., 2007). The
12 NECB of the shallow drained nutrient poor site is indeed consistent with values reported
13 for an extensive grassland in Germany, which was rewetted four years previously (Beetz
14 et al., 2013). However, in the case of nutrient poor organic soils, fluvial C losses were
15 not a negligible component of the overall NECB and could easily influence the final C
16 balance of these ecosystems.

17

18 4.5 Implications for reporting and climate change mitigation strategies

19 CO₂, CH₄, N₂O and DOC emission factors

20 NEE added to the C losses from the removal of biomass (assuming instantaneous
21 emission of cut biomass) and averaged over a multi-annual measuring period, gives a
22 best-possible estimate of emissions/removals and therefore the most accurate CO₂-C
23 emission factor (EF) (Couwenberg, 2011; Elsgaard et al., 2012). The EF for the nutrient
24 rich deep drained grassland (Site B) was 5.84 t CO₂-C ha⁻¹ yr⁻¹ which is in the range of
25 values given in the IPCC guidance (2013): 6.1 (5.0-7.3) t CO₂-C ha⁻¹ yr⁻¹ for this land use
26 category. It corresponds well with Tier 2 country-specific EFs for permanent grassland
27 over organic soils: 5 t CO₂-C ha⁻¹ yr⁻¹ in Germany (Federal Environmental Agency,
28 2013), 5.19 t CO₂-C ha⁻¹ yr⁻¹ in the Netherlands (Netherlands Environmental
29 Assessment Agency, 2009) and 5.17 t CO₂-C ha⁻¹ yr⁻¹ in Denmark (National
30 Environmental Research Institute, 2013).

31 In contrast, our nutrient poor sites produced much lower CO₂ EFs of 2.35 t CO₂-C ha⁻¹
32 yr⁻¹ in the deep drained and 0.62 t CO₂-C ha⁻¹ yr⁻¹ in the shallow drained. These figures
33 are outside the lower range of EFs for temperate drained nutrient poor sites in the IPCC
34 guidance (2014): 5.3 (3.7-6.9) t CO₂-C ha⁻¹ yr⁻¹. This default value is not further stratified
35 between shallow and deep drained grasslands and is based on only seven identified
36 study sites, six of which are from Germany (Drösler et al., 2013). Nonetheless, the multi-
37 site GHG investigation reported by Drösler et al. (2013) includes nutrient poor sites
38 (extensive grasslands), which were a CO₂-C sink (NEE minus biomass removal) in some
39 years. In Denmark, an EF of 1.25 t CO₂-C ha⁻¹ yr⁻¹ is applied to non-fertilised permanent
40 grassland. Along with the results presented here, this supports our premise highlighting
41 considerable heterogeneity in the C balance of grasslands on organic soils within the
42 European temperate zone. While the stratification of default CO₂ EFs according to

1 nutrient and drainage status is indeed supported by this study, variability in the impact of
2 these factors also reflects the historical and contemporary management practices
3 therefore advocating a progression towards the Tier 2 reporting level for countries with
4 significant areas of organic soils under grassland.

5 The EF for CH₄ at both the deep drained sites, regardless of nutrient status was zero
6 and for the shallow drained nutrient poor grassland was calculated at 18 kg CH₄ ha⁻¹ yr⁻¹.
7 The absence of CH₄ fluxes from both our well-drained organic soils is at variance with
8 the IPCC guidance which presents an EF of 16 kg CH₄ ha⁻¹ yr⁻¹ for nutrient rich and 1.8
9 kg CH₄ ha⁻¹ yr⁻¹ for nutrient poor. This over-estimation could be compensated by using
10 the EF for nutrient poor drained organic soils in the case of shallow drained nutrient poor
11 sites. If water table status is used as a primary factor, the only default EF provided is for
12 shallow drained nutrient rich at 39 kg CH₄ ha⁻¹ yr⁻¹ which would also lead to a large over-
13 estimation of CH₄ fluxes.

14 The EF for N₂O of zero at the nutrient poor site and 1.6 kg N ha⁻¹ yr⁻¹ at the nutrient rich
15 site are both very low compared to the default EF in the IPCC guidance (2014) however
16 our value is likely to be conservative as the fertilisation events were not included in this
17 figure. Despite being subject to high uncertainty, our results suggest again the likely
18 significance of management system, especially with regards to fertilisation and stocking
19 regime, factors which are currently missing from Tier 1 methodology.

20 In order to calculate an EF for DOC, it is critical to account only for the portion of DOC
21 that could be completely mineralised and re-emitted to the atmosphere. While lake
22 sediments can be sites for significant permanent burial of terrestrially fixed C, in most
23 river-lake catchments mineralisation far exceeds burial (Algesten et al., 2003), and
24 indeed studies have shown that most of the DOC flux from drained peat catchments can
25 be mineralised (Kölher et al., 2002; Jones et al., 2013). Despite such observations, the
26 oceans still receive considerable inputs of terrestrial organic matter (c. 0.4 Gt yr⁻¹
27 according to Richey, 2004) although the very minor amounts of unambiguously
28 terrestrial material identified in seawater and marine sediments imply that the vast
29 majority is re-mineralised (Bauer and Bianchi, 2011). As per the updated IPCC
30 guidance, a 90% fraction was used to calculate an overall EF for DOC of 22.1 t C ha⁻¹ yr⁻¹,
31 being much lower at the nutrient poor site (10.3 t C ha⁻¹ yr⁻¹) compared to the nutrient
32 rich site (33.9 t C ha⁻¹ yr⁻¹). The later value is within the range of the default EF for
33 temperate drained organic soils: 31 (19-46) g C m⁻² yr⁻¹ (IPCC, 2014). Given the high
34 inter-annual variability experienced at the nutrient poor site, further long-term study
35 would be warranted in order to support the development of a country-specific EF which
36 would include a 'nutrient status' stratification.

37 Overall, the current default EF may not be representative of the variety of grasslands
38 over organic soils present in the maritime fringes of Western Europe. As only one full
39 year was monitored in the nutrient rich Site B and 2 years at Site A, a longer monitoring
40 period to increase the range of climatic conditions and additional direct measurements
41 from other regionally representative drained organic soils would help attain more robust
42 emission factors by averaging inter-annual variability. In turn, this would help to
43 implement more effectively Tier 2 methodologies for IPCC inventories. In the context of

1 the new opportunity given by the IPCC 2013 Wetland Supplement to report rewetted
2 drained organic soils, this study underpins the need to investigate the rewetting of high
3 C content nutrient rich organic soils under grassland in Ireland as a priority. The
4 sustained high decomposition rates recorded at the nutrient rich site could only be
5 compensated by reducing microbial oxidation of peat via higher water table levels.
6 Rewetting of these site types should therefore be a priority of any climate change
7 mitigation strategy focusing on decreasing C losses from soils in Ireland. Furthermore,
8 the neutral or C sink capacity of nutrient poor organic soils may also be strongly affected
9 by climate change. With higher predicted temperatures affecting peat decomposition,
10 respiration rates might increase faster than photosynthetic assimilation due to nutrient
11 limitation. Predicted higher precipitation for the studied region could however work in
12 synergy with government-led actions to 1) introduce agricultural schemes to maintain a
13 relatively high water table (above -25 cm); and 2) reduce the grazing regime to actively
14 promote C uptake by these ecosystems.

15 **5 Concluding remarks**

16 This study has provided estimations of GHG fluxes and waterborne C losses from
17 contrasting grasslands over organic soils and has shown that these reflect not only
18 regional variability but also local management practices. Hence, despite the critical role
19 of nutrient and drainage status, detailed reporting of grassland management type may
20 provide significant additional information for more accurate regional extrapolation of C
21 fluxes. The large difference in net C balances between the main locations stems mostly
22 from 1) NEE estimations and 2) biomass exports, which are affected by the grazing
23 regime. NEE estimations were driven mainly by local climate, soil fertility, water table
24 level and potentially soil organic matter quality. These attributes are in turn intimately
25 linked to past and current management practices in terms of drainage duration and
26 intensity and inputs. Most Irish grasslands over organic soils were drained several
27 decades ago and are managed as extensively grazed pastures with little or no
28 fertilisation. Nutrient poor sites in Ireland may not exert appreciable impacts on the
29 atmosphere in terms of NECB, particularly where the mean annual water table remains
30 within -25 cm of the soil surface. In contrast, grassland over nutrient rich, drained
31 organic soils (with higher C content) produce much higher GHG emissions and also
32 represent hotspots for fluvial C losses. As the C balance of such sites is driven to a large
33 extent by soil respiration, the inverse relationship of this parameter with soil moisture
34 suggests these sites should be targeted for rewetting as a strategy to mitigate C
35 emissions.

36 This comprehensive investigation supplied effective information for Ireland to refine
37 emission factors currently proposed for Tier 1 drained organic soils. Longer monitoring is
38 required to capture inter-annual variability which affects NEE in particular (being the
39 small difference between two large terms GPP and R_{eco}) (Drösler et al., 2008). While a
40 'grassland type' stratification, representing various management systems, may not be
41 warranted at international guidelines levels, national research should be extended to
42 other sites to provide more accurate emissions from extensive drained organic soils
43 pastures and help Ireland progress towards Tier 2 methodologies.

1

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Table 1. Selected climatic data from two research locations during Year 1 (April 2011 to March 2012) and Year 2 (April 2012 to March 2013) and compared with the 30-year average (1981-2010) (Met Eireann, 2013).

	Site A	Site B
Precipitation (mm)		
Year 1	1211	1003
Year 2	1193	1081
30 year average	1076	941
Air temperature °C		
Year 1 (min;max)	10.2 (-3.6; 23.4)	9.6 (-6.6; 24.1)
Year 2 (min;max)	9.7 (-1.7; 22.6)	8.4 (-6.3; 26)
30 year average	9.8 (-1; 17.4)	9.1 (-3.2; 18)

Table 2. Soil and land use characteristics of the research sites.

	Site A		Site B
Location in Ireland	North-west, Atlantic coast		Midlands
Latitude; longitude	55°9'N; 7°34'W		53°39' N; 7°56'W
Soil description	Terric histosol		Limnic histosol
Fertility	Nutrient poor		Nutrient rich
Drainage	Deep drained Site A _d	Shallow drained Site A _s	Deep drained
pH (in water)	4.73	4.93	5.50
Bulk density (g/cm ³)	0.41	0.47	0.31
OM % (LOI)	30	42	70
C % ¹	17.4	23.1	38.1
N % ¹	1.09	1.10	2.25
C:N	16	21	17
Carbon density (g C dm ⁻³)	71.3	108.5	124.3
Nitrogen density (g N dm ⁻³)	4.5	5.2	7.0
P %	0.14	0.13	0.24
K (ppm)	197	188	278
NH ₄ -N (mg/L)	nd	nd	0.51
NO ₃ -N (mg/L)	0.99	0.42	2.61
Vegetation: dominant species	<i>Holcus lanatus</i> , <i>Agrostis stolonifera</i> , <i>Epilobium</i> <i>angustifolium</i>	As Site A _d + <i>Equisetum palustre</i> , <i>Juncus effusus</i> , <i>Juncus articulatus</i> in some plots	<i>Holcus lanatus</i> , <i>Agrostis</i> <i>stolonifera</i> , <i>Rumex acetosa</i> , <i>Cirsium arvensis</i> , <i>Phleum</i> <i>pratense</i>

C=Carbon; N=Nitrogen; P=Phosphorus; K = Potassium. OM = Organic Matter; LOI (Loss-on-ignition); nd: not detectable.¹ C and N were directly measured using Exeter Analytical CE440 CHN elemental analyser

Table 3. Estimated parameter values for gross photosynthesis (GPP) and ecosystem respiration (R_{eco}) models (see related equations in the text). Standard error of the model parameter in parentheses. Coefficient of determination (r^2) values and equation number are shown. p values for all parameters < 0.005 .

Study site	Site A	Site B
Model Parameters		
GPP		
P_{max}	5187.1 (289.6)	9055.3 (379.7)
k_{PPFD}	403.9 (44.1)	814.61 (65.1)
a	0.19 (0.02)	0.25 (0.02)
r^2	0.72	0.85
Equation No.	1	1
R_{eco}		
a	296.5 (31.1)	104.9 (6.3)
b	-6.06 (0.69)	-10.2 (3.6)
c	425.3 (22.1)	63.9 (2.7)
r^2	0.75	0.67
Equation No.	2	3

Table 4. Annual fluxes of ecosystem respiration (R_{eco}), gross primary production (GPP), net ecosystem exchange (NEE) of CO_2 , CH_4 , dissolved organic carbon (DOC), particulate organic carbon (POC), excess CO_2 and bicarbonate (HCO_3^-) together with biomass exports, CH_4 emissions from enteric fermentation and net ecosystem carbon balance (NECB). Standard errors are shown in parentheses for all gas fluxes R_{eco} , GPP, NEE and for fluvial fluxes. One standard deviation is shown in parentheses for all other components. Site A_d = nutrient poor deep drained, Site A_s = nutrient poor shallow drained and Site B= nutrient rich drained. Positive values indicate a loss of carbon from the site and negative values indicate an uptake of carbon by the site.

	R_{eco} g C m ⁻² yr ⁻¹	GPP g C m ⁻² yr ⁻¹	NEE g C m ⁻² yr ⁻¹	CH_4 g C m ⁻² yr ⁻¹	CH_4 livestock g C m ⁻² yr ⁻¹	Biomass export g C m ⁻² yr ⁻¹	NEE + Biomass g C m ⁻² yr ⁻¹	DOC g C m ⁻² yr ⁻¹	POC g C m ⁻² yr ⁻¹	Excess CO_2 g C m ⁻² yr ⁻¹	HCO_3^- g C m ⁻² yr ⁻¹	NECB g C m ⁻² yr ⁻¹
Year 1												
Site A_d	1736 (17)	-1651 (47)	85 (50)	0	3.6	225 (60)	310	18.1 (3.2)	6.4 (2.7)	13.4 (3.2)	6.1 (0.6)	358 (111)
Site A_s	1260 (49)	-1349 (54)	-89 (73)	1.3 (1.09)	3.6	174 (25)	85	18.1 (3.2)	6.4 (2.7)	13.4 (3.2)	6.1 (0.6)	134 (75)
Site B	2322 (53)	-2089 (57)	233(78)	0	9.5	351 (40)	584	37.7 (9.0)	7.9 (3.7)	4.3 (0.9)	19.9 (3.7)	663 (42)
Year 2												
Site A_d	1535 (6)	-1561 (50)	-26 (50)	0	3.6	185 (36)	159	4.9 (0.26)	2.0 (2.7)	18.5 (3.0)	5.4 (0.3)	193 (64)
Site A_s	1112 (36)	-1211 (58)	-99 (68)	1.4 (1.1)	3.6	137 (28)	38	4.9 (0.26)	2.0 (2.7)	18.5 (3.0)	5.4 (0.3)	73 (56)

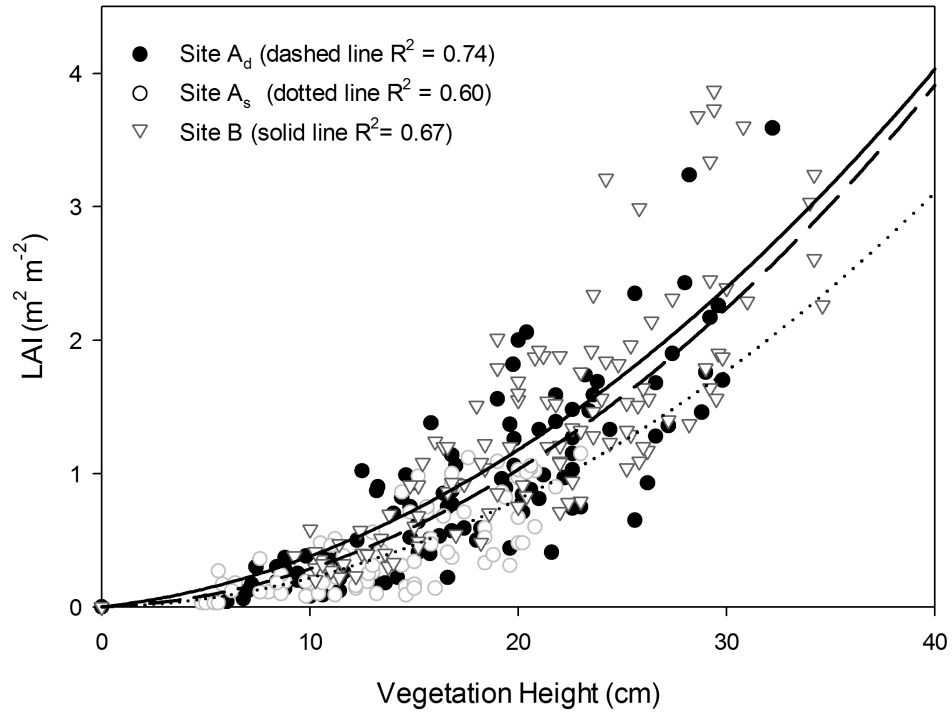


Fig. 1. Relationships between vegetation height (H) and leaf area index (LAI); the equations for the fitted regression lines are for Site A_d: $LAI = 0.0023 \cdot H^2 + 0.0056 \cdot H$; Site A_s: $LAI = 0.0019 \cdot H^2 + 0.003 \cdot H$ and Site B: $LAI = 0.0021 \cdot H^2 + 0.0171 \cdot H$.

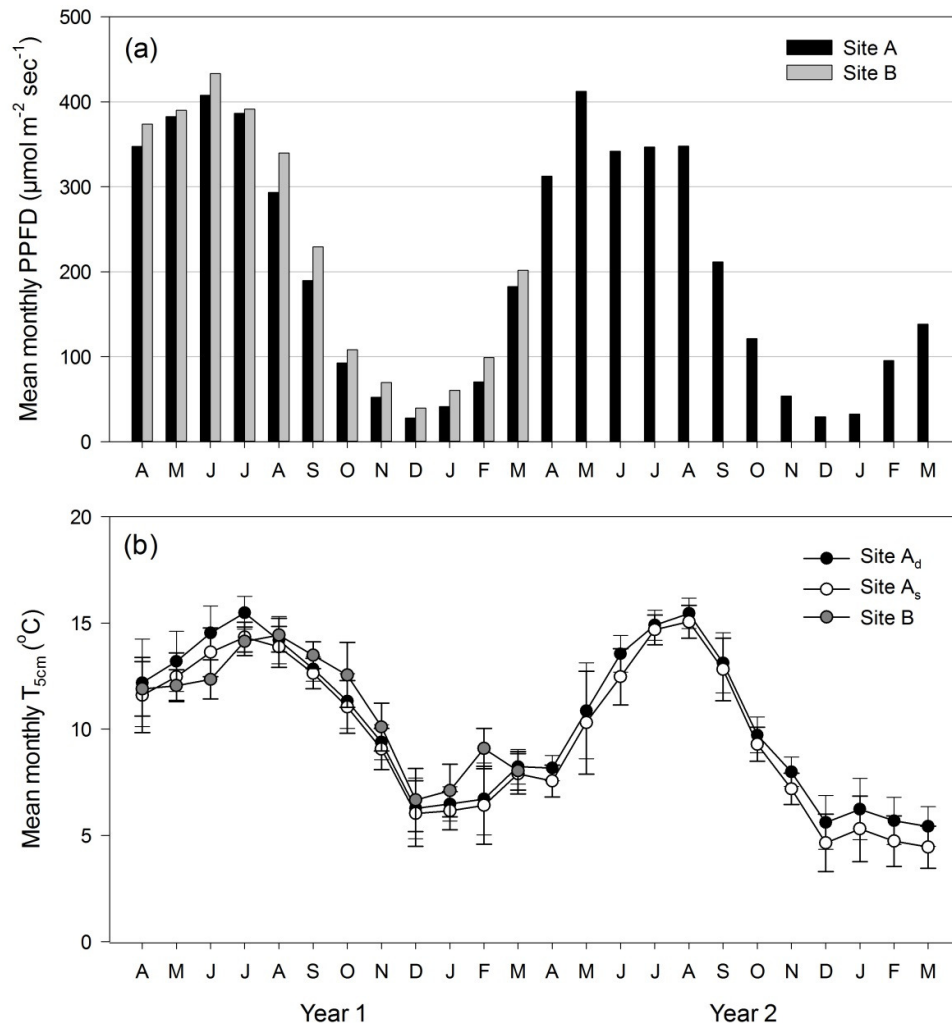


Fig. 2. (a) Mean monthly Photosynthetic Photon Flux Density (PPFD) ($\mu\text{mol m}^{-2} \text{s}^{-1}$) at Site A (April 2011 to March 2013) and Site B (April 2011 to March 2012) and (b) mean monthly soil temperature at 5 cm depth ($T_{5\text{cm}} \pm$ standard deviation) at Site A_d, Site A_s and Site B.

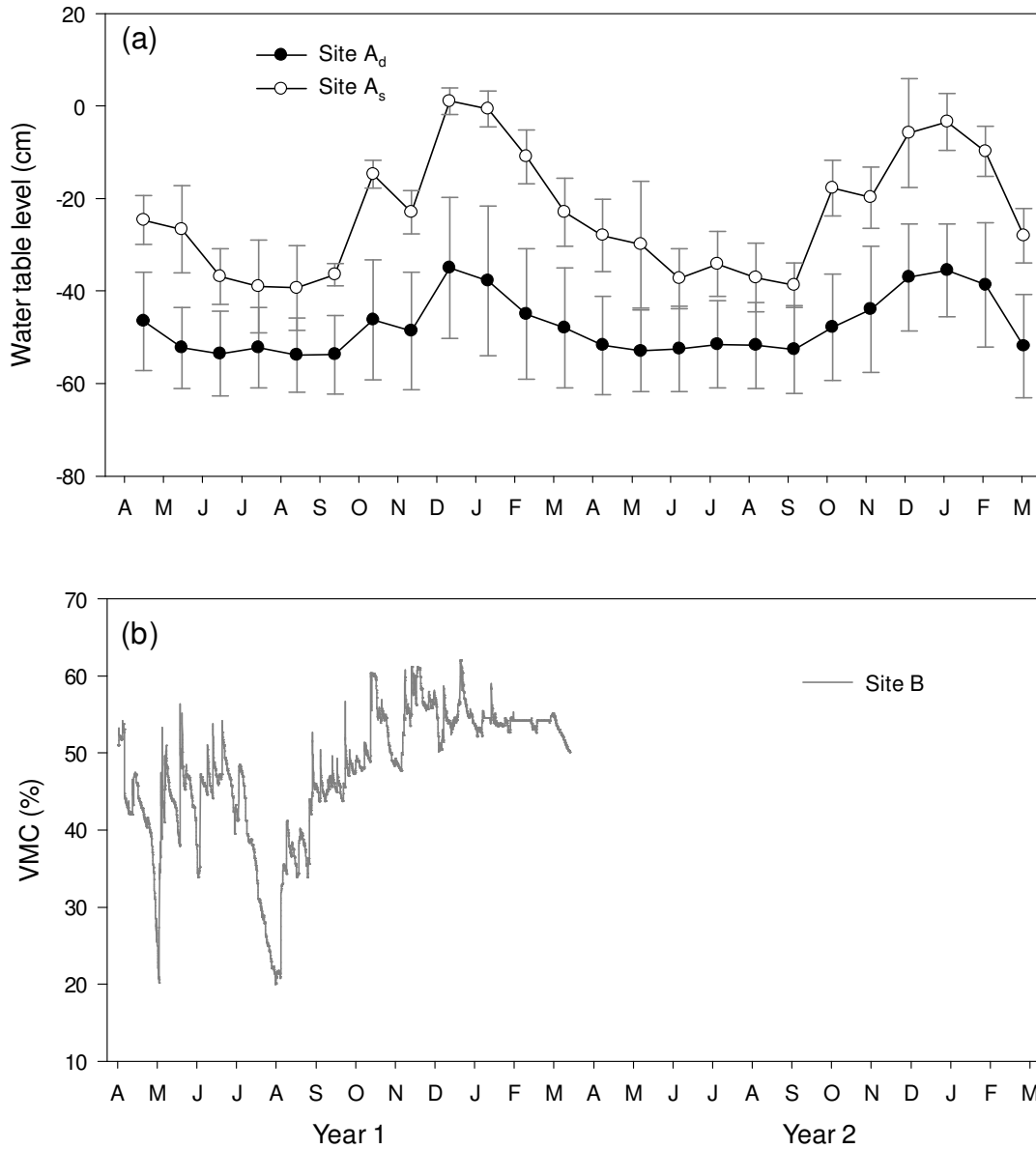


Fig. 3. (a) Mean monthly water table levels (cm) at Site A_d and A_s (error bars are standard deviation of the monthly means) and (b) Volumetric Moisture Content (% VMC) at Site B (in the surface 5 cm of the peat).

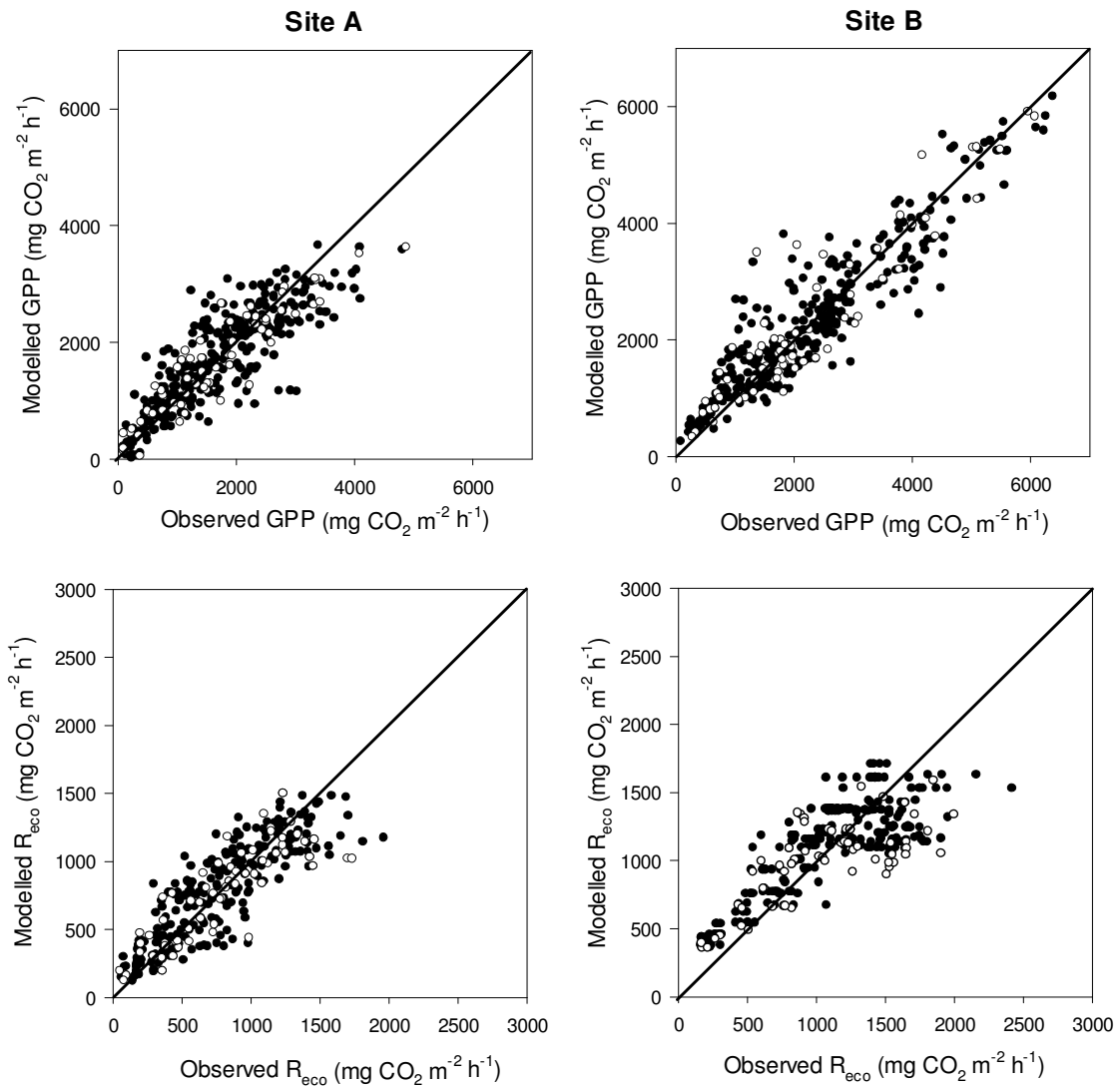


Fig. 4. Relationship between observed and modelled Gross Primary Production (GPP) and Ecosystem Respiration (R_{eco}) fluxes ($\text{mg CO}_2 \text{ m}^{-2} \text{ h}^{-1}$) for Sites A and B. Dark circles indicate data used in the construction of models and open circles indicate independent test data.

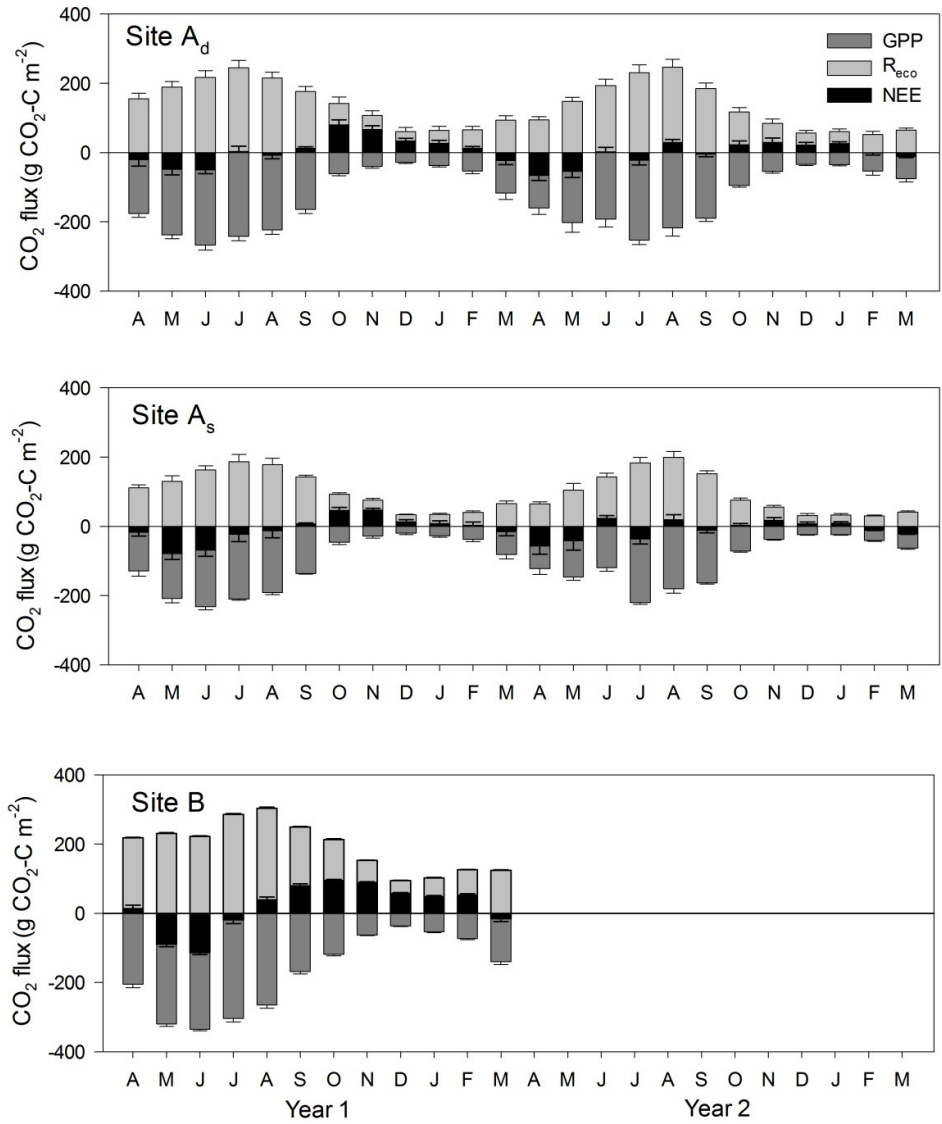


Fig. 5. Monthly mean gross primary production (GPP), ecosystem respiration (R_{eco}) and net ecosystem exchange (NEE) ($\text{g CO}_2\text{-C m}^{-2} \text{ month}^{-1}$). Error bars represent \pm standard deviation. $n=7$ for Site A_d, $n=5$ for Site A_s and $n=9$ for Site B. Negative NEE values indicate that the site was a net sink for CO₂-C for that month.

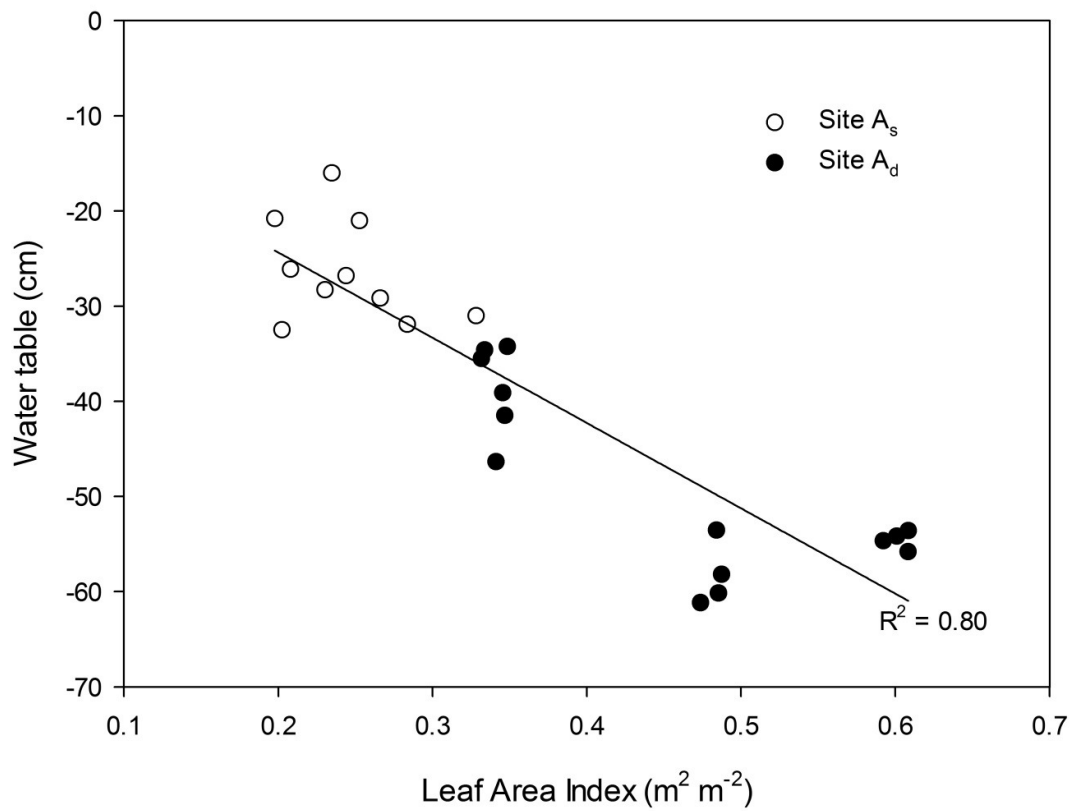


Fig. 6. Relationship between mean annual water table (cm) and mean annual Leaf Area Index (LAI) at the nutrient poor sites A_d and A_s (Year 1 and 2 combined).

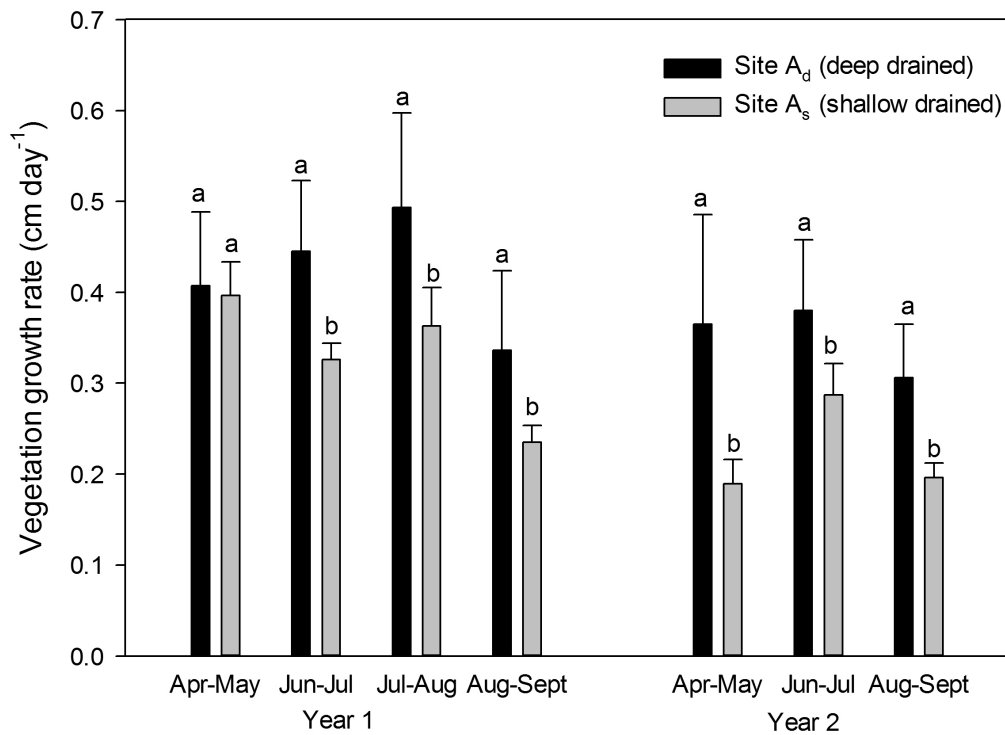


Fig.7. Vegetation mean growth rate (cm day⁻¹) at Sites A_d and A_s during periods between simulated grazing events (cuttings) during Year 1 and 2 (vertical bars denote +/- one standard error; n=7 for Site A_d and n=5 for Site A_s). Means with the same letter are not significantly different (p<0.001).

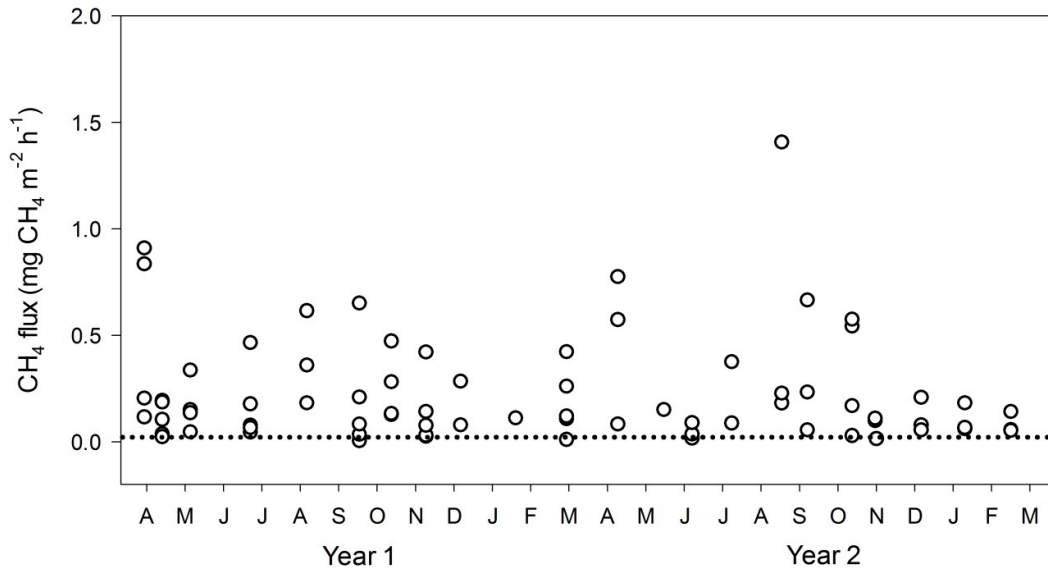


Fig. 8. Measured CH₄ fluxes (mg CH₄ m⁻² h⁻¹) at Site A₅ (nutrient poor, shallow drained).

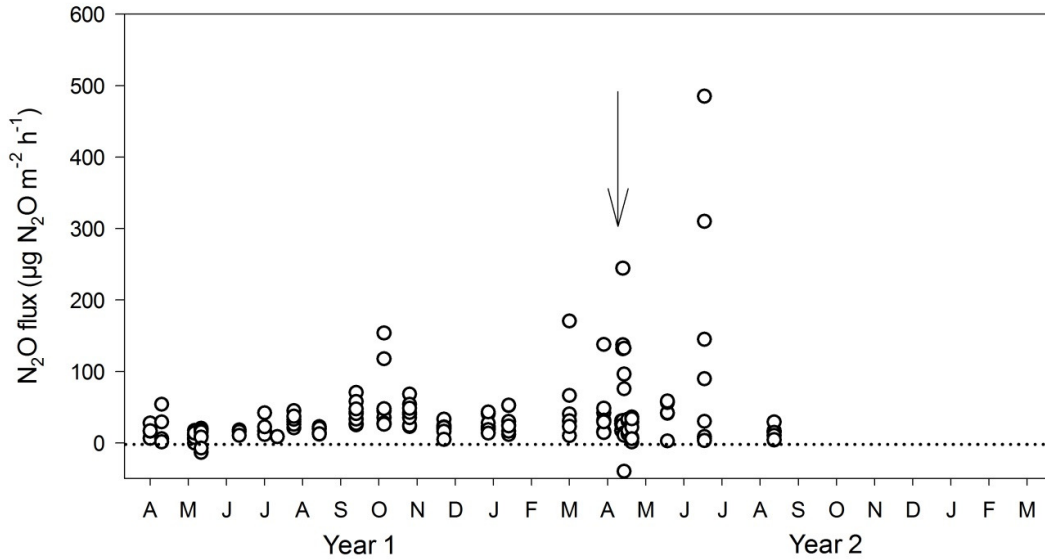


Fig. 9. Measured N₂O fluxes (µg N₂O m⁻² h⁻¹) at Site B. The arrow represents a fertilisation event (25kg N/ha).