

Carbon Loss from Drained Organic Soils under Grassland – CALISTO

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by

University College Dublin

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Executive Summary

Temperate grasslands on organic soils are diverse due to edaphic properties but also because of regional management practices, and this heterogeneity is reflected in the wide range of greenhouse gas (GHG) flux values reported in the literature. In Ireland, most grasslands on organic soils were drained several decades ago and are managed as extensive pastures with little or no fertilisation. This study describes a two-year study of the net ecosystem carbon balance (NECB) of two such sites and aims to deliver data to allow the Republic of Ireland (ROI) to progress towards a higher-tier reporting level for this land use category (LUC). It also provides a comprehensive overview of carbon (C) dynamics in organic soils by investigating all potential C imports and exports, including fluvial C loss.

Greenhouse gas (GHG) fluxes and waterborne C emissions in a nutrient-rich grassland were determined and compared with values measured from two nutrient-poor organic soils: (i) a deep-drained and (ii) a shallow-drained site.

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

The NECB of the nutrient-rich grassland was 663 g C m⁻² yr⁻¹, with biomass exports being the major component. The NECB of the nutrient-poor deep-drained site was less than half of the nutrient-rich site. Although NEE at the nutrient-poor shallow-drained site was negative ('C sink') in both years, high biomass export meant it was a net 'C source' (2-year mean NECB 103 g C m⁻² yr⁻¹). While the impacts of the nutrient and drainage status on NEE, biomass exports and fluvial C losses were

confirmed, inter-regional differences in management practice and climate were also significant factors that impacted on the overall NECB of these ecosystems. Contrary to expectations, the NECB of nutrient-poor drained organic soils under grasslands is not necessarily a large C source and this has implications for Ireland's choice of national GHG inventory reporting methodologies. This study can also aid the development of strategies to deliver reduced emissions tailored to local grassland types.

Final Observations and Recommendations

Observation 1: Our field research concluded that (i) nutrient status, (ii) drainage and (iii) local grassland-management practices impacted on GHG fluxes and fluvial C losses as well as biomass exports.

Recommendation 1: As well as using nutrient and drainage status, types of grassland-management practices provide significant additional information for more accurate regional extrapolation of C fluxes. These attributes are in turn intimately linked to past and current management practices in terms of drainage duration and intensity and inputs.

Observation 2: Most Irish grasslands over organic soils were drained several decades ago and are managed as extensively grazed pastures with little or no fertilisation. Nutrient-poor sites in Ireland may not exert appreciable impacts on the atmosphere in terms of NECB, particularly where the mean annual water table remains within -25 cm of the soil surface.

Recommendation 2: Nutrient-poor organic soils that are poorly drained should be targeted to remain wet (water table depth [WT] higher than -25cm) with a continued low input/output system.

Observation 3: Nutrient-rich organic soils under grasslands produce much higher GHG emissions and also represent hotspots for fluvial C losses.

Recommendation 3: Nutrient-rich organic soils should be targeted for rewetting as a strategy to mitigate C emissions.

Observation 4: Current default emission factors (EF) may not be representative of the variety of grasslands over organic soils in Ireland. Our field-measured EF will help Ireland to refine EF and implement Tier 2 methodologies for IPCC inventories more effectively.

Recommendation 4: Continued monitoring to improve annual flux estimates to develop more robust emission factors by averaging inter-annual variability.

Observation 5: While the impacts of the nutrient and drainage status on CO₂ exchanges, biomass exports and fluvial C losses were confirmed, inter-regional differences in management practice and climate are also significant factors that impact on the overall NECB of these ecosystems.

Recommendations 5: It is therefore critical to develop strategies to deliver reduced emissions tailored to local grassland types.

1 Introduction

1.1 Background

Climate change is a major threat to environmental and sustainable development and these concerns have resulted in the establishment of the United Nations Framework Convention on Climate Change (UNFCCC). Signs of the resultant adverse impacts of climate change are already evident in Ireland (Desmond *et al.* 2009). Under the UNFCCC and the Kyoto Protocol, Annex 1 countries, such as Ireland, are obliged to submit annual inventories of GHG emissions in the form of National Inventory Reports (NIR), detailing GHG emissions and sinks from six different sectors: (i) energy, (ii) industrial processes, (iii) solvents and other product use, (iv) agriculture, (v) Land use, Land Use Change and Forestry (LULUCF) and (vi) waste. The International Panel for Climate Change (IPCC) Good Practice Guidance has suggested three tier levels for the estimation of GHG emissions/removals. Tier 1 uses IPCC default emission factors (EFs) with limited separation of area data. Tier 2 uses country-specific EFs and finer-scale separation of data and Tier 3 uses more complex country-specific approaches (Penman *et al.* 2003). In order to provide a robust national inventory of emissions, the IPCC GPG recommends progression towards a higher tier methodology requiring certain gaps in data and in scientific understanding to be addressed.

In Ireland, agriculture is the largest contributor to overall national emissions, accounting 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 nineteenth century which intensified in the twentieth century due to national drainage Acts and agricultural schemes (Feehan *et al.* 2008). Nowadays, some 300,000 ha of organic soils is under grassland (with negligible areas cultivated for crops), on a par with organic soils drained for forestry. Altogether, managed organic soils (including industrial cutaway peatlands) represent nearly half of the total peat soil area or 10% of the total land area (Wilson *et al.* 2013).

The use of organic soils for agriculture is a contentious land-use option in terms of atmospheric impacts. 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 and CH₄ from Irish peat soils and related activities are estimated to account for c. 3 Mt C per annum (Wilson *et al.* 2013): this is 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, the new activity called ‘rewetting and drainage’ of organic soils has been included in the next post-Kyoto Protocol commitment period (2013–2017). The 2013 *Wetland Supplement of the Intergovernmental Panel on Climate Change* (IPCC, 2014) has developed new guidelines, together with default EFs for drained and rewetted organic soils, stratified by climate zone, nutrient status¹ and in some cases by drainage class.²

1 Defined by the GPG-LULUCF and the 2006 IPCC Guidelines (Section 7.2.1.1, Chapter 7).

2 Defined as the mean annual water table averaged over a period of several years; the shallow-drained class is defined as the mean annual water table depth of less than 30 cm below the surface; the deep-drained class is defined as the mean annual water table depth of 30 cm and deeper below the surface.

Using Tier 1 methodology of the 2013 Wetlands Supplement, instead of the 2006 GPG, would imply a significant increase in total emissions from drained organic soils because of the higher default EF for CO₂ and N₂O emissions as well as additional new default EFs reported for CH₄ and dissolved organic carbon (DOC). These adjusted EFs have been compiled from an increasing large body of research which has demonstrated that drainage and 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 all the GHG fluxes (but in particular CO₂) 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, the drainage of temperate organic soils typically leads to higher DOC fluxes (Wallage *et al.* 2006, Urbanová *et al.* 2011), so that management practices may alter both contemporary photosynthetic C uptake rates, but also the stability of older soil stores of sequestered C. Further, although fluvial C exports are increasingly recognised as potentially significant components of terrestrial C balances, full accounting of all fluvial C forms remains the exception rather than the rule, notably for dissolved inorganic carbon (DIC) and particulate organic carbon (POC).

Therefore, improvement in terms of accuracy of the reporting of this land use category (LUC) in Ireland is necessary as both the total area and magnitude of emissions involved is significant. In addition, an EU review of the National GHG Inventory submissions (Consistency_report-IE_01_2009) concluded that Ireland's use of default EFs for CO₂ emissions for managed organic soils should be strongly reconsidered for future reporting. As such, more accurate quantification of all C fluxes (CO₂, CH₄, and fluvial C) is essential to allow reporting to move to Tier 2 levels, bringing Ireland's reporting in line with other EU member states with a high proportion of organic soils. Moreover, given that agricultural soils are the main source of N₂O emissions in the Republic of Ireland (ROI), there

is an obvious need to fill the knowledge gap of actual N₂O emissions from organic soils under grassland and investigate their spatial and temporal variability. This project not only delivers data to allow ROI to progress towards a higher tier reporting level for this LUC but also provides a comprehensive overview of C dynamics in organic soils by investigating all potential C imports and exports including fluvial C loss. While quantifying exported aquatic C from this organic soil is important and has only been done once in Ireland from a near-intact peatlands (Koehler *et al.* 2011), quantifying its bioavailability (ie. its actual contribution to CO₂ emissions downstream) is a critically new research area in Ireland. If we consider the whole grassland system on organic soils, C exports in the form of off-site emissions from exported biomass, digesting and enteric fermentation by cattle of the cut herbage and meat and/or milk, together with C imports in the form of cattle manure, fertiliser and concentrate feed should also be included. Altogether, the results from this study allow us to present the first complete net ecosystem carbon balances (NECB) for managed organic soils under grassland in Ireland. [Figure 1.1](#) represents all GHG fluxes and other C input and removal pathways.

Finally, management of organic soils has been proposed as a measure for mitigating GHG emissions from agricultural ecosystems. Therefore, this project reviews the most important mitigation practices available and tests the hypothesis that optimising the water table levels (i.e. the depth of drainage) would be an effective reduction/mitigation of C emissions at both the field and national level.

1.2 Objectives

The project aimed to produce improved estimates of GHG emissions and fluvial C losses from organic soils under grassland. Hence, the main product of this project is the experimentally derived EFs for typological classes of this LUC that are most representative in Ireland, therefore giving practical tools for Tier 2 national inventory reporting.

The detailed objectives of this project were to:

- 1 Provide an overview of the most up-to-date state of knowledge regarding managed organic soils in Ireland, their land-uses and management practices as well as their associated C emissions (known and unknown) and implications for policy;

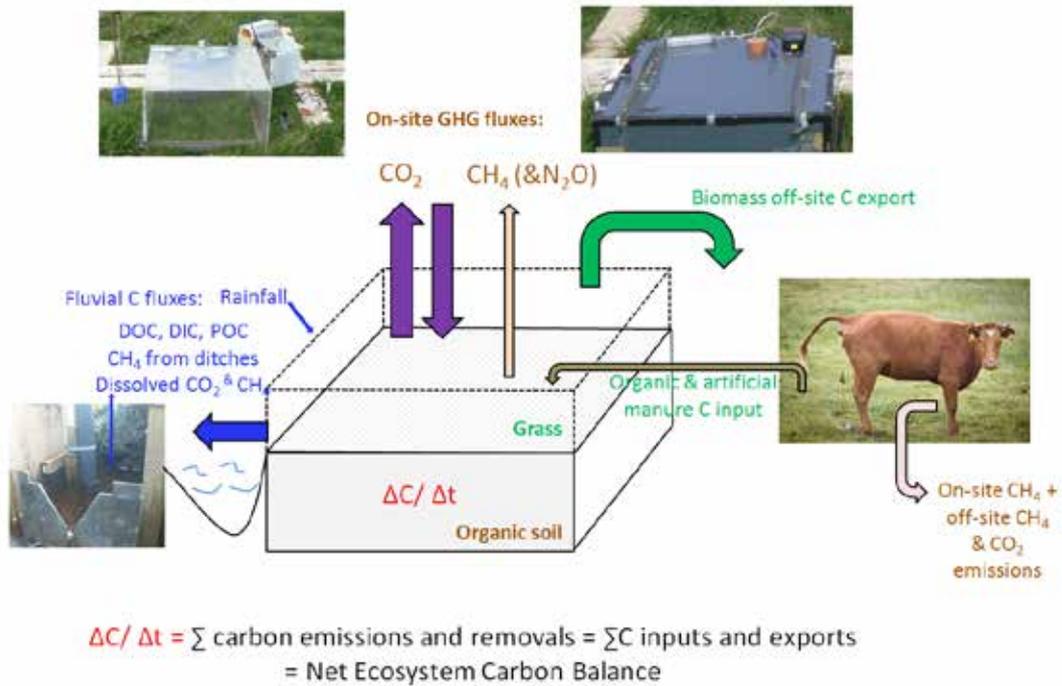


Figure 1.1. Summary of greenhouse gas (GHG) fluxes (carbon dioxide [CO₂], methane [CH₄], nitrous oxide [N₂O]) as well as fluvial C fluxes (dissolved organic carbon [DOC], dissolved inorganic carbon [DIC], particulate organic carbon [POC]) from drained organic soils used for a typical extensive grassland.

- 2 Quantify, through field research, the annual fluxes of CO₂ (gross photosynthesis, ecosystem respiration), CH₄, N₂O, DOC, POC and DIC of the selected grassland sites and present their associated uncertainties;
- 3 Develop annual average EFs for CO₂, CH₄ and N₂O as well as DOC from organic soils under grassland and their associated uncertainties;
- 4 Provide a better characterisation of fluvial C losses by estimating the proportion of DOC likely to return to the atmosphere as CO₂ using dissolved organic matter (DOM) bioavailability and assess its relative contribution to the annual C budget from each field site;
- 5 Estimate the complete NECB of each investigated site (annual net gaseous and fluvial C budget);
- 6 Assess the potential of mitigation measures to reduce C emissions through management of these organic soils (e.g. through water table manipulation) and assess the cost and benefits associated with these.

2 Managed Organic Soils: State of Knowledge

2.1 Peat Soils, Organic Soils and Histosols

Organic soils and peat soils have been variously defined, depending on country and scientific discipline. Peat is included in soil classification systems across European countries under names such as 'peat soils', 'muck soils' and 'organic soils', but at an international level these are commonly referred to as 'histosols' (FAO-UNESCO 1974, FAO-UNESCO-ISRIC 1990, FAO 1998). The 2006 IPCC Guidelines also largely follow the FAO definition of 'histosols' and equate organic soils to peat soils. However, they omit the depth criterion from the FAO definition to allow for country-specific definitions of organic soils.

For the purpose of IPCC methodologies (2006 IPCC Guidelines and 2013 Wetland Supplement), an organic soil is a soil with a high concentration of organic matter and must satisfy Requirements 1 and 2 or 1 and 3 below (2013 Wetland Supplement, p. 1.7):

- 1 Thickness of organic horizon greater than or equal to 10 cm. A horizon of less than 20 cm must have 12 percent or more organic carbon when mixed to a depth of 20 cm.
- 2 Soils that are never saturated with water for more than a few days must contain more than 20 percent organic carbon by weight (i.e., about 35 percent organic matter).
- 3 Soils are subject to water saturation episodes and have either:
 - a. At least 12 percent organic carbon by weight (i.e., about 20 percent organic matter) if the soil has no clay;
 - or
 - b. At least 18 percent organic carbon by weight (i.e., about 30 percent organic matter) if the soil has 60% or more clay; or
 - c. An intermediate proportional amount of organic carbon for intermediate amounts of clay.

The 2006 IPCC Guidelines do not give any definition for peat or peatlands but it is considered to be included in 'organic soils'. The 2013 Wetlands Supplement

follows the 2006 definitions with the addition that 'it is good practice that when a country uses another definition of organic soil in accordance with its national circumstances, the concept of organic soil applied is clearly defined and that the definition is applied consistently both across the entire national land area and over time' (p. 1.7).

As part of the EPA-funded BOGLAND project (Renou-Wilson *et al.* 2011), the following scientific definition of 'peat soil' was agreed by a national expert group,³ based on the classification of peat soils by Gardiner and Radford (1980) and Hammond (1981) as well as international working groups such as the International Peat Society and the International Mire Conservation Group (Joosten and Clarke 2002):

Peat soil: organic soil materials which have sedentarily accumulated and have at least 30% (dry mass) organic matter over a depth of at least 45 cm on undrained land and 30 cm deep on drained land; the depth requirement does not apply in the event that the peat layer is over bedrock.

(Renou-Wilson *et al.* 2011, p.14)

This definition coincides with the FAO definition of 'histosol' with the notable exception that the FAO definition includes areas with shallow peat layers over ice or rock. Using the widely accepted ratio between organic carbon (OC) and organic matter (OM) – OC:OM of 1:1.72 – a content of 30% soil organic matter (SOM) would equate to ~17% soil organic carbon (SOC). According to the National Soils Database (Fay *et al.* 2007), the term 'organic soils' was used for all soils with a SOC content >15% (~25% SOM).

In the context of IPCC methodologies, the definition of organic soils is heterogeneous across the European Union (27) and not transparently provided in the national

3 Present were representatives of Teagasc, National Parks and Wildlife Service, Bord na Móna, Soil Science Department, University College Dublin (UCD) (including the late Bob Hammond), Bio-system Engineering-UCD, Biology and Environment-UCD, Environmental Engineering-Trinity College Dublin and National University of Ireland Galway.

GHG inventory reports. In 2013 only four member states, including Ireland, provided a definition.

In Ireland, organic soils were defined as having high organic matter (greater than 20% OM). When the peat depth was greater than 30 cm they were classified as peat soils. If the organic or peat layer is less than 30 cm then the soil is classified as 'organo-mineral' (or 'peaty-mineral'). These correspond to areas of natural wetlands that have been drained for the purpose of human exploitation (O'Brien 2007). 'Wet organic soils' are defined as those that have a water table between 0 and 30 cm.

Finland and Denmark have a very similar definition of organic soils with soils having >20% OM in the top 20 or 30 cm respectively.

In the Wetlands Supplement, a wet soil is not defined by the water table but as a soil that is inundated or saturated by water for all or part of the year to the extent that biota, particularly soil microbes and rooted plants, adapted to anaerobic conditions control the net annual GHG emissions and removals.

2.2 Extent of Organic Soils and Utilisation

Definitions of soil types are important when it comes to mapping their extent. It has been acknowledged that the distribution of peat soils is probably more accurately portrayed by the Map of Organic Carbon in Topsoils of Europe (Jones *et al.* 2004) than by the European Soil Map and Database (Commission of the European Communities [CEC] 1985). In Ireland, the first great survey of Ireland's peatland was carried out between 1809 and 1814 (Commissioners 1814). Hammond's Peatland Map of Ireland with its associated *Soil Survey Bulletin* was the first publication that classified the extent of the various peat types in the country (Hammond 1981). Using Hammond's data together with other sources of spatial data in the form of the Indicative Soil Map of Ireland (R. Fealy, Teagasc, Pers. Comm.) and CORINE 2000 (EPA 2003) the spatial extent of peat soils in Ireland was estimated. A new Derived Irish Peat Map Version 2 (Connolly and Holden 2009) was produced showing that peat soils cover 1,466,469 ha or 20.6% of the national land area with an overall accuracy of 88% (in contrast to the 17% reported under 'Wetland soils' in National Inventory Rreport [NIR] 2009

[(McGettigan *et al.* 2009])). This research confirmed that organic soils are a key soil type in Ireland, with the majority being ombrotrophic (nutrient poor) in nature. An analysis of main peatland land-use categories with a view to reporting their carbon emissions and removals estimated that cutover peatlands were the largest land-use category covering 612 kha followed by grassland (295 kha) and with similar area forestry (293 kha), leaving 269 kha of natural peatlands (Wilson *et al.* 2013). The latest estimation of organic soils under grassland cover is 282 kha (Common Reporting Format [CRF] Table 5.C, Inventory 2011–2013), which is a decrease from previous NRI estimations (295 kha in CRF Table 5.C, Inventory 2007–2009). The NIR (Duffy *et al.* 2013) acknowledges the high uncertainty associated with their method of overlaying a CORINE 1990 grassland cover data set over the General Soil Association Map of Ireland (Gardiner and Radford 1980). The majority of grasslands on organic soils are semi-natural wet grassland habitats.

Nearly all organic soils, including peat soils, have been impacted by anthropogenic activities over the course of history. Agriculture has been a common land use of peat soils for several centuries. Reclamation of peatlands for grassland was accelerated during the eighteenth and nineteenth centuries as a result of population pressures (Feehan *et al.* 2008). The reclamation and drainage of organic soils was intensified in the twentieth century as a result of several Acts and schemes, including the 1945 Arterial Drainage Act, the Farm Improvement Programme and the Programme for Western Development. Most of these reclaimed soils have been used for grassland. The twentieth century saw a sharp increase in the use of peat soils because of:

- The introduction of mechanised turf-extraction schemes (both industrial and domestic);
- Afforestation schemes;
- Intensification of agriculture through the Common Agriculture Policy; and
- Land reclamation through drainage schemes.

Overall, some 85% of Irish peatlands have been drained and converted to agriculture, forestry or extracted for fuel. Thus, organic soils now occur under different land uses: forest, grassland and a very small area under agricultural crops. According to national expert opinion (Teagasc), the drainage of organic soils for crops is

insignificant in Ireland but the NIR acknowledges a high uncertainty associated with this LUC, which should be resolved with additional research.

In addition, Bord na Móna successfully transformed approximately 2,500 ha of industrial cutaway peatlands into productive grasslands and these have been subsequently sold to individual farmers (under IPCC methodologies, previously exploited cutaway peatlands converted to grassland are assumed to be improved grassland). However, Bord na Móna has no plans to designate any future cutaways for grassland development (McNally 2008). In conclusion, the area of organic soils (including industrial cutaway peatlands) to be converted to grassland is now deemed to be very limited.

2.3 Management of Organic Soils under Grassland

2.3.1 Grassland Types

Irish grasslands over organic soils are managed for feeding domestic herbivores, mostly through direct grazing with only minimum areas used for forage production (either hay or silage). Most grasslands have been managed to some degree by grazing, mowing, fertilisation application or drainage. Therefore, the classification of grassland is usually made on the level of improvement or management, that is improved versus unimproved. Improved grasslands (usually species poor and intensively managed) are by far the most widespread type in Ireland. Improved grasslands are occasionally reseeded with biomass to increase the yield and are assumed to be artificially drained, fertilised and heavily grazed. Unimproved grassland may receive some inputs of fertiliser but usually this consists only of organic fertiliser through animal dung. They are not intensively managed and are not reseeded, therefore increasing their species richness with a high representation of 'agricultural herbs'. Such management is likely to represent most grassland over organic soils. The inclusion of a new semi-improved grassland class has recently been proposed, most of which has been found predominantly in the west of Ireland (Sullivan *et al.* 2010). In addition to this continuum

between the improved/unimproved grassland types, grasslands can occur in a range of water table levels and the occurrence of wet grassland is still relatively high despite widespread drainage (Hickie *et al.* 1999).

2.3.2 Drainage (Water Table Depth)

While organic soils under grassland are either drained or ditched, that is the water table level is at least temporarily below natural levels, they nevertheless exhibit significant fluctuations in groundwater table during the year. Depending on precipitation, the ground water table may remain well below the drainage depth or close to the surface (inundation being common in those sites that are usually close to river systems). These variations in groundwater levels associated with weather patterns will directly affect the microbial decomposition of the SOM, shifting from aerobic to anaerobic and vice-versa, therefore influencing in turn the GHG balance. To date, there is no data available at a national scale regarding water table regimes.

2.3.3 Fertilisation (Mineral and Organic)

Improved grasslands receive high amounts of mineral and organic fertilisers. Application varies according to stocking rate but intensively managed grassland typically receives between 200 and 300 kg N ha⁻¹ year⁻¹. A low rate of mineral fertiliser may be applied on the most productive semi-improved grassland (between 25 and 50 kg N ha⁻¹ year⁻¹) but dung from the animals is usually the sole annual fertiliser input in unimproved and semi-improved grasslands. Very little additional farm manure or slurry application occur on these pastures as the farm system is usually solely based on livestock production with winter animal dung from housing usually spread on the more productive, improved grassland.

2.3.4 Grazing/Cutting Regime

Grassland in Ireland is used solely for feeding livestock. Continuous livestock grazing is usually the only removal of biomass in unimproved and semi-improved grasslands. On rare occasions, biomass is removed from semi-improved or unimproved grassland with an initial silage cut followed by grazing during the growing season.

3 Study Sites

3.1 Climatic Conditions

The study sites were located in a maritime temperate climate zone in two distinct Irish ecoregions, both characterised as landscapes rich in organic soils (Figs 3.1 and 3.2). Site A at Glenvar, Co. Donegal (Latitude: 55° 9' N, Longitude: 7° 34' W) is situated at 40 m elevation in the north-west of the country, less than 1 km from the sea shore. The climate here is typical of western maritime Ireland with warm winters (very few ground frosts) but cool summers. Site B in Lanesborough, Co. Longford (Latitude: 53° 39' N,

Longitude: 7° 56' W) is also low-lying, situated at 38 m elevation but is typical of the Midlands landscape, with late spring frosts occurring as late as the beginning of June. Despite a comparable long-term annual mean temperature, lower minimum and higher maximum temperatures are typically recorded at Site B. A minor gradient in precipitation is also present with higher mean rainfall in Site A (Table 3.1). Potential evapotranspiration in that region is estimated at 32%–36% of annual precipitation (estimated long-term annual runoff of c. 700 mm), compared to 44% (c. 586 mm) for Site B (Mills 2000).

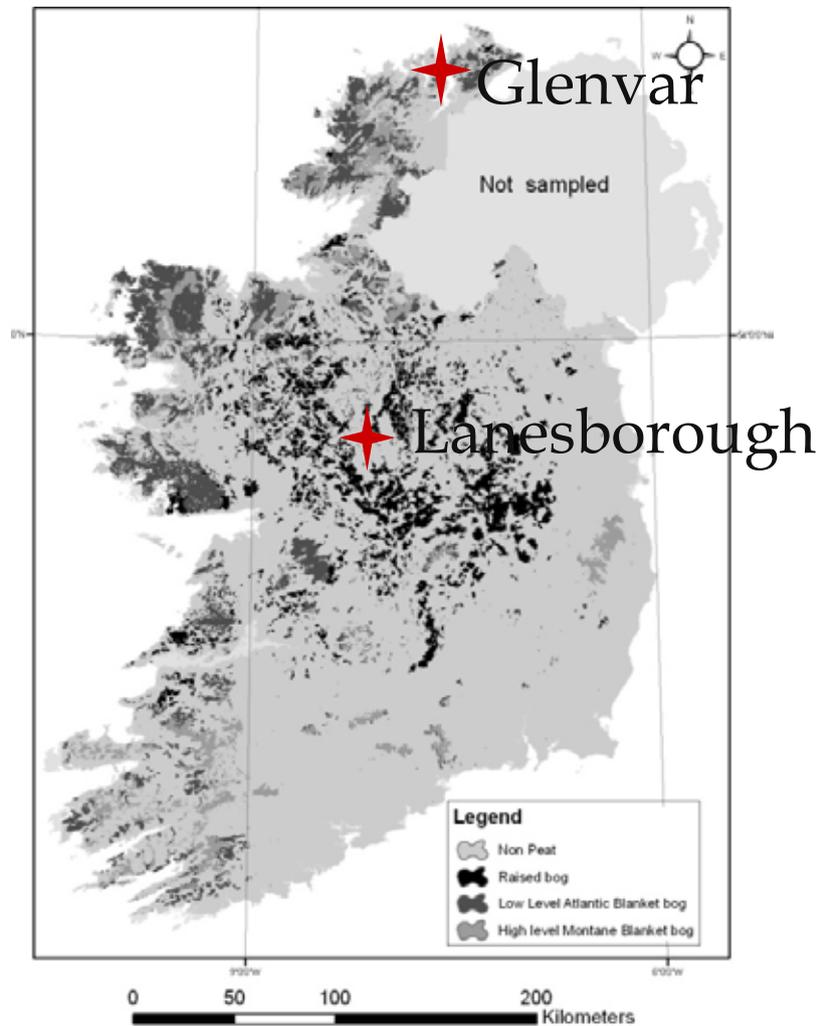


Figure 3.1. Locations of study sites on map of peat soils (re-copied from Connolly and Holden, 2009).



(a)



(b)



(c)

Figure 3.2. Photos of experimental plots in a deep-drained (a) and shallow-drained (b) nutrient-poor site in Glenvar, Co. Donegal and drained nutrient-rich site (c) in Lanesborough, Co. Longford.

Table 3.1. 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 Éireann 2013).

	Site A Glenvar, Co. Donegal	Site B Laneborough, Co. Longford
Precipitation (mm)		
Year 1	1211	1003
Year 2	1193	1081
30-year average	1076	941
Mean annual rain		
days (> 0.2mm day ⁻¹)	247	246
30-year average	226	209
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)

3.2 Land Use and Soil Characteristics

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

Both Sites A and B have been drained for over 60 years and the soils are categorised as 'terric' and 'limnic histosols' respectively (FAO 1998). With an organic matter content higher than 30% (Table 3.2) over a depth greater than 40 cm, they satisfy the definition of peat soils in both the Irish soil classification (Hammond 1981) and the FAO key to soil types (FAO 1998), which form the basis of definitions used in the IPCC

2006 and 2013 guidelines (IPCC 2006, 2013). They are typical 'earthy peat', that is reclaimed drained peat soils characterised by a well-aerated structure and a relatively firm surface horizon containing few or no recognisable plant remains. The peat in Site A overlies bedrock of Precambrian schist and gneiss and contains on average 20% C (Table 3.2). The base geology of Site B is of limestone origin and the overlying peat has a higher pH (5.5) and almost double the C content (40%) and is situated over a thick layer of marl (pH 7.2). Despite similar C:N ratios (16–21), Sites A and B differ in nutrient status and, based on N, phosphorus (P) and potassium (K) content, Site A is nutrient poor while Site B is nutrient rich (Table 3.2).

Site B has a uniform hydrological soil profile throughout the experimental area. In contrast, Site A can be divided into two distinct sub-sites: a deep-drained area 'Site A_d' (defined in the 2006 IPCC guidelines as the mean annual water table depth of 30 cm and deeper below the surface) and a shallow-drained area 'Site A_s' (defined as the mean annual water table depth of less than 30 cm below the surface).

Table 3.2. 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 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 stolonifera</i> , <i>Rumex acetosa</i> , <i>Cirsium arvense</i> , <i>Phleum 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 CE440 CHN Exeter Analytical Elemental Analyser

3.3 Vegetation Profile

Both sites displayed a moderate to low-quality sward dominated by *Holcus lanatus* and *Agrostis spp.* Wetland species, such as *Equisetum palustre*, *Juncus effusus* and *Juncus articulatus*, were present in the shallow-drained Site A_s while Site B was characterised by a greater cover of *Rumex acetosa* and *Cirsium arvense*. While the moss layer was almost non-existent in Site B, Site A_d and A_s had an average ground cover of 20% and 60% respectively, composed solely of *Rhytidiadelphus squarrosus*. Experimental plots were fenced off from cattle for the duration of the study. To simulate local activities, the vegetation within and surrounding each collar was cut to a stubble height of 5 cm in order to mimic the cow-grazing regime, so that the height of the vegetation was always comparable to that in the grazed fields. The interval between cuttings (considered as grazing events) was not strict but rather followed the concomitant status of the vegetation in the fields. The number of cuttings also varied each year reflecting the

biomass production. The vegetation was collected from each plot and oven-dried to a constant mass at 75°C. Carbon and N contents were measured (CE440 Exeter Elemental Analyser) using a pooled, homogenised sample of all biomass cut from each plot from the first growing season (Table 3.3). Vegetation height (cm) and leaf area index (LAI) was measured regularly throughout the year and systematically during GHG measurements and before cutting events (see Renou-Wilson *et al.* 2014 for LAI and vegetation height regression analysis used for modelling).

Table 3.3. Biomass chemical characteristics for each site.

	Site A _d	Site A _s	Site B
C %	42.9	42.5	43.8
N %	2.04	1.95	3.30
P %	0.13	0.10	0.20
K %	1.25	1.25	1.07

4 Greenhouse Gas Exchange in Drained Organic Soils under Grassland

4.1 State of Knowledge

4.1.1 On-site CO₂ Emissions

Carbon dioxide is fixed by the vegetation during photosynthesis (GPP) and released during ecosystem respiration (R_{ECO}). Ecosystem respiration is a combination of two processes – autotrophic (vegetation derived) and heterotrophic (microbial derived) respiration. The relative contribution of either to R_{ECO} is strongly controlled by the position of the water table. In contrast to drained mineral soils, decomposition of the organic matter in drained organic soils is likely to be higher than CO₂ uptake by the vegetation and Net Ecosystem Exchange ($\text{NEE} = \text{GPP} - R_{\text{ECO}}$). The NEE is considered the largest component in the surface-atmosphere exchange of GHG and is the starting point in the construction of the more complete GHG balance and ultimately NECB. To date, there have been no published C flux studies from drained organic soils under grassland in the Republic of Ireland. However, some insight into C gaseous exchange on organic soils can be gleaned from studies on drained industrial cutaway peat soils. Emissions of CO₂-C from these highly organic soils have been reported to be in the range of 1.9 to 8 tonnes C ha⁻¹ yr⁻¹ (Wilson and Farrell 2007, Wilson *et al.* 2007) with water table depth and soil temperature the main drivers of heterotrophic respiration. Net ecosystem exchange studies elsewhere have shown that grasslands on organic soils are likely to be a net CO₂ source with values ranging from 2.2 to 7.5 tonnes C ha⁻¹ yr⁻¹ across both boreal and temperate climatic zone (e.g. Nieveen *et al.* 2005, Jacobs *et al.* 2007, Maljanen *et al.* 2007, Lohila 2008). A review by Oleszczuk *et al.* (2008) concluded that nutrient-poor and nutrient-rich organic soils used for grassland emit about the same: a median of 4.6 tonnes C ha⁻¹ yr⁻¹. However, more studies have emerged showing that nutrient-rich and nutrient-poor grassland behave significantly differently with less CO₂ emissions likely from nutrient-poor organic soils (Kuntze 1992, Drösler *et al.* 2013). The nutrient status is defined in the 2006 Guidelines where ombrogenic organic soils are characterised as nutrient poor, while

minerogenic organic soils are characterised as nutrient rich. However, the water table level has also been found to be a significant factor affecting NEE (Jacobs *et al.* 2003, Lloyd 2006, Drösler *et al.* 2013), warranting a further stratification in the Wetland Supplement where data was sufficient. Drainage class is defined as the 'mean annual water table averaged over a period of several years' with shallow drained being less than 30 cm below the surface and deep drained being 30 cm or deeper below the surface.

4.1.2 CH₄ and N₂O Emissions and Removals

In order to give a complete account of the GHG balance of drained organic soils under grassland, it is necessary to consider not only CO₂ but also non-CO₂ GHG such as CH₄ and N₂O. Drainage of organic soils affects many environmental parameters which, in turn, can increase plant root respiration and reduce CH₄ emissions (except for ditches). Methane emissions were originally thought to be totally suppressed after drainage and this is based on many studies which have shown that in well-drained organic soils CH₄ emissions are low as full CH₄ oxidation is expected in the surface layers (Nykänen *et al.* 1995, Strack *et al.* 2004, Kasimir Klemedtsson *et al.* 2009) and, in some cases, CH₄ uptake in the soil may occur (Maljanen *et al.* 2007, Wilson *et al.* 2007). However, the persistence of certain vegetation such as aerenchymous species (e.g. *Juncus effusus*) in poorly drained grassland over organic soil may still lead to the transfer of methane from the soil profile to the atmosphere. In addition, new studies have confirmed previous data (Best and Jacobs 1997, Minkinen and Laine 2006), showing that CH₄ emissions can occur from the drained land surface as well as from the ditches (Schrier-Uijl *et al.* 2010, Teh *et al.* 2011, Vermaat *et al.* 2011, Hyvönen *et al.* 2013).

In drained agricultural organic soils, N₂O emissions, which are the result of either nitrification or denitrification activities, have been shown to be significant (Regina *et al.* 2004, Pihlatie 2007), and emissions of both N₂O and CO₂ may be higher

in nutrient-rich than in nutrient-poor organic soils (Kasimir-Klemmedtsson *et al.* 1997, Maljanen *et al.* 2003, Schils *et al.* 2008). Nitrous oxide emissions are caused by nitrogen mineralisation associated with organic matter decomposition. There is also a greater risk of N₂O emissions under increasing acid conditions as the N₂O:N₂ product ratio increases with decreasing soil pH (Simek and Cooper 2002). When fertiliser or organic amendments are applied to the grassland, this causes an additional source of anthropogenic N₂O which is measured in tandem with N mineralisation from the organic matter decomposition. However, the latter emissions are reported separately within the IPCC inventory context (see [Box 4.1](#)).

Box 4.1. Note on terms used in this report

CO₂-C represents the carbon atom contained within the CO₂ molecule. In terms of the overall molecular weight of CO₂, the carbon atom accounts for 12/44 or 27%. Thus, a multiplier of 3.667 is required in order to convert CO₂-C values to CO₂.

CH₄-C represents the carbon atom contained within the CH₄ molecule. In terms of the overall molecular weight of CH₄, the carbon atom accounts for 12/16 or 75%. Thus, a multiplier of 1.334 is required in order to convert CH₄-C values to CH₄.

N₂O-N represents the nitrogen atoms contained within the N₂O molecule. In terms of the overall molecular weight of N₂O, the nitrogen atom accounts for 28/44 or 64%. Thus, a multiplier of 1.571 is required in order to convert N₂O-N values to N₂O.

Negative gas flux values indicate an uptake by the peatland and positive gas flux values indicate a loss from the peatland to the atmosphere.

4.2 Methodology for Measuring Greenhouse Gas

Twelve permanent sample plots were established systematically within Site A (seven in the deep-drained Site A^d, and five in the shallow-drained Site A^s) and nine sample plots at Site B (at least 15 m from field border or river). Each sample plot consisted of a stainless steel collar (60 x 60 cm) that was inserted

to a depth of 20 cm into the soil before the start of the study. Data loggers and a weather station were established at each study site and recorded hourly soil temperatures (°C) at 5, 10 and 20 cm depths as well as recorded photosynthetic photon flux density (PPFD, μmol m⁻² s⁻¹) and volumetric soil moisture (%; Site B only). Perforated PVC pipes (internal diameter: 2 cm) were inserted adjacent to each sample plot to measure water table position (WT). Wooden boardwalks were built around the sample plots to minimise damage to the vegetation and to avoid compression of the peat during gas sampling. Carbon dioxide, CH₄ and N₂O fluxes were measured using the static chamber method (Alm *et al.* 2007) biweekly during the growing season and monthly during the winter time. A full description of the methodology for the measurement and modelling of the data can be found in Renou-Wilson *et al.* (2014). Site A was monitored starting April 2011 (first two years are reported here) while Site B was only monitored for one year (April 2011 to March 2012) with no fertiliser application taking place during that time. Additional N₂O measurements were taken at Site B in April–June 2012 during an intensive campaign after a typical fertilisation event of 25 kg N/ha.

4.3 Results and Discussion

4.3.1 Weather Conditions

In Year 1 (starting April 2011), Site A displayed higher precipitation (1211 mm) as well as a higher annual mean air temperature (10.2°C) compared to Site B (1003 mm; 9.6°C) ([Table 3.1](#)). Rainfall and temperature patterns were similar, with both sites having higher mean annual precipitation (7–12%) and air temperature (5–10%) compared to the 30-year average values ([Table 3.1](#)). Both sites also had cooler summer but warmer autumn and winter temperatures compared to the 30-year monthly averages. There was no significant difference between monthly mean soil temperatures between all the sites ($p=0.801$) ([Fig. 4.1](#)). In Year 2, Site A received slightly lower precipitation to Year 1 (1193 mm) but it remained above the long-term average and was driven by high values during June and July, September and October and December and January. Both the mean annual and monthly air temperatures were consistently below the long-term averages (by 8–9%).

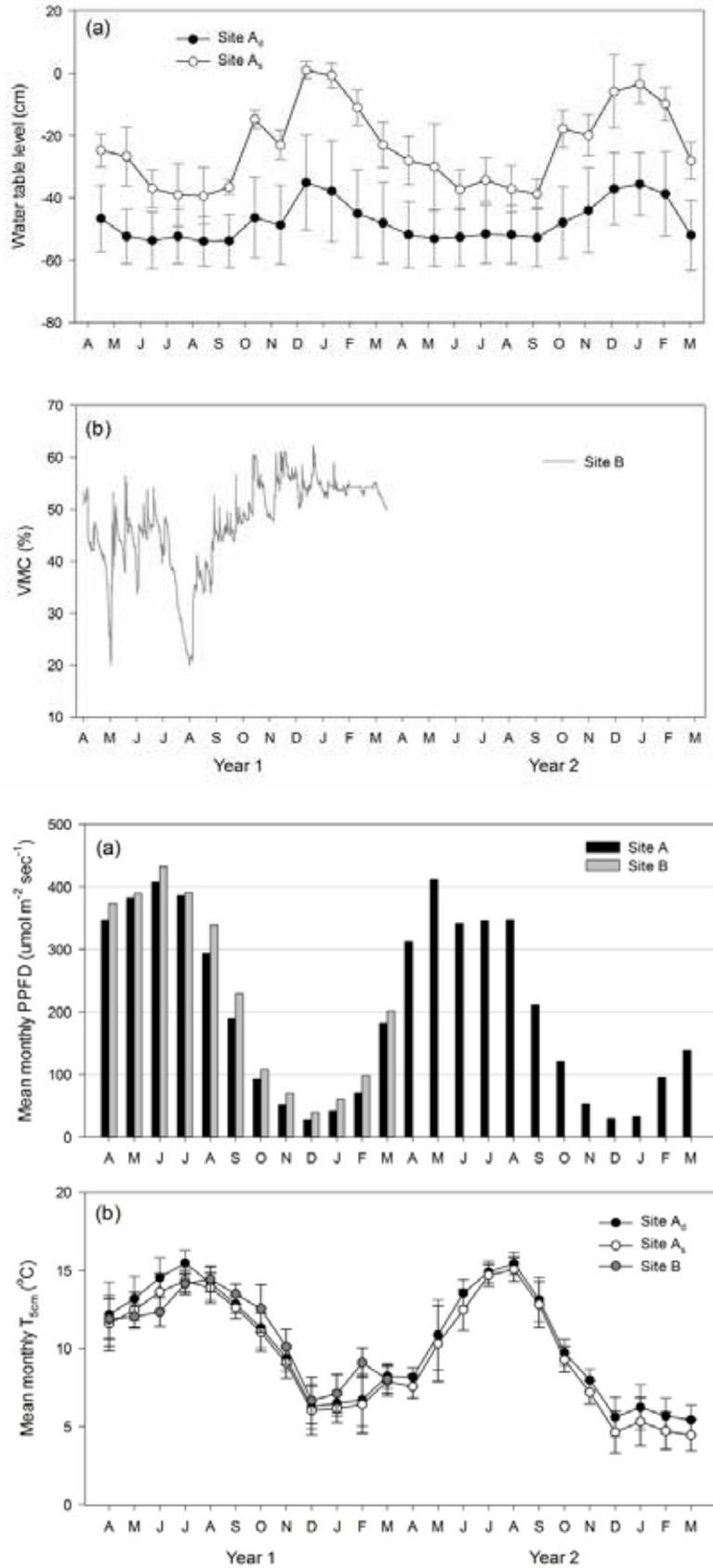


Fig. 4.1. Mean monthly water table levels, volumetric moisture content (VMC), photosynthetic photon flux density (PPFD) and soil temperature at 5 cm deep recorded during the monitoring period at the study sites (Year 1 starting April 2011).

There was a typical seasonal trend in mean monthly PPFD values at both locations (Fig. 4.1) associated with day length and seasonal cloud cover, with the highest values observed in summer (June to August) and lowest in the winter (December to February). Annual and mean monthly PPFD values were significantly higher ($p < 0.001$) at Site B than Site A during Year 1. At Site A, mean PPFD was significantly higher in Year 1 than Year 2, with greater values in eight months of the year. The two sites therefore contrasted climatically but displayed similar climatic change trends as predicted by Sweeney *et al.* (2008).

Seasonality of precipitation was reflected in WT depth at Site A and soil moisture content at Site B (Fig. 4.1). Despite rising slightly above -40 cm in winter, the monthly WT site averages at the deep-drained Site A_d were relatively stable and similar annual means were recorded in both years (-47.8 and -47.4 cm). Monthly means were significantly different between Sites A_d and A_s ($p < 0.001$), the latter being characterised by seasonal fluctuations with monthly WT below -30 cm during late summer/early autumn periods, rising quickly to levels above -10 cm during winter periods. Despite this, the annual WT mean at the shallow-drained Site A_s was similar between years at -22.9 cm in Year 1 and -24.2 cm in Year 2. At Site B, volumetric moisture content

(VMC) averaged 47.5% and ranged from 20 to 62% with two significantly drier periods in May and August in Year 1.

4.4 Biomass Production/Export

In Year 1, the cumulative biomass production was superior at Site B (351 g C m^{-2}), with Site A_d and A_s 35% and 50% less productive respectively (Table 4.1). Biomass production in Site B started to increase sooner than Site A and was consistently higher, peaking during the warmest months (July–August) (Fig. 4.2). Biomass production also lasted for a longer period at Site B allowing for biomass removal to take place as late as October (eight cutting events in total). Productivity was lower throughout Year 2 with poor growth during late summer months a particular feature. Lower temperatures throughout the second growing season was likely the driver behind the decrease in productivity with biomass production 18% lower at Site A_d (from 225 to 185 g C m^{-2}) and 21% at Site A_s (from 174 to 137 g C m^{-2}) (Table 4.1). The nitrogen content of the biomass was similar at both Site A_d and A_s (2%) but higher at Site B (3.3%). Nitrogen export in biomass followed C export trends with the nutrient-rich Site B losing the highest amount

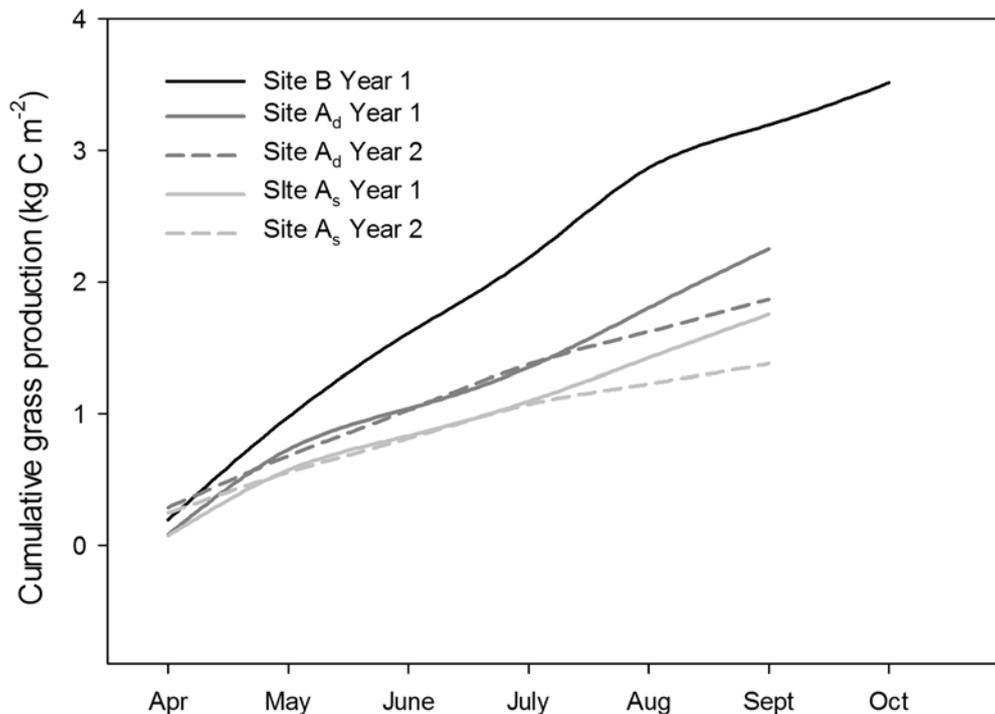


Figure 4.2. Cumulative biomass production (kg C m^{-2}) at all study sites. See Table 3.3 for carbon (C) content of biomass.

at 265 (27) kg N ha⁻¹ yr⁻¹ compared to 107 (45) and 80 (12) kg N ha⁻¹ yr⁻¹ in Site A_d and Site A_s. Nitrogen biomass exports were 17 to 19% lower in Year 2 in Site A_d and A_s respectively.

4.4.1 CO₂ Fluxes

Annual NEE differed significantly at each site (Table 4.1) with the nutrient-poor shallow-drained site (A_s) a net annual sink of CO₂-C in both years of the study (-89 g and -99 g CO₂-C m⁻² yr⁻¹ in Years 1 and 2 respectively) and the nutrient-poor deep-drained site (A_d) a net source of 85 g CO₂-C m⁻² yr⁻¹ in Year 1 and a net sink of CO₂-C in the second year (-26 CO₂-C m⁻² yr⁻¹). In contrast, the nutrient-rich drained Site B was a net annual source of 233 g CO₂-C m⁻² yr⁻¹. This NEE value is at the lower range of data reviewed from temperate and boreal studies (e.g. Maljanen *et al.* 2010, Elsgaard *et al.* 2012, Leiber-Sauheitl *et al.* 2013). Annual losses of CO₂ from our nutrient-rich drained site B were more in line with those determined for low-intensity Dutch permanent grasslands on organic soils (Jacobs *et al.* 2007, Veenendaal *et al.* 2007) and for extensive wet nutrient-poor grasslands in Germany (Drösler *et al.* 2013). This corroborates that Site B is a low-intensity grassland (no cultivation and low inputs). The importance of a 'grassland type' appears even more pronounced in the case of nutrient-poor sites as investigated in this study. Net annual uptake of CO₂-C in both the deep-drained and shallow-drained nutrient-poor sites in Year 2 of our investigation contrast significantly with studies from similar site types in temperate climatic regions where a net annual loss of CO₂-C has been reported (Drösler *et al.* 2013, Leiber-Sauheitl *et al.* 2013). The NEE values for the nutrient-poor shallow-drained sites in this study represent similar sinks (small source in some year) to those reported by Skiba *et al.* (2013) for grazed drained nutrient-poor organic soils (moorland) in a maritime temperate climate. Our results would therefore point to the significance of edaphic conditions as well as the low-intensity management system encountered at our sites. An indication of such influences can be found in the contrasting GPP and R_{eco} parameters found at all sites (which are fully discussed in Renou-Wilson *et al.* 2014).

4.5 Methane Fluxes

Fluxes of CH₄ were not detectable at the deep-drained sites, namely Site A_d and Site B and were relatively low at Site A_s. A strong seasonal effect could be observed in both years at Site A_s. Methane fluxes followed a general trend where the highest values were observed in late summer and the lowest in winter time. However, high flux values for this site (0.9 and 1.4 mg CH₄ m⁻² hr⁻¹) were also observed in April and November. Spatial variation in emissions at Site A_s was very evident), with the higher emissions associated with plots that contained *Juncus effusus*. No statistical relationship was observed between CH₄ fluxes and any of the environmental parameters. Therefore, in order to calculate an annual CH₄-C balance, linear interpolation between the observed fluxes was performed and the values integrated over a 12-month period (Beetz *et al.* 2013). Annual CH₄-C emissions were not significantly 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 m⁻² yr⁻¹ in Year 2 (Table 4.1).

As CH₄ production is expected to occur mainly within the anoxic soil horizons, i.e. below the groundwater table, the absence of CH₄ emissions at Sites B and A_d are consistent firstly with annual mean water table depths (below -30 cm) and secondly with the absence of aerenchymous plant species that can transport CH₄ from the soil to the atmosphere (Couwenberg 2009). Emissions close to zero were also recorded from drained grasslands over peat in Germany (Drösler *et al.* 2013, Leiber-Sauheitl *et al.* 2013) and Denmark (Petersen *et al.* 2012).

The estimated mean annual CH₄ fluxes for the shallow-drained nutrient-poor Site A_s (18 ± 15 kg CH₄ ha⁻¹ yr⁻¹ over 2 years) are at the lower range of values reported for wet grasslands in Germany (Leiber-Sauheitl *et al.* 2013) and are only 10% of values reported for managed fen meadow and extensively managed grassland on peat in the Netherlands (Kroon *et al.* 2010, Schrier-Uijl *et al.* 2010). The low or even absence of CH₄ fluxes may also be due to the relatively shallow peat depth (<1 m) as suggested by Levy *et al.* (2012) and the fact that these sites have been drained for a very long time (Flasse

et al. 1998). The poor quality and supply of substrates may also restrict methanogenesis (Couwenberg 2009). Given that a recent meta-analysis has demonstrated that vegetation is the strongest explanatory parameter for CH₄ flux variability (Levy *et al.* 2012), it is likely that the higher CH₄ fluxes measured from the shallow-drained nutrient-poor sites were also a product of the presence of *Juncus effusus* (aerenchymous species) as seen in other studies (Petersen *et al.* 2012, Herbst *et al.* 2013). Overall, our results demonstrate that CH₄ fluxes were negligible, if not absent, components of the C balance of extensively drained grasslands over organic soils in a maritime temperate climate, emitted only in very small amounts when the mean annual water table was around -23 cm.

4.6 Nitrous Oxide Fluxes

Fluxes of N₂O were not distinguishable from zero at both Site A_s and A_o during the monitoring period. The absence of N₂O emissions from the nutrient-poor sites was consistent with the lack of artificial fertiliser use for several decades at these sites and similar results have been reported from an extensive drained moorland in Scotland (Skiba *et al.* 2013). At the nutrient-rich Site B, a seasonal trend was observable during the 1 April 2011 to 31 March 2012 period with the highest values (153 µg N₂O m⁻² hr⁻¹) observed in October. Nitrous oxide uptake (~14 µg N₂O m⁻² hr⁻¹) was observed during May. Following a fertilisation event at Site B in Year 2, N₂O emissions increased considerably and reached an observed maximum of 484 µg N₂O m⁻² hr⁻¹. As with CH₄ fluxes, no statistical relationship was observed between N₂O fluxes and any of the environmental parameters. Instead, linear interpolation between the observed fluxes was performed and the values

integrated over a 12-month period. (Note: The integration period was from 1 April 2011 to 31 March 2012 and, therefore, did not cover the fertilisation experiment.) Annual N₂O-N emissions for this period were estimated at 1.6 ± 0.34 kg N₂O-N ha⁻¹ yr⁻¹.

In nutrient-rich sites, the potential for N₂O emissions is much higher, and is additionally forced by fertiliser applications to these more productive systems. Estimated annual N₂O emissions from Site B are nonetheless still low in comparison to most grasslands situated over peat (closer to the IPCC default values for nutrient-rich shallow-drained soils). This study provides further evidence that N₂O fluxes are predominantly affected by the grassland management type, which is not represented in the categories within the IPCC guidelines. Most investigations are carried out in intensively managed grasslands. While nutrient rich, Site B is still very much 'extensive' given the low artificial N input. Our values are however comparable to Dutch figures reported by Langeveld *et al.* (1997) and van Beek *et al.* (2011) where grasslands had typically less fertiliser and manure inputs over time. Overall, our results are not sufficient to conclude firmly on the insignificance of N₂O fluxes in extensive grassland over organic soils, and further investigations in this area are warranted to ascertain the potential for large denitrification losses. Since N₂O uptakes were detected on occasion, N₂O reductase activity could be high and therefore large dinitrogen (N₂) emissions may be possible. This study also highlights the need for further research on intensive measurement of N₂O based on C:N ratios, which may not reflect soil organic C degradation as seen in the contrasting ecosystem respiration measurements from our two sites with similar C:N ratios.

Table 4.1. 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)

5 Fluvial Carbon Losses from Drained Organic Soils under Grassland

5.1 State of Knowledge

Recent studies have shown that aquatic C loss is the second largest component of the C budget at a catchment level (Worrall *et al.* 2009a, Dinsmore *et al.* 2010). The significance of fluvial C losses has been acknowledged and EFs for DOC are now included in the IPCC Wetland Supplement 2013.

Barry and Foy (2008) measured DOC in rivers draining the catchment of Lough Melvin, which is dominated by peat and humic gley soils and found that annual export rates of DOC averaged 170 kg C ha⁻¹, with loss rates for individual sub-catchments in excess of 240 kg C ha⁻¹. Soil H project shows that peatland catchment had by far the highest levels of DOC in stream-flow compared to catchments dominated by mineral soils with values reaching 150 kg C ha⁻¹ yr⁻¹ (Kiely *et al.* 2014).

5.2 Methodology

A complete methodology is described in Barry *et al.* (2015). In short, fluxes of DOC, POC and DIC were determined for each site for the hydrological year beginning October 2011. For Site A, a second year of monitoring covered the hydrological year commencing October 2012. Discharge measurement, laboratory analyses and C flux estimation methods employed are briefly described below and detailed in Barry *et al.* (2015).

In order to accurately estimate waterborne C fluxes representative of the area on which the terrestrial GHG fluxes were conducted, runoff measurements were confined to the smallest catchment areas encompassing the GHG study sites. Field drains adjacent to each of the study sites were selected and these drained relatively small areas of between 3 and 5 ha. Critically, soils and land use within these catchments was uniform and consistent with that of the GHG study sites. Although areas outside of the study sites yielded water, this approach has advantages in that sampling is close to source, limiting the residence time between exports from the soil environment and thereby lessening the impact of processes that can alter

aqueous fluxes (e.g. remineralisation and atmospheric evasion of CO₂). Continuous-level recorders (Orpheus mini, OTT Germany) and flow-gauging instrumentation were installed at each site. Carbon concentrations were determined at fortnightly to monthly intervals throughout each monitoring period. Dissolved organic carbon concentration was measured by high temperature Pt-catalysed combustion on glass-fibre filtered samples (Whatman GF/C; 1.7µm), POC concentrations by loss on ignition of samples filtered onto glass-fibre filters, and DIC according to Stumm and Morgan (1996), from alkalinity (gran titration), and *in situ* measurements of pH, temperature and specific conductance. Annual C loads were calculated as the product of annual or seasonal flow-weighted mean concentrations and runoff volumes, corresponding to 'method 5' given by Walling and Webb (1985), and DOC rainwater inputs were subtracted based on concentrations determined for rainwater during the study. Fluxes are expressed as areal rates according to catchment area. Ninety-five percent confidence limits for flux estimates were calculated according to Hope *et al.* (1997).

As DIC reflects both CO₂ derived from soil respiration and C derived from the atmosphere and rock weathering, total DIC exports were apportioned to reflect terrestrial organic matter dynamics alone (Stumm and Morgan 1996, Telmer and Veizer 1999). All CO₂ above atmospheric equilibrium was considered derived from respiration of organic matter. For Site B, where base geology is limestone, up to half of the HCO₃⁻ C export can be derived from soil organic matter whereas for Site A the geology (Precambrian Gneiss and Schist) and alkalinity imply most bicarbonate C derives from organic matter mineralisation.

5.3 Results and Discussion

Fluvial exports were measured at the field scale at Site A and so encompass both sites A_o 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 respectively. Dissolved organic carbon export rates were 18.1 g C m⁻² yr⁻¹ in Year 1 and 4.9 g C m⁻² yr⁻¹ in Year 2, representing 41% and 16% of the fluvial

C flux in each year respectively (Table 4.1). The mean drainage water $p\text{CO}_2$ in Years 1 and 2 respectively was 102 and 146 times in excess of atmospheric equilibrium concentrations, and consequently the export of excess CO_2 (above equilibrium) was substantial at 13.4 and 18.5 $\text{g C m}^{-2} \text{yr}^{-1}$ respectively. Excess CO_2 represented 30 and 60% of the fluvial C flux each year and dominated the export of DIC (excess $\text{CO}_2 + \text{HCO}_3^-$) which overall comprised 44% and 78% of the fluvial C flux in each year. As with DOC, POC fluxes were also lower in Year 2, declining from 6.4 $\text{g C m}^{-2} \text{yr}^{-1}$ to 2.0 $\text{g C m}^{-2} \text{yr}^{-1}$.

The fluvial C flux was greater at Site B at 68.9 $\text{g C m}^{-2} \text{yr}^{-1}$, comprising 54% DOC, 35% DIC and 11% POC. By comparison to Site A, this total flux largely reflected a greater export of DOC, particularly relative to the second year at site A. Despite differences in base geology between sites, the flux of DIC (estimated as derived from soil respiratory processes) was comparable to both years at site A, but differed in that it was primarily in the form of bicarbonate (29% total flux), implying that Site B yielded a greater carbonate dissolution but had lower evasion losses due to buffering geology at this site.

Studies have shown that fluvial C losses can be large components of terrestrial C balances at field and landscape scales, particularly where organic soils predominate (Worrall *et al.* 2009a, Dinsmore *et al.* 2010, Buffam *et al.* 2011). Furthermore, the importance of this component can be magnified by management practices given that large increases in DOC export can occur following drainage of organic soils, related to greater DOC production under aerobic conditions and soil redox conditions during wet and dry periods (Wallage *et al.* 2006, Clark *et al.* 2009, Urbanová *et al.* 2011). In this study, DOC fluxes were greatest for the drained nutrient-rich site but lower than values reported for drained peatland catchments in the same climatic region (Gibson *et al.* 2009, Rowson *et al.* 2010). More significantly, the 2-year mean DOC flux from the nutrient-poor site (11.5 $\text{g C m}^{-2} \text{yr}^{-1}$) is at the low end of values reported for temperate semi-natural peatland

catchments in Ireland and the UK (Dawson *et al.* 2004, Dinsmore *et al.* 2010, Koehler *et al.* 2011, Kiely *et al.* 2014). This suggests that DOC fluxes from nutrient-poor organic soils under extensive grasslands in Ireland may be more similar to semi-natural ecosystems. Annual DOC fluxes from the nutrient-poor site were almost four times lower in Year 2, reaching a low value of 4.9 $\text{g C m}^{-2} \text{yr}^{-1}$. It is likely that inter-annual variability in DOC flux at this site is caused by reduced precipitation and greater evapotranspiration during the second hydrological year (runoff 897 mm and 745 mm in Years 1 and 2 respectively). However, the decline in DOC export from Year 1 to 2 was partially offset by a greater flux of aqueous CO_2 export, implying that drier conditions were more conducive to soil OM and soil pore water OM mineralisation. In both Years 1 and 2, the flux of DIC (derived from soil respiration) exceeded the flux of DOC, and was predominantly in the form of CO_2 . The high $p\text{CO}_2$ of samples implies that accurately accounting for aqueous CO_2 fluxes requires that samples are taken close to source to account for C that is otherwise rapidly evaded to the atmosphere, particularly with waters with low alkalinity. While the literature on aqueous inorganic C fluxes from agricultural catchments is not extensive, our observations are consistent with findings for small boreal streams where exports of CO_2 were also substantial and atmospheric evasion of CO_2 from streams was equivalent to approximately half of the total stream C flux (Wallin *et al.* 2013). Our findings also provide support for arguments for greater incorporation of stream C fluxes to improve regional and global C accounting (e.g. Downing *et al.* 2012). At Site B the DIC flux was also considerable, but comprised a lower fraction of the total. Notably, this DIC export was mainly in the form of bicarbonate so that the soil respiratory origins of much of this fraction are indicated to behave more conservatively during downstream transport but nevertheless remain an important part of the terrestrial C balance. These results support an argument for the requirement to report other fluvial C components in GHG inventories, notably DIC and POC (e.g. Hope *et al.* 2001, Dinsmore *et al.* 2010).

6 Dissolved Organic Matter Bioavailability

6.1 State of Knowledge and Methodology

A full understanding of the fate of DOC lost by aquatic pathways is currently poor. As an operationally defined variable, DOC can consist of a multitude of organic components with differing bioavailabilities. While most of the world's lakes are heterotrophic (they produce more CO₂ than they consume), humic substances have a generally low bioavailability so that much can either be trapped in lake sediments via flocculation or adsorption onto suspended sediments or move downstream to the sea without being fully used as a microbial substrate. Nevertheless, organic matter exported from the landscape remains available for biochemical processing, remineralisation and subsequent efflux to the atmosphere as CO₂ and CH₄, but also permanent burial in sediments. Thus, proportions of the exported organic matter may constitute atmospheric sources and sinks over variable timescales, depending on its molecular composition and physico-chemical environment. The varying environmental influences on organic-matter reactivity make accurate assessments of its fate complicated. However, the implication is that most of the exported organic matter can be remineralised, but a large proportion may constitute a net C sink if delivered to environments such as lake sediments that are not conducive to oxidation.

A study from Lough Melvin, Co. Fermanagh-Leitrim where soils consist largely of humic rankers and blanket peat, showed that sediment carbon burial retained approximately 7% of the fluvial C input to the lake, and that C evasion exceeds sediment burial by a factor of 4 (Barry and Foy 2008). Therefore, while quantifying exported C from peatland soils is important, quantifying its bioavailability is critical if its contribution to CO₂ emissions downstream of the organic soil is to be assessed.

The biological oxygen demand (BOD) of filtered (DOC) and unfiltered samples (TOC; DOC+POC) was measured manometrically using the Oxitop® system (WTW, Xylem, USA). This approach measures oxygen consumption over time derived from the gas pressure change in the sample bottle, reflecting aerobic respiration. Samples were set to run for 90 days with

measurements logged at 6-hour intervals. All samples reached asymptotic values within this period. In all cases a sufficient excess of oxygen was present to meet respiratory demands. Biological oxygen demand values measured over time were converted to C units using a respiratory quotient (RQ) of 1. The labile component of organic matter exports was assessed as the proportion of the total C content of each sample (DOC+POC) remineralised during the incubations (Barry *et al.*, 2015).

6.2 Results and Discussion

At the nutrient-poor Site A, the percent of carbon content remineralised (labile C) from the drain samples was broadly similar for DOC and TOC, ranging between 10 and 26%. A much lower proportion of the organic content of river samples was remineralised (c.10–15%). The amount remineralised was somewhat greater on average for DOC than TOC for the river samples, and in view of the results for the drain samples the implication is that the bioavailable POC fraction is rapidly remineralised after export to streams.

At the nutrient-rich Site B, the labile DOC fraction of drain samples ranged between 20 and 25% and was therefore constrained to a much narrower range than for the nutrient-poor site: however, this may partly reflect the lower sample number. The labile fraction of drain TOC was also constrained to c. 10%, the lower percentage relative to that of DOC, suggesting a lower lability of the particulate C fraction. The labile DOC content of river samples was greater than that of the drain samples at 13% and 33% at the 10th and 90th sample percentiles respectively. Labile TOC for the river samples was greater than for the drain samples and ranged from 20–48%. The river catchment is large and so probably has multiple organic matter sources compared to the drain. Some of these sources may be anthropogenic, from waste-water discharges, agriculture or C losses associated with peat exploitation and these may drive the elevated labile C fraction of the river samples compared to the drain. There is also a possibility of enhanced bioavailability due to greater UV exposure.

Overall, the mean bioavailable fraction of organic carbon in all samples was 19.5% and ranged from 6.8% to 53.1%. Expressed on a catchment area basis, remineralisation of organic C at Site A was equivalent to 4.5–4.9 g C m⁻² yr⁻¹, and at Site B from 5.5–9.4 g C m⁻² yr⁻¹. While the range of bioavailable fractions were similar at both sites the greater areal rates for site B

reflect the greater export of organic C from this site. The labile fractions of DOC were low yet variable, but within the ranges reported for similar studies, indicating small fractions of 5–10% remineralised over the typical time courses of downstream transit of days to weeks. These implications for the terrestrial C balances are further discussed in Barry *et al.* (2015).

7 Net Ecosystem Carbon Budget of Drained Organic Soils under Grassland

7.1 Methodology

The net ecosystem carbon budget (NECB) is positive when more carbon is lost than gained and negative when more carbon is gained than lost: i.e. the ecosystem has accumulated carbon. European grasslands used for livestock grazing and forage production are usually considered to be a carbon sink in terms of net biome productivity (NBP). When calculating the GHG balance, including off-site emissions of CO₂ and CH₄ as a result of the digesting and enteric fermentation by cattle of the cut herbage, the grasslands are considered neutral (e.g. Soussana *et al.* 2004, Soussana *et al.* 2007). This neutral balance was also true for a fertilised grassland on a peaty podzol to brown podzol site in Ireland (Leahy *et al.* 2004) where a strong C uptake was counteracted by N₂O emissions from fertilisation and emissions from ruminants. However, the GHG balance may be substantially reduced or become a large source in less productive grasslands where C uptakes are small (Byrne *et al.* 2005). In addition, none of these balances account for fluvial carbon losses which would almost certainly tilt the final NECB to becoming a source. Annual mean GHG balance in boreal grassland over peat soil has been estimated to be a large source (2260 CO₂ eq. m⁻²), similar to other managed peatlands (Maljanen *et al.* 2010). Even such a balance does not include CH₄ emissions from ditches which form a part of the drainage network and could contribute significantly to the total GHG budget.

The full NECB for this grassland on organic soil is composed of the gaseous fluxes of CO₂ and CH₄, as well as fluvial C losses in the form of DOC, POC, DIC and dissolved CO₂ and CH₄ and C emissions associated with livestock grazing (enteric fermentation and on-site manure deposits). The following nationally derived emission rates for CH₄ from enteric fermentation were employed as detailed by O'Mara *et al.* (2007): 74 kg CH₄ head⁻¹ yr⁻¹ for suckler cows; 22.4 kg CH₄ head⁻¹ yr⁻¹ for calves; 8 kg CH₄ head⁻¹ yr⁻¹ for sheep. Combining these rates with contemporary

stocking rates at the sites yielded further emissions of 3.6 g C m⁻² yr⁻¹ for Site A and 9.5 g C m⁻² yr⁻¹ for Site B. Given the very low-intensity farming systems at both sites, we assume the balance of C import/export from on-site manure deposits negligible (there was no fertilisation at Site B in the first year).

7.2 Complete Net Ecosystem Carbon Budget

Concurrent GHG and fluvial C balance studies on organic soils have been undertaken on only a few sites worldwide and this is the first full NECB of a temperate drained organic soil under grassland. At the nutrient-rich site, NECB was relatively high (663 ± 42 g C m⁻² yr⁻¹) considering that the grassland is extensive and this value is greater than those reported for intensive grassland on peat in both Germany (Beetz *et al.* 2013) and the Netherlands (Veenendaal *et al.* 2007). However, the value falls within the range of annual emission rates of 410–760 g C m⁻² yr⁻¹ reported by Couwenberg (2011), who reviewed several temperate drained peat soils under grassland. It should be noted, however, that the values above do not include fluvial C losses, which in this study accounted for 11% of the total NECB for the nutrient-rich site. While annual biomass exports were at the low end of literature values (e.g. 317–515 g C m⁻² yr⁻¹) (Veenendaal *et al.* 2007, Beetz *et al.* 2013), they accounted for 53% of the NECB (Table 4.1). The two-year mean NECB of the deep and shallow-drained nutrient-poor sites were considerably smaller than the nutrient-rich site (40% and 16% respectively). Biomass exports were also the biggest components of the NECB at these sites (Table 4.1) and were sufficient to influence the source-sink balance of the shallow-drained site in both years and at the deep-drained site in Year 2 (i.e. shift from a NEE sink to a net source of C). Given the impact of the biomass removal on NECB, controls on grazing regimes could therefore significantly influence the C sink capacity of these ecosystems, as documented for an abandoned former pasture over peat in the Netherlands (Hendricks *et al.*

2007). The NECB of the shallow-drained nutrient-poor site is indeed consistent with values reported for an extensive grassland in Germany, which was re-wetted four years previously (Beetz *et al.* 2013).

However, in the case of nutrient-poor organic soils, fluvial C losses were not a negligible component of the overall NECB and could easily influence the final C balance of these ecosystems.

8 Global Warming Potential

The GHG budgets and associated GWP were calculated using IPCC methodologies (2007). The GWP of the nutrient-rich drained organic soil was dominated by N₂O and reached 26.7 t CO₂eq ha⁻¹ yr⁻¹ which is very similar to results (2-year mean of 26 t CO₂eq ha⁻¹ yr⁻¹) from an intensive grassland on peat in Scotland (Beetz *et al.* 2013). These values are however in the lower range of organic soils under grasslands reported by Petersen in Denmark (15–62 CO₂eq ha⁻¹ yr⁻¹) and Kasimir-Klemedtsson in Sweden (10–70 CO₂eq ha⁻¹ yr⁻¹) while being on a par with German sites (12–31 CO₂eq ha⁻¹ yr⁻¹) (Leiber-Sauheitl *et al.* 2013). Using Tier 1 default N₂O values, the calculated GHG budget for Site B would have been more than triple that of the budget measured in this study. A large error in the calculation of GHG budget from these site types is associated with N₂O fluxes and this is confirmed by the large

variability in the quantitative estimates of N₂O emissions in studies published across ‘temperate’ Europe.

At the nutrient-poor sites, the annual estimates of GWP varied from a small warming effect (2-year average of 2.3 CO₂eq ha⁻¹ yr⁻¹) at the deep-drained site to a net cooling effect at the shallow-drained site (-1.8 CO₂eq ha⁻¹ yr⁻¹) and were dominated by CO₂ exchange (Table 4.1). Our annual estimates of GWP are in line with figures presented by Beetz *et al.* (2013) for both an extensive and rewetted grassland over organic soils in Germany. The authors warn of high variability of those estimates depending on the period of measurements used. Although we report two years of measurements in this report, long-term monitoring would allow for more robust multi-year average GHG budgets, which could then be extrapolated to the regional scale.

9 Implications for Reporting: CO₂, CH₄, N₂O and DOC Emission Factors

Net ecosystem exchange added to the C losses from the removal of biomass (assuming instantaneous emission of cut biomass) and averaged over a multi-annual measuring period gives a best-possible estimate of emissions/removals and therefore the most accurate CO₂-C EF (Couwenberg 2011, Elsgaard *et al.* 2012). The EF for the nutrient-rich deep-drained grassland (Site B) was 5.84 t CO₂-C ha⁻¹ yr⁻¹ which is in the range of values given in the 2013 Wetlands Supplement IPCC guidance (Table 9.1): 6.1 (5.0–7.3) t CO₂-C ha⁻¹ yr⁻¹ for this land-use category. It corresponds well with Tier 2 country-specific EFs for permanent grassland over organic soils: 5 t CO₂-C ha⁻¹ yr⁻¹ in Germany (Federal Environmental Agency 2013), 5.19 t CO₂-C ha⁻¹ yr⁻¹ in the Netherlands (Netherlands Environmental Assessment Agency 2009) and 5.17 t CO₂-C ha⁻¹ yr⁻¹ in Denmark (National Environmental Research Institute 2013).

In contrast, our nutrient-poor sites produced much lower CO₂ EFs of +2.35 t CO₂-C ha⁻¹ yr⁻¹ in the deep drained and +0.62 t CO₂-C ha⁻¹ yr⁻¹ in the shallow drained (Table 9.2). These figures are outside the lower

range of EFs for temperate-drained nutrient-poor sites in the IPCC guidance (2013): 5.3 (3.7-6.9) t CO₂-C ha⁻¹ yr⁻¹. This default value is not further stratified between shallow and deep-drained grasslands and is based on only seven identified study sites, six of which are from Germany (Drösler *et al.* 2013). Nonetheless, the multi-site GHG investigation reported by Drösler *et al.* (2013) includes nutrient-poor sites (extensive grasslands), which were a CO₂-C sink (NEE minus biomass removal) in some years. In Denmark, an EF of 1.25 t CO₂-C ha⁻¹ yr⁻¹ is applied to non-fertilised permanent grassland. Along with the results presented here, this supports our premise highlighting considerable heterogeneity in the C balance of grasslands on organic soils within the European temperate zone. While the stratification of default CO₂ EFs according to nutrient and drainage status is indeed supported by this study, variability in the impact of these factors also reflects the historical and contemporary management practices, therefore advocating a progression towards the Tier 2 reporting level for countries with significant areas of organic soils under grassland.

Table 9.1. CO₂-C, CH₄-C, N₂O-N and DOC emissions/removals factors (tonnes ha⁻¹ yr⁻¹) from organic soils under grasslands in the temperate zone as per IPCC Wetlands Supplement (2014). Positive values indicate a flux from the peatland to the atmosphere. 95% confidence interval in brackets.

Grassland type	CO ₂ t C ha ⁻¹ yr ⁻¹	CH ₄ kg CH ₄ ha ⁻¹ yr ⁻¹	DOC t C ha ⁻¹ yr ⁻¹	N ₂ O-N kg N ha ⁻¹ yr ⁻¹
Nutrient poor				
Drained	+5.3 (3.7-6.9)	+1.8 (0.72-2.9)	+0.31 (0.19-0.46)	+4.3 (1.9-6.8)
Nutrient rich				
Deep drained	+6.1 (5-7.3)	+16 (2.4-29)	+0.31 (0.19-0.46)	+8.2 (4.9-11)
Shallow drained	+3.6 (1.8-5.4)	+39 (-2.9-81)	+0.31 (0.19-0.46)	+1.6 (0.56-2.7)

Table 9.2. CO₂-C, CH₄-C, N₂O-N and DOC emissions/removals factors (tonnes ha⁻¹ yr⁻¹) from our study sites (organic soils under grasslands in a maritime temperate climate). Positive values indicate a flux from the peatland to the atmosphere. 95% confidence interval in brackets.

Grassland type	CO ₂ t C ha ⁻¹ yr ⁻¹	CH ₄ kg CH ₄ ha ⁻¹ yr ⁻¹	DOC t C ha ⁻¹ yr ⁻¹	N ₂ O-N kg N ha ⁻¹ yr ⁻¹
Nutrient poor				
Deep drained	+2.35 (1.3-3.4)	0	+0.11 (0.09-0.13)	0
Shallow drained	+0.62 (0.3-0.9)	+18 (17-19)	+0.11 (0.09-0.13)	0
Nutrient rich				
Deep drained	+5.84	0	+0.38	+1.6 (1.3-1.9)

The EF for CH₄ at both the deep-drained sites, regardless of nutrient status, was zero and, for the shallow-drained nutrient-poor grassland, calculated at 18 kg CH₄ ha⁻¹ yr⁻¹ (Table 9.2). The absence of CH₄ fluxes from both our well-drained organic soils is at variance with the IPCC guidance which presents an EF of 16 kg CH₄ ha⁻¹ yr⁻¹ for nutrient rich and 1.8 kg CH₄ ha⁻¹ yr⁻¹ for nutrient poor (Table 9.1). This over-estimation could be compensated by using the EF for nutrient-poor drained organic soils in the case of shallow-drained nutrient-poor sites. If water table status is used as a primary factor, the only default EF provided is for shallow-drained nutrient rich at 39 kg CH₄ ha⁻¹ yr⁻¹ which would also lead to a large over-estimation of CH₄ fluxes.

The EF for N₂O of zero at the nutrient-poor site and 1.6 kg N ha⁻¹ yr⁻¹ at the nutrient-rich site are both very low compared to the default EF in the IPCC guidance (2013). Despite being subject to high uncertainty, our results suggest again the likely significance of management system, especially with regards to fertilisation and stocking regime, factors which are currently missing from Tier 1 methodology.

In order to calculate an EF for DOC, it is critical to account only for the portion of DOC that could be completely mineralised and re-emitted to the atmosphere. While lake sediments can be sites for significant permanent burial of terrestrially fixed C, in most river-lake catchments mineralisation far exceeds burial (Algesten *et al.* 2003), and indeed studies have shown that

most of the DOC flux from drained peat catchments can be mineralised (Kölher *et al.* 2002, Jones *et al.* 2013). Despite such observations, the oceans still receive considerable inputs of terrestrial organic matter (c. 0.4 Gt yr⁻¹ according to Richey 2004) although the very minor amounts of unambiguously terrestrial material identified in seawater and marine sediments imply that the vast majority is re-mineralised (Bauer and Bianchi 2011). As per the IPCC guidance (2013), a 90% fraction was used to calculate an overall EF for DOC of 22.1 t C ha⁻¹ yr⁻¹, being much lower at the nutrient-poor site (10.3 t C ha⁻¹ yr⁻¹) compared to the nutrient-rich site (33.9 t C ha⁻¹ yr⁻¹). The later value is within the range of the default EF for temperate drained organic soils: 31 (19–46) g C m⁻² yr⁻¹ (Table 9.2). Given the high inter-annual variability experienced at the nutrient-poor site, further long-term study would be warranted in order to support the development of a country-specific EF which would include a 'nutrient status' stratification.

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

10 Implications for Climate Change Mitigation Measures

10.1 Water Table Management as Mitigation Strategy

In the context of the new opportunity given by the IPCC Wetland Supplement (2013) to report rewetted drained organic soils, this study underpins the need to investigate the rewetting of high C content nutrient-rich organic soils under grassland in Ireland as a priority. The sustained high decomposition rates recorded at the nutrient-rich site as reported here could only be compensated by reducing microbial oxidation of peat via higher water table levels. Rewetting of these site types should therefore be a priority of any climate change mitigation strategy focusing on decreasing C losses from soils in Ireland. Furthermore, the neutral or C sink capacity of nutrient-poor organic soils may also be strongly affected by climate change. With higher predicted temperatures affecting peat decomposition, respiration rates might increase faster than photosynthetic assimilation due to nutrient limitation. Predicted higher precipitation for the studied region could however work in synergy with government-led actions to introduce agricultural schemes for less intensive grassland management or even complete restoration. The former would require the maintenance of a relatively high water table (above -25 cm) and the reduction of the grazing regime to actively promote C uptake by these ecosystems. However, Lloyd (2006) noted that whilst higher water table levels reduced C losses from respiration, there were difficulties in maintaining uniform levels throughout the field. Controlled drainage on cultivated peat soils has been somewhat successfully implemented in Finland although they concluded that it is a slow process (Merja Myllys and Kritina Regina, MTT Agrifood Research Finland, Jokioinen, pers. comm.). Another issue relates to the impact of bringing the water table closer to the surface on the vegetation. Vegetation will typically develop towards a new climax and these changes in vegetation communities play an important role in the CO₂ and CH₄ exchanges (Strack and Waddington 2007). An increasing proportion of wet species such as *Juncus* spp. and *Equisitum* spp. will change the productivity and therefore the overall C sink potential. This decline may be compensated by the absence of biomass removal but the dynamic of such vegetation composition may be

critical in further maintaining the water table levels and maintaining productivity (some species may strongly compete in order to 'maintain' ideal conditions for their growth. In addition, the aesthetic value of such land use may become critical as a field could quickly be colonised by tall rushes for example.

10.2 Socio-economic Prospects of Climate-friendly Strategies

An increasing number of scientific studies support the premise for the management of organic soils as a measure for mitigating GHG emissions from agricultural ecosystems (Smith *et al.* 2007). The most important mitigation practices are: (i) avoiding the drainage of these soils in the first place, (ii) re-establishing a high water table (Freibauer *et al.* 2004) or (iii) optimising the position of the water table (Lloyd 2006). However, the socio-economic realities need to be addressed fully in order to appraise these strategies. First, the relative importance of climate change versus land management on the GHG emissions needs to be determined. Very little is known about the efficacy and wider landscape effects of some proposed mitigating management techniques. Second, while organic soil restoration or rewetting have a host of biodiversity/environmental co-benefits, there are also direct economic implications for individual farm livelihoods under changed management practices (opportunity cost of crop production lost from this land). Such an economic impact will depend upon whether farmers receive payment for the GHG emission reduction. Studies elsewhere have shown that acceptance of mitigation measures requiring less intensive grassland management, for example, was very low. However, these studies involved mainly intensive, very profitable farms (Schaller *et al.* 2011). Therefore, the cost of these strategies was high in terms of re-organisation and farm adaptation and would have meant an intensification of production in alternative areas (pushing the climate issue around to another land through added fertiliser for example). In less profitable regions such as parts of Ireland where organic soils are typically found, financial compensation may be a good incentive as it would not require re-organisation, and subsidising farmers' losses would

be relatively easy to calculate. Similarly, environmental schemes such as Rural Environment Protection Scheme (REPS) have been successful in these areas where farmers can easily adopt a new management strategy without any significant financial costs. This proposal should be backed up by the social benefits brought by such climate-friendly strategy and which have yet to be monetised. Far from 'bearing' the cost of a climate-friendly strategy, farmers should be seen as delivering a sustainable agriculture which benefits everybody. Regional circumstances will be important in developing these strategies and further research into a full C budget, including fluvial losses (Höll *et al.* 2009) and associated cost and benefits of these management options (Worrall *et al.* 2009b) are key to developing efficient mitigation strategies.

10.3 Other Types of Drained Organic Soils and their Associated Carbon Balance

Organic soils have been reclaimed primarily for agriculture use but a large proportion has also been drained for forestry and peat extraction. In both cases the impact of drainage on C flows essentially consists of increased oxidation of the peat matrix, leading to increased CO₂ release to the atmosphere and to DOC in water outflows.

10.3.1 Under Cropland, including Forestry

In Ireland, some 292,930 ha of forests (43% of the total forest estate) are located on organic soils (Black *et al.* 2008). Lindsay (2010) has recently reviewed the much-discussed question of the C implications of forestry on peat bogs, and new figures show that the overall loss of C from an afforested peat bog may exceed the C gains made by the plantation forest within a 100-year time-frame. This would contradict older publications (Cannell *et al.* 1999, Hargreaves *et al.* 2003) and thus this topic is currently the subject of much forest-based research. Similarly, there are some concerns regarding the C benefit in transforming conifer forests on peat soils back to living bogs. This is a legitimate question given that the UNFCCC have made a major step towards wetland restoration with the IPCC officially working on guidance for calculating how much GHG is emitted from degraded carbon-rich peat soils (decision taken by the Subsidiary Body for Scientific and Technological Advice [SBSTA] in Bonn, Germany in June 2014). This translates to a broad acknowledgement among Parties under the

Convention that emissions from drained C-rich soils are significant and that rewetting or restoring is an important contribution to decreasing total emissions of GHGs. The critical question in the context of restoring forested peatlands is how does the restoration work impact on CH₄ and CO₂ exchange? In Ireland, 2,570 ha of Coillte peatlands have been restored under the LIFE programme with further commitment to restore more Western forested peatlands (Delaney 2008). While the restoration of the environmental conditions (i.e. water table, vegetation recolonisation) required to promote the return of the C sink function has received some attention in the past (Komulainen *et al.* 1998, Komulainen *et al.* 1999, Bortoluzzi *et al.* 2006), very little information on GHG dynamics after restoration is available in Ireland.

10.3.2 After-use of Industrial Cutaway Peatlands

The IPCC Wetlands Supplement impacts directly on the fate of the 86,878 ha of industrial cutaway peatlands currently in Bord na Móna ownership and the smaller proportion of cutaway owned by private horticultural companies. This could prove crucial for Ireland if these vast tracks could be restored and offset GHG emissions from drained bogs where peat is currently extracted (including turf-cutting). The new EF default value of 2.8 t C/ha given by the IPCC Wetlands Supplement corresponds to values recorded by Wilson *et al.* (2007) on Bord na Móna cutaway peatlands ranging from 2.5 to 3 t C ha⁻¹ yr⁻¹. This is much higher than previous EF and therefore will dramatically increase Ireland's contribution to GHG emissions from industrial cutaway and cutover. In the context of reporting these emissions, the default methodology requires the reporting also of the area of drained peatlands where peat is currently extracted. Currently, this applies only to the cutaway peatland in production currently owned by Bord na Móna; other small-scale commercial enterprises and domestic turf-cutting areas are not included.

While restoration of the C sink function in peatlands damaged by peat extraction has been investigated globally (Tuittila *et al.* 2000, Waddington *et al.* 2003, Yli-Petäys *et al.* 2007), research has been slower to develop in Ireland. A rewetted cutaway peatland in Bellacorrick, Co. Mayo has shown that appropriate after-use management can provide an important sink for CO₂ while managing the release of CH₄. However, the majority of the industrial cutaway peatlands are located in the Midlands, where the substrate is

alkaline and not readily suitable for recolonisation by ombrotrophic bog species, such as *Sphagna* (Renou *et al.* 2006). In addition, these cutaways may not receive sufficient rainfall (compared to the west of Ireland) and this challenges the establishment of appropriate species. However, fen development following the rewetting of such cutaway may still be a climate-friendly option; a current study in Blackwater, Co. Offaly is monitoring the C sink potential over several years in order to capture inter-annual variability (Wilson 2013). The heterogeneity of the cutaways in terms of wetness across the bog (due to topographical elevation differences between peat-extraction fields) means that vegetation re-establishment needs to be facilitated and tailored so that the capacity for the site to sequester C is not jeopardised by other areas of the cutaway where drier, bare peat release C to the atmosphere (Malloy and Price 2014).

Other land-use options for industrial cutaway peatlands have been investigated with regards to their capability to sequester C. In 1999, a suite of C gas exchange studies were initiated under the Bord na Móna funded CARBAL project (Wilson and Farrell 2007) in afforested, naturally regenerated and rehabilitated cutaway peatlands. In all scenarios, the authors reported large losses of soil CO₂ from the residual peat as have other studies in Finland (Mäkiranta *et al.* 2007) and Sweden (Tagesson and Lindroth 2007). More research is required on 'abandoned' cutaway peatlands where rewetting may not be feasible as other land use option may be more beneficial to offset the constant C release from the atmosphere (e.g. paludiculture). Sphagnum or medicinal plant species or even biomass as a renewable energy source are all potential alternative after-uses of cutaway which need to be further investigated under Irish conditions.

11 Final Observations and Recommendations

Observation 1: Our field research concluded that nutrient status, drainage and local grassland management practices impacted on GHG fluxes and fluvial C losses as well as biomass exports.

Recommendation 1: As well as using nutrient and drainage status, grassland management practices provide significant additional information for more accurate regional extrapolation of C fluxes. These attributes are in turn intimately linked to past and current management practices in terms of drainage duration and intensity and inputs.

Observation 2: Most Irish grasslands over organic soils were drained several decades ago and are managed as extensively grazed pastures with little or no fertilisation. Nutrient-poor sites in Ireland may not exert appreciable impacts on the atmosphere in terms of NECB, particularly where the mean annual water table remains within -25 cm of the soil surface.

Recommendation 2: Nutrient-poor organic soils which are poorly drained should be targeted to remained wet (WT higher than -25cm) with a continued low input/output system.

Observation 3: Nutrient-rich organic soils under grasslands produce much higher GHG emissions and also represent hotspots for fluvial C losses.

Recommendation 3: Nutrient-rich organic soils should be targeted for rewetting as a strategy to mitigate C emissions.

Observation 4: Current default EF may not be representative of the variety of grasslands over organic soils in Ireland. Our field-measured EF will help Ireland to refine EF and implement Tier 2 methodologies for IPCC inventories more effectively.

Recommendation 4: Continued monitoring to improve annual fluxes estimates to develop more robust emission factors by averaging inter-annual variability.

Observation 5: While the impacts of the nutrient and drainage status on CO₂ exchanges, biomass exports and fluvial C losses were confirmed, inter-regional differences in management practice and climate are also significant factors that affect the overall NECB of these ecosystems.

Recommendations 5: It is therefore critical to develop strategies to deliver reduced emissions tailored to local grassland types.

12 Conclusions

This study has provided estimations of GHG fluxes and waterborne C losses from contrasting grasslands over organic soils and has shown that these reflect not only regional variability but also local management practices. Hence, despite the critical role of nutrient and drainage status, detailed reporting of grassland management type may provide significant additional information for more accurate regional extrapolation of C fluxes. The large difference in net C balances between the main locations stems mostly from: (i) NEE estimations and (ii) biomass exports, which are affected by the grazing regime. Net ecosystem exchange estimations were driven mainly by local climate, soil fertility, water table level and potentially soil organic matter quality. These attributes are in turn intimately linked to past and current management practices in terms of drainage duration and intensity and inputs. Most Irish grasslands over organic soils were drained several decades ago and are managed as extensively grazed pastures with little or no fertilisation. Nutrient-poor sites in Ireland may not exert appreciable impacts on the atmosphere in terms of NECB, particularly where the mean annual

water table remains within -25 cm of the soil surface. In contrast, grassland over nutrient-rich, drained organic soils (with higher C content) produce much higher GHG emissions and also represent hotspots for fluvial C losses. As the C balance of such sites is driven to a large extent by soil respiration, the inverse relationship of this parameter with soil moisture suggests these sites should be targeted for rewetting as a strategy to mitigate C emissions.

This comprehensive investigation supplied effective information for Ireland to refine emission factors currently proposed for Tier 1 drained organic soils. Longer monitoring is required to capture inter-annual variability which affects NEE in particular (Drösler *et al.* 2008). While a 'grassland type' stratification, representing various management systems, may not be warranted at international guidelines levels, national research should be extended to other sites to provide more accurate emissions from extensive drained organic soils pastures and help Ireland progress towards Tier 2 methodologies.

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Acronyms and Annotations

CH ₄	Methane
CO ₂	Carbon dioxide
DIC	Dissolved Inorganic Carbon
DOC	Dissolved Organic Carbon
ECD	Electron Capture Detector
GHG	Greenhouse gas
GPP	Gross photosynthesis
GWP	Global Warming Potential
HCO ₃ ⁻	Bicarbonate
IPCC	Intergovernmental Panel for Climate Change
N ₂ O	Nitrous oxide
NEE	Net ecosystem exchange
pCO ₂	Excess CO ₂
POC	Particulate Organic Carbon
PPFD	Photosynthetic Photon Flux Density
R _{ECO}	Ecosystem respiration
TOC	Total Organic Carbon
UNFCCC	United Nations Framework Convention on Climate Change

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Background

Grassland is the predominant land use in Ireland. It is estimated that an area of approximately 300,000 ha of peat are drained for use as agricultural grassland. This leads to on-going loss of carbon from the soil and the emissions of greenhouse gases. The rate of emissions is sensitive to local conditions such as water table, meteorology and climate. This research presents findings from field measurement on two sites with diverse management regimes and an assessment of whether IPCC default emission factors are consistent with Irish conditions.

Identifying Pressures

Agriculture, particularly grassland based livestock farming, is the dominant land use in Ireland, and is recognised as the principle primary industry in the State. As a consequence, Agriculture is responsible for the largest share of GHG emissions compared to other sectors of the economy. Nevertheless, there is also the potential for removal of carbon dioxide from the atmosphere through good practice in the management of agricultural and other lands. Both the emissions and removal potential need to be quantified at a national level in order to better inform decision making on future land management. Drained organic soils are recognised as a potential “hot spot” source of emissions and additional information is required to establish the potential for mitigation of emissions through policies aimed at improved management of these lands.

Informing Policy

- Findings from this study indicate that the carbon losses on two field sites on agricultural drained organic soils in Ireland were less than the IPCC default emission factors. The rate of emissions depends on a variety of factors, including water table, nutrient status etc.
- On the high nutrient status organic soils lose carbon at a rate of 5.8tC ha⁻¹, was observed, and is consistent with the IPCC default value of 6.1tC ha⁻¹. However, the loss rate for low nutrient status soil was found to be 2.35 tC ha⁻¹, significantly less than the IPCC default of 5.3 tC ha⁻¹.
- The study also investigated carbon loss to water, methane and N₂O emissions associated with land use and management at the sites.
- The study identifies the need for additional research into the impact of management on drained organic soils under grasslands in Ireland, and also for an investigation of mitigation options to avoid emissions.

Developing Solutions

- The research identifies the options for mitigation of climate change in the agricultural land use sector related to the management of drained organic soils.

