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Quantifying and mitigating greenhouse gas emmissions from horticultural peat soils

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Quantifying and mitigating greenhouse gas emissions from horticultural peat soils

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to Bangor University

by

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This thesis is dedicated to the memory of Joan Bennett… one seriously inspiring lady.

Summary

Global greenhouse gas emissions associated with peat soils under horticultural production may be substantial, and contribute towards degradation of a key global carbon stock. Emissions from horticultural peat soils are currently poorly quantified, restricting the scope for prioritising sectorial mitigation efforts. This study provides the first UK estimate of emissions of CO_2, N_2O and CH_4 from peat soils under horticultural production, and focuses the evidence base for mitigation efforts, using multiple, complementary methods. Emissions quantification was addressed in a field study over a complete annual cropping cycle, and using a mathematical modelling approach. Identification and prioritisation of mitigation measures was achieved through a search of the scientific literature, followed by short-listing of measures using a Best-Worst Scaling survey approach, and field-testing of a selected small number of candidate measures in a mesocosm experiment. This study indicated that IPCC emissions estimates may represent an underestimate of peatland N_2O emissions, and that net emissions of CO_2 and CH_4 represent a substantial annual soil carbon loss. Calibrated emissions models provide a means of upscaling estimates, but the DNDC (Denitrification-Decomposition) model offers limited potential at the present time. A holistic approach to mitigation is recommended, encompassing retailers and consumers as well as farmers, in an effort to reduce food waste at all stages of the food chain. On-farm mitigation should focus in the short-term on nutrient management, and in the longer term, on considering alternative land uses.

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Abbreviations and acronyms

BSI	British Standards Institute
BWS	Best-Worst Scaling
CF	Carbon footprint
CH_4	Methane
CH_4-C	Methane in terms of carbon
CI	Confidence interval
CO_2	Carbon dioxide
CO_2-C	Carbon dioxide in terms of carbon
CO_2-e	Carbon dioxide equivalents
CV	Coefficient of variation
DECC	Department of Energy and Climate Change
DEFRA	Department of Agriculture, Food and Rural Affairs
DNDC	Denitrification-Decomposition model (a biogeochemical model)
EF	Emission factor
FAO	Food and Agriculture Organization
g	Gram
GHG	Greenhouse gas
GWP_{100}	Global warming potential, over a hundred-year time period
ha	Hectare
IPCC	Intergovernmental Panel on Climate Change
ISO	International Standards Office
K	Potassium
kg	Kilogram
KP	Kyoto Protocol
kPa	Kilopascal
LCA	Life cycle assessment
LCTP	Low Carbon Transition Plan
m	Metre
MACC	Marginal abatement cost curve
MM	Mitigation measure
MNL	Multinomial logit
N	Nitrogen
N_2	Dinitrogen
N_2O	Nitrous oxide
N_2O-N	Nitrous oxide in terms of nitrogen
NH_3	Ammonia
NH_4^+	Ammonium
NO	Nitric oxide
NO_2^-	Nitrite
NO_3^-	Nitrate
P	Phosphorus
PAS	Publicly Available Specification
S	Sulphur
SO_4^{2-}	Sulphate
t	Ton
WFPS	Water-filled pore space
yr	Year

Introduction

Global atmospheric greenhouse gas (GHG) concentrations have been rising since the Industrial Revolution (c. 1750). This trend has shown an apparent acceleration recently, with average concentrations increasing by c. 70% between 1970 and 2004 (Solomon *et al.*, 2007). The collective increase in atmospheric GHG concentrations is strongly associated with global climate change, and has been primarily attributed to anthropogenic activity (IPCC, 2007; IPCC, 2013a). Agriculture is an important emissions source, contributing an estimated 13.5% of global emissions, while emissions from land use, land use change and forestry (LULUCF, including croplands) may account for a further 17.3% (IPCC, 2007; Smith *et al.*, 2008). The contribution of agricultural emissions in the UK in 2012 was close to the global average, at 10.1% of UK emissions, representing the second largest emissions sector after energy (84.1% of emissions; Salisbury *et al.*, 2014; Webb *et al.*, 2014). The UK LULUCF sector has been a small net emissions sink since 2001 (-1.3% of emissions in 2012), but its cropland and wetland components constitute net sources of GHGs (Salisbury *et al.*, 2014).

Of the GHGs listed under the Kyoto Protocol, carbon dioxide (CO_2), nitrous oxide (N_2O), and methane (CH_4) represent the majority of agricultural and LULUCF emissions (IPCC, 2006). Agriculture accounts for approximately 56% of global emissions of N_2O and CH_4, and approximately 85% and 47% of UK N_2O and CH_4 emissions, respectively (IPCC, 2013a; Salisbury *et al.*, 2014). A c. 20% reduction in these GHGs has been observed since 1990 in the UK, chiefly in the livestock sector, through reducing livestock numbers and reducing mineral nitrogen (N) fertiliser application rates to pasture (Webb *et al.*, 2013). Little direct information exists relating to trends in the arable sector, but emissions of N_2O from mineral N application are thought to be declining, in line with a general reduction in fertiliser use across all tillage crops (Holmes, 2013). Since 1990, CO_2 emissions from existing UK cropland (reported under the IPCC LULUCF category) have increased by 36%, but emissions resulting from the conversion of other land uses to cropland have declined by 43% (NAEI, 2014). Inventory estimates for the LULUCF sector are based on a relatively sparse evidence base, and are commonly averaged across the entire global temperate zone (Webb *et al.*, 2013). As a result, emissions from LULUCF, and the amount of C stored under different land use scenarios, constitute considerable sources of uncertainty in national inventory estimates. In particular, there is a notable absence of robust evidence concerning the magnitude of CO_2, N_2O and CH_4 emissions from historically drained and cultivated peatlands (IPCC, 2006; Salisbury *et al.*, 2014; Webb *et al.*, 2013).

Peatlands, also known as organic soils or Histosols, represent an important global carbon (C) stock, covering only 3% of Earth's terrestrial surface but containing an estimated 30% of the total soil C stored (Parish *et al*., 2008). When cultivated, Histosols are highly vulnerable to degradation in the form of oxidation of soil carbon to CO_2, and wind and water erosion (Dawson and Smith, 2007). Emissions of N_2O also tend to be higher from managed than from natural peatlands (Kasimir-Klemedtsson *et al*., 1997). It is estimated that half of the historic global peatland loss has been a result of agricultural degradation (Joosten and Clarke, 2002). Within Europe, the UK possesses one of the highest percentages of peat soil cover, at 18.3% of its land area, although most of this area is upland nutrient-poor peatland not used for intensive agriculture (Montanarella *et al*., 2006). Nevertheless, peatlands in the UK represent a potentially substantial source of GHG emissions.

Currently, estimating total emissions from cultivated peat soils in the UK is challenging, as the total peat area under different types of agriculture is unknown. The IPCC definition of 'historically drained and cultivated' peat soils considers managed peat soils across all agricultural sectors, and in England, applies to lowland peats only (Webb *et al*., 2013). While other areal estimates exist, no whole-UK studies have disaggregated cultivated peatlands into livestock and arable sectors (e.g. Bain *et al*., 2011; Smith *et al*., 2010a). Additionally, very little reliable information is available on the rates and patterns of CO_2, N_2O and CH_4 emissions under different types of agricultural management. The IPCC methodology used in the most recent national GHG inventory uses an average value across all temperate zones and agricultural sectors for N_2O emission, and a UK-specific proxy value for CO_2 emission, while it neglects CH_4 fluxes altogether (Webb *et al*., 2013). This approach may result in considerable errors in estimates, given that GHG emissions may vary quite widely under different regional or local conditions of climate, soil, and farm management (e.g. Li, 2007).

One agricultural sector which has received relatively little attention from a GHG perspective is the horticultural sector. Horticultural crops are by definition intensively managed, so may be considered as a potential 'worst-case scenario' candidate for peat soil degradation and GHG emission (Parish *et al*., 2008; Tigchelaar and Foley, 1991). No known complete annual GHG budgets (CO_2, N_2O and CH_4) have been published, for any temperate horticultural crops grown on peatlands, although a few exist for other arable crops such as cereals (e.g. Elder and Lal, 2008; Maljanen *et al*., 2004). In relation to the UK, published annual GHG budgets for Histosols under continuous arable cropping are entirely lacking (Evans *et al*., 2011; Worrall *et* *al*., 2011). This represents a significant knowledge gap, which urgently needs addressing in order to refine the methodology applied to the UK GHG inventory approach. An annual estimate of GHG emissions from field-grown vegetables on peat soils is critical, as field-grown vegetables represent the largest UK horticultural sector in terms of both production area (81.0% in 2012) and home production marketed (52.1% in 2012; DEFRA, 2014).

Refining the UK GHG inventory method is important from two key perspectives. Firstly, a more accurate evaluation of the annual contribution of horticultural peat soils to overall UK emissions will aid the prioritisation of sectorial mitigation efforts. Secondly, UK-specific knowledge of the factors contributing towards temporal and spatial variability in emissions could help to indicate the type of mitigation measures that might be most effective at reducing or offsetting emissions. As signatories to the Kyoto Protocol, the UK government is bound to the EU's commitment to reduce national emissions by 12.5% compared to a 1990 baseline, during the period 2008 to 2012 (Webb, 2014). Further legislative targets have been subsequently set under the UK Climate Change Act (2008), proposing a reduction of at least 34% by 2020, and at least 80% by 2050 against the 1990 baseline, using a series of legally binding 5-year targets (DECC, 2011; UK Parliament, 2008). In meeting these ambitious targets, it is crucial to address significant knowledge gaps in national estimates of GHG emissions.

The primary objectives of this work were to provide an estimate of GHG emissions from peat soils under horticultural vegetable production in the UK, and investigate the potential for reducing net GHG emissions from UK horticultural peat soils.

The specific research aims are:

1. To produce a full annual GHG budget for a number of commercially important vegetable crop rotations, grown on soils of contrasting, medium-to-high soil organic matter content ('horticultural peat soils').
2. To assess the performance of a biogeochemical model (the DNDC model) in predicting GHG emissions from horticultural peat soils.
3. To identify mitigation options with the potential to reduce net GHG emissions from UK horticultural peat soils.
4. To evaluate the effectiveness at reducing emissions, and practicality of implementation, of a selected shortlist of potential GHG mitigation options.

The thesis structure, and relationships between its six chapters, are summarised in Fig. 1. The individual chapters sequentially address the research aims listed above, in the following way:

Chapter one reviews the published and grey literature. The boundaries of the system under investigation are defined. Carbon and nitrogen biogeochemistry in relation to GHG fluxes are discussed, including a synthesis of existing knowledge on the environmental and farm management factors influencing GHG flux magnitude and patterns. An outline is given of the methods available for measuring fluxes of CO_2, N_2O and CH_4, and the relative utility of each method is discussed. Finally, consideration is given to the range of potential GHG mitigation options available for application to horticultural systems on peat soils.

Chapter two addresses the first research aim and comprises a field study of seasonal variation in GHG emissions over a complete annual cycle. Fluxes of soil respiration CO_2, N_2O, and CH_4 were monitored on a monthly basis, using a closed chamber method. Three contrasting peat soil types of increasing organic matter content were monitored, under a range of commercially important crops. Annual estimates were computed from monthly data, and the environmental factors contributing to seasonal emissions variation were determined.

Chapter three compares measured field data from chapter one with simulated emissions from a bio-geochemical model, the DNDC (Denitrification-Decomposition) model. This comparison addressed the second research aim, by comparing model performance with and without calibration, on two fields of contrasting organic matter content. A sensitivity analysis was conducted to inform the calibration process. Finally, a validation exercise was executed for the remaining two fields from each site, to verify the success of the calibration process.

Chapter four responds to both the third and fourth research aims, using a utility choice model approach. A review of the scientific and grey literature provided a list of potential GHG mitigation options which might be applicable to horticultural peat soils. An initial online survey of academic and industry experts reduced the list to a short-list of priority measures. The short-list was deployed through two further online surveys, which used the Best-Worst Scaling method to gauge expert opinion on the relative efficacy of proposed mitigation measures for reducing net GHG emissions, and farmer opinion on the relative practicality of implementing the same interventions on the horticultural peatland on their farm.

Chapter five provides further evidence to address the ‘effectiveness’ aspect of research aim four, and ran in parallel to the preceding chapter. This chapter explored the overall effect on emissions of a novel application of several treatments to nutrient-rich horticultural fen peats, previously only applied to nutrient-poor blanket peats. A small number of potential mitigation options were selected for evaluation in a field situation, on a soil of high organic matter content. An outdoor mesocosm experiment using soil cores was conducted, to assess the effects of the following on the net GHG budget: water table raising, no-till cultivation, horticultural fleece application, and soil incorporation of lettuce residue. An intensive monitoring approach provided detailed information on the temporal variation in soil properties and associated GHG emissions under each treatment.

Chapter six comprises a general discussion of the experimental chapters, focussing on strengths and weaknesses of individual studies, links between chapters, and an analysis of the work in a wider context. Recommendations are provided for future work.

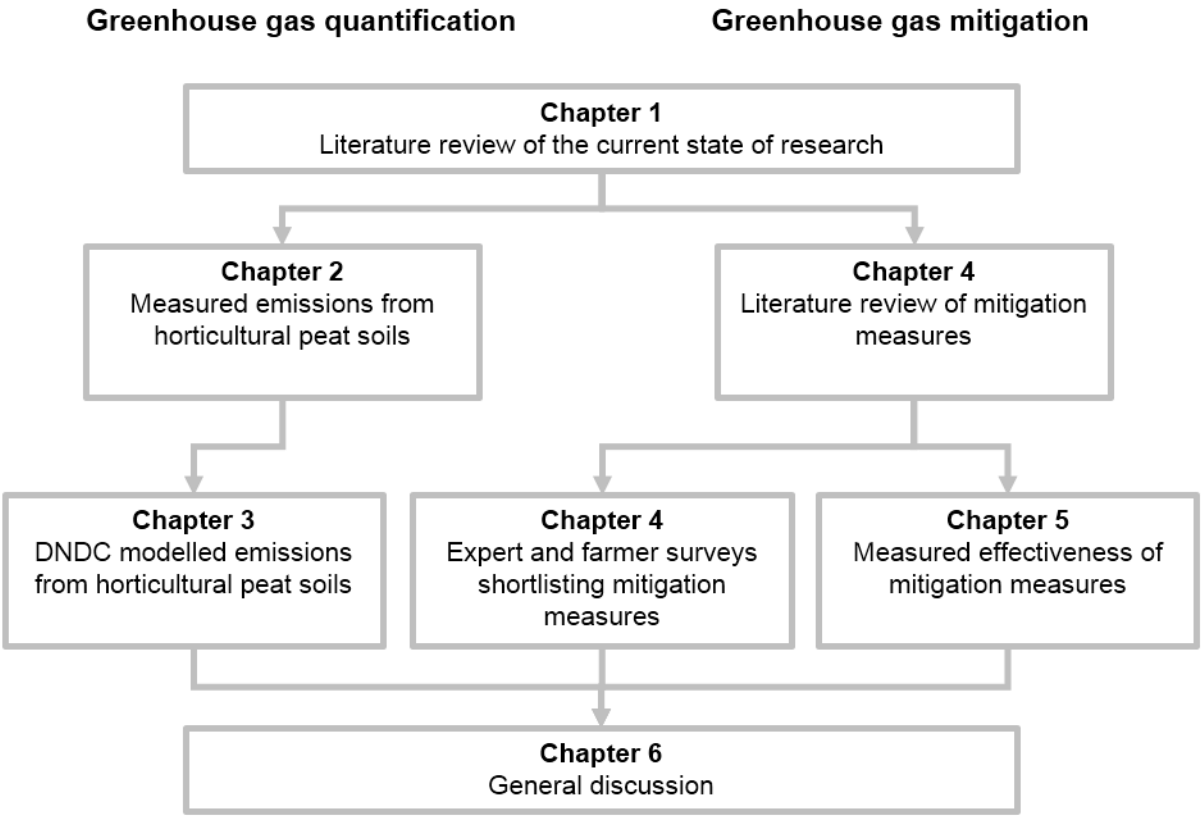

Figure 1. Schematic of chapter content and progression.

Chapter 1

Greenhouse gas emissions from horticultural peat soils:

Mechanisms, measurement and mitigation

Abstract

Greenhouse gas emissions associated with agricultural and land use sources are of global significance, comprising up to a third of net global emissions. Historically drained and cultivated peat soils, currently under intensive cropping systems such as those found in horticultural production, may constitute a significant emissions source. In the UK, direct emissions measurements from horticultural peat soils are lacking, and the evidence base for GHG mitigation is incomplete. This review compiles and critically analyses evidence from the scientific and grey literature with regards to the importance of the global and national peatland carbon stock, the contribution of intensely cultivated peatlands to UK greenhouse gas emissions, and the potential for mitigation. A brief overview of available methods of analysis is also presented. Key knowledge gaps identified in this review include (1) The lack of a complete annual GHG budget from lowland peat soils under horticultural production in the UK; (2) High uncertainty surrounding the area of UK peat soils under intensive horticultural production; (3) A scarcity of literature detailing the effects of horticultural management practices on peat soil emissions; and (4) High uncertainty over the effectiveness and practicality of potential GHG mitigation practices.

1.1. Introduction

A forecasted expansion of the global population to c. 9 billion by the middle of this century, will increase the demand for food production to previously unsurpassed levels (Godfray *et al*., 2010). The resulting pressure on the world's environmental resources is likely to exacerbate existing issues of environmental degradation such as loss of natural habitats and species diversity, soil erosion, air and water pollution, desertification, salinization, and greenhouse gas (GHG) emissions (FAO, 2007; Godfray *et al*., 2010; Tilman *et al*., 2002). Of particular concern is the impact of GHG emissions on global climate, because of its widespread effects on Earth's physical and biological systems, intensification of the effects of other global environmental problems, and potential irreversibility (Solomon *et al*., 2007). The increasingly erratic weather patterns predicted as a result of climate change, are likely to compromise food production security over the next 50 years via changes in the global distribution of crop productivity, including an increase in crop losses due to drought and flooding events (Solomon *et al*., 2007).

Emissions associated with agricultural production and land use change make up almost a third of global GHG emissions (Smith *et al*., 2008; Solomon *et al*., 2007). Although CO_2, N_2O and CH_4 are all significant in terms of agricultural emissions, N_2O and CH_4 are of particular concern because their radiative forcing effect on the atmosphere over a 100-year time period, is 310 and 21 times greater than that of CO_2, respectively (Webb *et al*., 2014). Consequently, even relatively small emissions of these gases can have a disproportionately greater impact on climate when compared to the effect of CO_2. Emissions of N_2O from agricultural soils are a particular source of uncertainty in national emissions estimates (IPCC, 2006).

To address concerns over climate change impacts, the UK is committed to reducing future net GHG emissions from all sectors under the UK Climate Change Act and subsequent UK Low Carbon Transition Plan (DECC, 2011; UK Parliament, 2008). Within the food production sector, over half of the market share of supermarkets in 2010 had a company climate change policy, encouraging sourcing of low-emission products from growers (DEFRA, 2012). Emissions reduction policies of the four market leaders include targets for reducing GHG contributions from growers, transport, packaging and storage (Asda, 2014; Morrisons, 2014; Sainsbury's, *s.d.*; Tesco, 2013). Knowledge of the contribution of suppliers to overall food chain emissions is required so that supermarket policy-makers can prioritise allocation of emission reduction efforts at critical control points in the supply chain. More broadly, addressing knowledge gaps surrounding agricultural sectors significantly contributing to national- and sector-level GHG emissions, is also important in meeting the ambitious targets set out in international, national and sectorial policies. Emissions from horticultural crops grown on drained and cultivated peat soils represent one such knowledge gap, particularly with respect to N_2O. This review collates and critically analyses evidence from the scientific and grey literature relating to horticultural peatland emissions.

1.2. Defining the system: What are horticultural peat soils?

To quantify baseline emissions of CO_2, N_2O and CH_4, and measure impacts of emissions mitigation within a system, the bounds of the system of interest must be adequately defined. In the IPCC literature used for current UK national inventory purposes, emissions resulting from horticultural use of peat soils are described and accounted for under several sub-categories within the AFOLU (Agriculture, Forestry and Other Land Use) reporting sector, comprising emissions from two source categories: Agriculture; and Land Use, Land Use Change and Forestry (LULUCF) (IPCC, 2000; 2006; Webb *et al*., 2014). While the term 'horticulture' is not defined as a specific sub-category, it is included under the remit of *cultivated Histosols* in relation to N_2O emissions, and *Drained and cultivated organic soils*, *Croplands*, or *Managed Wetlands* when accounting for CO_2 or CH_4 emissions (IPCC, 2000; 2006). Similarly, although peat soils are most frequently referred to as *organic soils* in the IPCC literature, several apparently interchangeable terms are used: *organic soils*, *Histosols,* and *peat lands*. Table 1.1. illustrates the range of definitions relating to horticultural peat soils presented in the IPCC methodology.

The variable definition of horticultural peat soils used in IPCC methodology may reflect the wide range of synonyms and partial synonyms found in the scientific literature. At least twenty separate words describing different types of peat land exist, leading to some confusion when assessing the relevance of the literature (Bussell *et al*., 2010; Joosten and Clarke, 2002). Further, soil classification systems can vary in their definition of peats according to depth or percentage of organic matter (e.g. Avery, 1980; FAO, 2006; USDA, 2010). Use of the term *horticulture* also appears to be context-dependent, but is normally used as a generic term describing intensively produced temporary or permanent crops including vegetables, fruit, and ornamental plants (FAO, 2006; EC, 2012; DEFRA, 2010).

To allow comparison with IPCC inventory estimates of emissions, this study adheres to the definition of Histosols given in Table 1.1. throughout, but uses the terms *Histosols*, *peat soils*, *peatlands* and *organic soils* interchangeably to describe Histosols. In this review, the term horticulture should be taken to mean the full range of horticultural crops defined by DEFRA (2010), but with a particular emphasis on annual vegetable crops (consistent with the majority of the available literature).

Table. 1.1. IPCC emissions source categories used in the UK GHG National Inventory Report, relating to horticultural use of peatlands (IPCC, 2006).

SOURCE CATEGORY	GHG	DESCRIPTION	DEFINITIONS
4D: AGRICULTURAL SOILS **4D1: Agricultural soils: Direct soil emissions** **(iv) Cultivation of Histosols (organic soils)**	N_2O	*'Soil nitrogen mineralisation due to cultivation of organic soils (i.e. Histosols).'*	'***Organic soils*** *are soils described as Histosols which are defined as: "Organic soils that have organic soil materials in more than half of the upper 80 cm, or that are of any thickness of overlying rock or fragmented materials that have interstices filled with organic soil materials.' An organic soil material is defined as: 'soil materials that are saturated with water and have 174 g* kg^{-1} *or more organic carbon if the mineral fraction has 500 g* kg^{-1} *or more clay, or 116 g* kg^{-1} *organic carbon if the mineral fraction has no clay, or has proportional intermediate contents, or if never saturated with water, have 203 g* kg^{-1} *or more organic carbon (SSSA, 1996)." '* '***Histosols*** *are soils containing an organic-rich surface layer at least 40 cm in thickness, with a minimum of 20% organic matter if the clay content is low, and a minimum of 30% organic matter where the clay content exceeds 50%.*'
5B: CROPLAND **5B1: Cropland remaining Cropland**	CO_2	*'Ongoing carbon stock changes in soils arising from historical land use change to Cropland more than 20 years before the inventory reporting year', plus* *'(ii)* ***organic soil carbon emissions*** *from* ***fenland*** *drainage'* *'**fenland** areas of England were drained many decades ago for agriculture which allowed oxygen into previously water logged soils. As a result, soil carbon in these areas continues to oxidise and be released as* CO_2*, resulting in an ongoing change in soil carbon stock.'*	No further definition of **organic soil** given. No definition of fenland given. *'**Cropland (Arable and Horticulture)**: All arable crops such as different types of cereal and* ***vegetable crops****, together with orchards and more specialist operations such as market gardening and commercial flower growing, freshly ploughed land, fallow areas, short-term set-aside and annual grass leys, are also included in this category.'*
5B2: Land converted to Cropland	CO_2	*'Carbon stock changes and biomass burning emissions due to conversion of other land categories to Cropland in the previous 20 years before the reporting year'* *'biomass burning emissions occur in the same year as the land use conversion, while loss of soil carbon occurs over a longer period'* *'both mineral and organic soils are included'*	
5D1: Wetlands remaining Wetlands	CH_4	*'**Peatlands** cleared and drained for production of peat for energy, horticultural and other uses'* *'Wetlands include any land that is covered or saturated by water for all or part of the year, and that does not fall into the Forest Land, Cropland, or Grassland categories. Managed wetlands are those where the water table is artificially changed (i.e. raised or drained) or those created by human activity.'* *'…emissions from on-site peat production and off-site emissions from horticultural peat…'*	

1.3. Peatland carbon and nitrogen cycling

1.3.1. Carbon and nitrogen storage in soils

Soils act as an important sink for C and N at the global and national scale, and are an important potential source of GHG emissions. Limitations to accurate quantification of global C and N stocks exist due to the large scale and incomplete coverage of available data, and considerable variability of soil C and N content even within soil types (Batjes, 1996). Taking these sources of error into account, it is estimated that the Earth's soil C stock (c. 2500 Gt C) may be approximately three times that of the atmosphere (c. 760 Gt C) and five times that held in vegetation (c. 560 Gt C), but only approximately one fifteenth the oceanic C pool size (c. 38,400 Gt C) (IPCC, 2007; Lal 2008). Quantitative estimates of the N stock in global soils and other N pools are scarce. The total reserve of soil N in the upper 1 m of global soils is thought to be approximately 5% that of soil C, at between 133 and 140 Gt N, and is likely to be a very small fraction of the N held in the atmosphere or oceans (c. 3,925,000 Gt N), but many times greater than that contained within plant or microbial biomass (c. 12 Gt N) (Batjes, 1996; Jenkinson, 1990). Almost half of the C and N in world soils is estimated to be held in the top 30 cm of the soil profile (c. 63 to 67 Gt N) (Batjes, 1996), making it more vulnerable to exposure and oxidation than more deeply buried reserves.

Up to 30% of the world's soil C stock may be contained within peat soils (Limpens *et al*., 2008; Parish *et al*., 2008). It is unclear from the literature what proportion of the global soil N pool is present in Histosols, although estimates of N concentrations to 1 m depth for all world soil types suggest that Histosols may have amongst the highest N content of all soil types (Batjes, 1996). Despite their importance in terms of C and N stock size, peat soils are estimated to cover only an estimated 3% of the Earth's land surface (Limpens *et al*., 2008). Any ongoing changes in global peatland area, depth or quality could therefore disproportionately affect the global C and N budgets (Bellamy *et al*., 2005).

The largest proportion of Earth's peatland area lies within the boreal, subarctic and low arctic areas of the Northern hemisphere, while an estimated 7 to 12% of the global peatland area is located in Europe (Bain *et al*., 2011; Montanarella *et al*., 2006; Parish *et al*., 2008). Within Europe, the majority of peat soils are distributed throughout Scandinavia (c. 70%), and in the UK and Ireland (c. 20%; Montanarella *et al*., 2006). Estimates of the total peat soil area of the UK, and the magnitude of its total C stock, vary considerably, with their accuracy dependent primarily on the spatial resolution, soil depth, and soil organic carbon (SOC) threshold considered for mapping purposes. No known quantitative studies of UK soil N stocks are available. Between 18% and 33% of the UK land area is thought to be covered by peats or peaty soils, primarily in Scotland and Northern Ireland, and in the uplands of England and Wales (Fig. 1.1.; Bain *et al*., 2011; Bradley *et al*., 2005; Montanarella *et al*., 2006). This represents an estimated total UK peat soil C stock of between 3,200 and 4,605 Mt, with c. 75% held in the upper 1 m depth of soil (Bain *et al*., 2011; Bradley *et al*., 2005).

The total C storage within UK peatlands specifically under horticultural production has not been quantified in the literature; studies relating to cultivated peatlands tend to focus on arable cropland as a whole, without subdivision into cereals, horticulture, and temporary grass leys and set-aside. The area of UK Histosols under arable production is also uncertain. A recent study by Smith *et al*. (2010a) estimated the area of arable land in Great Britain on organic soils (> 12% SOC content) at 211,574 ha, with a further 1,267,751 ha on organo-mineral soils (5-12% SOC content). In contrast, the IPCC method applied to current UK GHG inventorying, suggests a value of 150,000 ha of UK arable fen peat soils in England, and 8,500 ha of drained lowland fen with humose topsoils in the rest of the UK, principally in Northern Ireland (Cannell *et al*., 1999; Webb *et al*.; 2013). The land area occupied by horticultural crops in the UK in 2013 was at the lower end of this range of estimates, at 163,355 ha (DEFRA, 2014), and might be taken as a maximum area of peat soils under horticultural production. While it is very unlikely that all UK horticultural production occurs on peat soils, peatlands are an important component of the UK's most productive land grades; for example, the English East Anglian Fens are classified as 49% Grade 1 land and 34% Grade 2 land (Graves and Morris, 2013). Given that the Fens represent the largest area of cultivated nutrient-rich fen peat in the UK, a crude minimum estimate might therefore be calculated by assuming the proportion of peat soils occupied by horticultural crops is proportional to the percentage of UK fen peat occupying grade 1 agricultural land; this gives a value of between 77,665 and 103,671 ha.

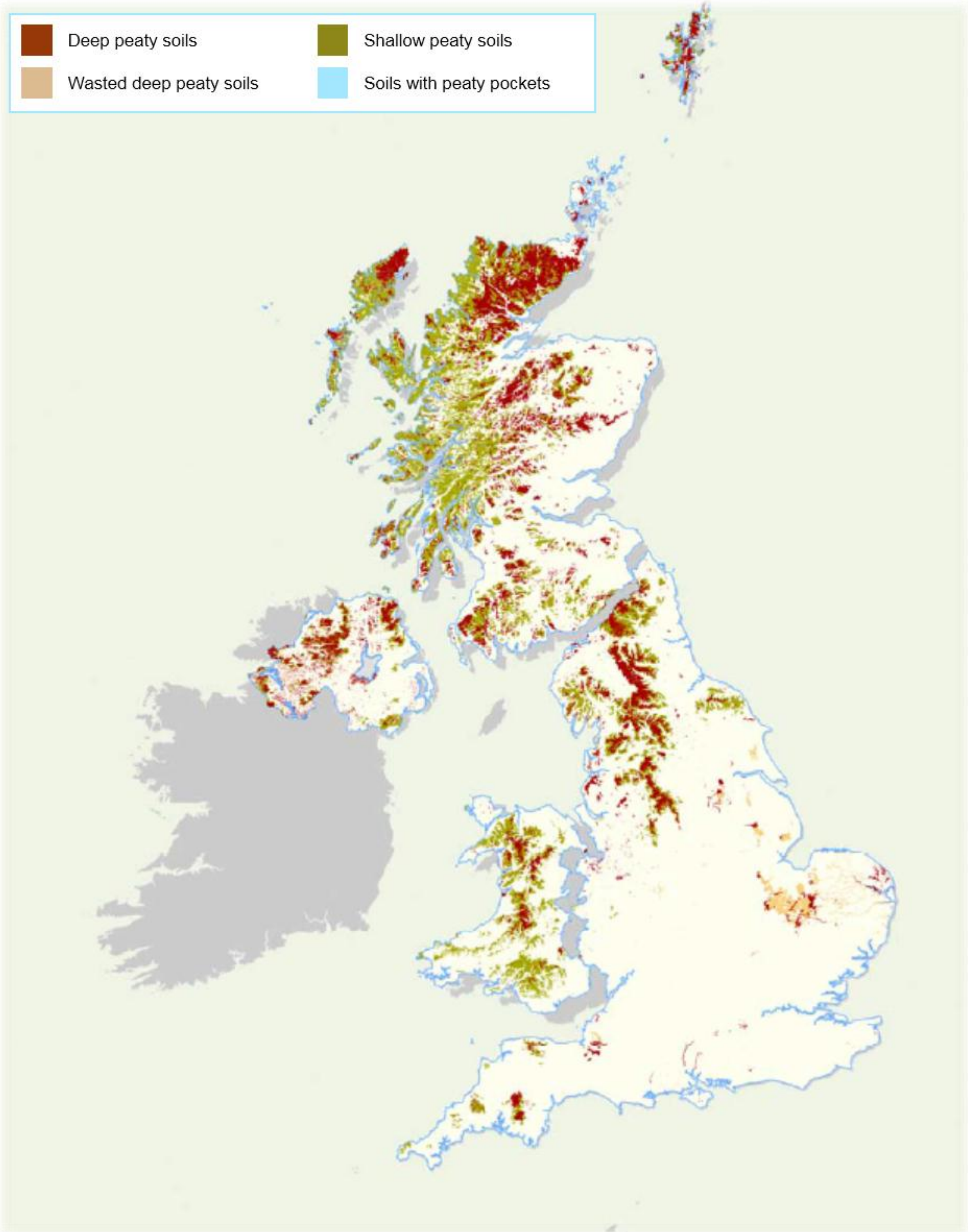

Figure 1.1. Peat and peaty soils of the United Kingdom – JNCC (2011). Reproduced and adapted from JNCC (2011), by kind permission of JNCC. Original reproduction by permission of OS on behalf of HMSO @ Crown copyright and database Right 2010, MLUIR 100019294, AFBI 1:50000 soil digital Data, National soil Maps @ Cranfield University, BGS 1:50000 digital data (license 2006/072).

1.3.2. Carbon and nitrogen cycling in cropped peat soils

The carbon (C) and nitrogen (N) cycles are closely coupled, and have many components and transfer routes in common. Since changes in one cycle often impact on the other, as well as each interacting with the phosphorus (P) and sulphur (S) cycles, they cannot be considered in isolation (Gruber and Galloway, 2008; IPCC, 2007). While there is some variation in estimates of the size of global C and N stocks, and of the magnitude of flows between pools, there is a reasonably high level of agreement over the biogeochemical processes involved (e.g. Solomon *et al*., 2007). Here, C and N biogeochemistry are considered with respect to the generation and absorption of CO_2, CH_4 and N_2O in cropped peat soils.

Generation of CO_2, CH_4 and N_2O from soils are all microbially mediated processes; in addition, CO_2 generation is plant-mediated (Smith *et al*., 2003). The transport of GHGs within and between soil horizons, and from the soil to the atmosphere, is controlled by a variety of complexly interacting physical, chemical and biological factors (Li, *et al*., 1992a; Li, 2007). Absorption of GHGs from the atmosphere into the soil may occur by simple chemical assimilation, or by the action of soil microbes. Descriptions corresponding to the current knowledge of these processes are given in the following sections.

Within cropped agroecosystems, C and N are stored in the form of vegetation (e.g. crops, weeds, woody vegetation on marginal land), soil organic and inorganic matter, and decomposing organic matter (crop residues and soil biota) (IPCC, 2007; Li, *et al*., 1992a). Carbon flows from the atmospheric pool into the terrestrial pool when plants take in CO_2 via the process of photosynthesis, and when soil microbes oxidise CH_4 (a process known as methanotrophy) (Lal, 2008; Le Mer and Roger, 2001). Transfers of N from the atmosphere occur primarily via N_2 fixation by plants, with smaller inputs provided by atmospheric deposition of nitrate (NO_3^-) and ammonium (NH_4^+) (Gruber and Galloway, 2008). Weathering and erosion of rocks and minerals provide longer-term natural inputs of inorganic C and N (IPCC, 2007). An important additional anthropogenic input into cropped soils is that of mineral or organic (manure) fertilisers (e.g. Skiba *et al*., 2012).

Nutrient flows within the terrestrial ecosystem occur between vegetation, soil and microbial C and N stores. Breakdown of vegetative matter as result of natural root and myccorhizal turnover, or from crop residue breakdown, provide substrates for microbial respiration, along with simple sugars exuded from plant roots (Lal, 2004). Simple inorganic and organic N compounds produced from breakdown of OM or addition of fertilisers may also be taken up by plant roots or soil microbes (Dalal *et al*., 2003). Some of the resulting decomposed matter is incorporated in the soil as SOM or microbial biomass. Microbial biomass is cycled within the system on the event of microbial death and decomposition (Li *et al*., 1992a).

Outflows of C and N from the terrestrial pool arise via GHG emissions, leaching or erosion of nutrients into watercourses, or wind erosion (e.g. Dawson and Smith, 2007). Vegetative, myccorhizal, and soil (microbial) respiration are the primary mechanisms through which CO_2 is released to the atmosphere from soils in agroecosystems, while CH_4 is produced via the microbially-mediated process of methanogenesis (Dalal *et al*., 2008; Smith *et al*., 2003). Generation of N_2O may occur via the action of nitrifying and denitrifying soil microbes in nitrification and denitrification processes, or through abiotic nitrite (NO_2^-) and NO_3^- reduction (Chapuis-Lardy *et al*., 2007; Venterea and Rolston, 2000). Denitrification also produces NO and N_2 effluxes under some conditions (Dalal *et al*., 2003). Transfer of C and N into watercourses may occur through leaching of dissolved organic or inorganic forms, or eroded as particulate matter (e.g. Dawson *et al*., 2002). After C and N are leached or eroded into watercourses, additional indirect GHG emissions may occur through further *in-situ* microbial action.

The mechanisms involved in the emission of CO_2, N_2O and CH_4 vary between GHGs. Transfers of CO_2 and N_2O from the soil to the atmosphere tend to occur by simple diffusion (Smith *et al*., 2003). While diffusion is also an important route for CH_4, CH_4 may additionally be transported from the soil by ebullition (bubbling of methane pockets to the soil surface) or plant-mediated transport (Le Mer and Roger, 2001; Smith *et al*., 2003). The extent to which different transport routes dominate, is dependent on soil conditions (section 1.4.1.).

1.3.3. Carbon and nitrogen losses from cultivated peat soils

Losses of CO_2 through oxidation of cultivated peat soils in the boreo-temperate zone are estimated in the literature at between 8.0 and 83.4 t CO_2-e ha^{-1} yr^{-1}, with most estimates in the mid- to upper range, and considering soil respiration only (Table 1.2.). The one study that provided an estimate of both soil respiration and NEE, estimated gross primary production at only c. 60% of NEE, i.e. crop photosynthetic activity did not mitigate total soil respiration (Morrison *et al*., 2013). Differences in respiration rates appear to be partially explained by latitudinal position, with an average higher soil respiration from temperate zones than from boreal zones. Cultivated peat soils are thought to yield on average, 2-4 times higher CO_2 fluxes than cultivated mineral soils (Elder and Lal, 2008). Mean C loss from organic soils in England and Wales during 1978-2003 was estimated to be an order of magnitude faster than from mineral or man-made soils, at a rate of > 2% yr^{-1} in soils with an SOC content of > 100 g kg^{-1} (Bellamy *et al*., 2005). Very little further information is available on oxidative losses from UK cultivated peats, with only a single study on lettuce found in the literature (Table 2.1.; Morrison *et al*., 2013). Emissions (again, primarily soil respiration) from undrained semi-natural peatlands tend to be lower in comparison to those from cultivated peats, although not always significantly (e.g. Bussell *et al*., 2010).

Soil subsidence (loss of soil height) is partly attributable to CO_2 emission, and is considerably more pronounced in recently drained peat soils than historically drained soils (e.g. Hooijer *et al*., 2012). This difference is largely a function of the contrasting processes governing subsidence rates during its two principal phases. When Histosols are first drained, soil height loss is primarily a function of water volume loss, with accompanying shrinkage and settlement of soil layers above the groundwater table, and peat consolidation below groundwater level (Kasimir-Klemedtsson *et al*., 1997). During this phase, oxidation loss is thought to be a minor component of SOC loss (although it may still be substantial in absolute terms). While the physical processes of further shrinkage due to evaporation, and compression from farm management operations, continue during the second phase of subsidence, peat oxidation becomes dominant as the primary cause of soil height loss over time (Hooijer *et al*., 2012). As SOC is lost over time, mineral matter collects in subsiding soil and stabilises the collapse rate, with sometimes partial restoration of SOM through crop residue inputs. Thus, loss of depth commonly slows with time, but soil quality often declines concurrently, with an accompanying decline in absolute CO_2 emissions (Cannell *et al*., 1999). Oxidation may account for 35-100% of total peat subsidence, with a common assumption of 50% oxidation loss (Leifeld *et al*., 2011). Average subsidence rates in the East Anglian fens since the early 20th century are reported at between 0.27 and 3.09 cm yr^{-1} (Richardson and Smith, 1977; Hutchinson, 1980), while the area under arable production was estimated to be losing 1.10 to 1.48 cm yr^{-1} between 1982 and 2004 (Dawson *et al*., 2010). It is unclear what proportion of these losses were attributable to compression, consolidation and oxidation, or to other losses such as wind and water erosion, leaching, and crop adherence. In particular, the use of subsidence rates as a proxy measure for oxidative CO_2 loss must be used with some caution, as its reliable use is dependent on site-specific conditions such as timing of initial and subsequent drainage events, soil BD and SOC. Even if these factors are known, it should be used only under steady state conditions, and even then as an approximate measure of CO_2 loss only (Couwenberg and Hooijer, 2013; Hooijer *et al*., 2012).

Methane emissions from undrained organic soils usually exceed those from mineral soils (Le Mer and Roger, 2001). Conversely, drained and cultivated Histosols tend to experience a sustained decline in CH_4 emissions shortly after drainage (Kasimir-Klemedtsson *et al*., 1997). Most arable peat soils are net sinks for, or negligible sources of CH_4 (Table 2.1.; Bussell *et al*., 2010; Maljanen *et al*., 2010).

Emissions of N_2O from cultivated organic soils tend to be considerably higher than from cultivated mineral soils (e.g. Bouwman *et al*., 2002; Flessa *et al*., 1998). Uptake of N_2O by cropped soils has also been documented, including on peat soils; however, uptake is usually restricted to short-lived incidents rather than a net annual uptake (Chapuis-Lardy *et al*., 2007). Estimated net annual fluxes from arable peat soils vary widely, between 1.8 and 48.9 ± 3.3 t CO_2-e ha^{-1} yr^{-1} (Table 2.1.).

Accounts of C and N losses from arable peats soils via leaching and erosion are scarce, while estimates of loss from peat adherence to crops are unknown (e.g. Evans *et al*., 2011). Between 20% and 25% of all N added to terrestrial ecosystems may be lost via rivers (Mulholland *et al*., 2008). The proportion of N lost to waterways that is eventually transformed to indirect N_2O emission is subject to a great deal of uncertainty (IPCC, 2007b). Ditches and gullies may be particular hotspots for emissions in semi-natural and cultivated peatlands, accounting for a large percentage of soil surface emissions, or even exceeding them in some cases. This is particularly true of CH_4 emissions, which may exceed c. 42 t CO_2-e ha^{-1} yr^{-1} (IPCC, 2013c; McNamara *et al*., 2008; Schrier-Uijl *et al*., 2008). One recent study of a catchment in East Anglia comprising mineral and peat soils, estimated that 86% of indirect N_2O emissions originated from drainage channels, equating to c. 14.6 t CO_2-e ha^{-1} yr^{-1} (Outram and Hiscock, 2012). Soil C loss via wind erosion is potentially substantial, although apparently smaller than losses to waterways, with peak soil losses in the east of England estimated at > 3 t ha^{-1} (Dawson and Smith, 2007).

Table 1.2. Annual GHG emissions from cultivated Histosols in boreal and temperate zones.

GHG	Land use	Location	Lat, Long	Climatic zone	Peat SOM%	Peat C/N	Peat bulk density g cm^{-3}	Annual emission t CO_2-e ha^{-1} yr^{-1}	Reference
CO_2	Drained for crops	Finland / USSR	*n.d.*	**BS**	*n.d.*	*n.d.*	*n.d.*	8.0 ^{UR}	Gorham (1991) [1]
	Barley	N. Europe	*n.d.*	**B** [2]	*n.d.*	*n.d.*	*n.d.*	17.7 ± 1.16 ^{NEE}	Maljanen *et al*. (2010) [1]
	Bare soil	Finland	66°55'N, 23°51'E	**BS**	40 to 75	*n.d.*	0.27 to 0.50	25.3 to 28.9 ^{R}	Maljanen *et al*. (2004)
	Barley	Finland	66°55'N, 23°51'E	**BS**	40 to 64	31	0.38 to 0.45	30.4 ^{R}	Maljanen *et al*. (2004)
	Lettuce	UK	52°32'N, 0°28'E	**TO**	62 [3]	17	*n.d.*	10.6 to 28.3 ^{NEE}	Morrison *et al*. (2013)
	Cereals	Sweden	58°18'N, 13°55'E	**TO**	*n.d.*	*n.d.*	*n.d.*	20.0 ^{R}	Kasimir-Klemedtsson *et al*. (1997)
	Corn, wheat	USA	41°0'N, 82°84'W	**TS**	56 to 58	15	0.52 to 0.57	69.3 to 83.4 ^{R}	Elder and Lal (2008)
N_2O	Bare soil	Finland	62°40'N, 30°50'E	**BS**	*n.d.*	19	*n.d.*	1.8	Kasimir-Klemedtsson *et al*. (1997)
	Potato	Finland	60°49'N, 26°01'E	**BS / TS**	41 [3]	21	*n.d.*	4.9	Regina *et al*. (2004)
	Barley	N. Europe	*n.d.*	**B** [2]	*n.d.*	*n.d.*	*n.d.*	5.3 ± 1.6	Maljanen *et al*. (2010)
	Lettuce, onion, celery	Canada	45°09'N, 73°40'E	**TS**	64 [3]	18	*n.d.*	1.8 to 19.6	Rochette *et al*. (2010)
	Cereals	Sweden	58°18'N, 13°55'E	**TO**	*n.d.*	*n.d.*	*n.d.*	4.9 ± 3.4	Kasimir-Klemedtsson *et al*. (1997)
	Potato, maize	Germany	48°69'N, 11°15'E	**TO**	59 [3]	14	0.37	7.6	Flessa *et al*. (1998)
	Arable	Denmark	55°79'N, 11°65'E	**TO**	*n.d.*	*n.d.*	*n.d.*	11.4	De Klein *et al*. (2001) [1]
	Potato, rye, grass	Germany	48°67'N, 11°22'E	**TO**	72 [3]	15	0.42	27.5	Flessa *et al*. (1998)
	Corn, wheat	USA	41°0'N, 82°84'W	**TS**	56 to 58	15	0.52 to 0.57	47.1	Elder and Lal (2008)
	Carrots	Sweden	58°20'N, 13°30'E	**TO**	78	n.d.	0.29	48.9 ± 3.3	Weslien *et al*. (2012)
CH_4	Barley	N. Europe	*n.d.*	**B** [2]	*n.d.*	*n.d.*	*n.d.*	-0.01 ± 0.04	Maljanen *et al*. (2010) [1]
	Bare soil	USA	41°0'N, 82°84'W	**TS**	56 to 58	15	0.52 to 0.57	-0.1	Elder and Lal (2008)
	Cereals	Sweden	58°18'N, 13°55'E	**TO**	*n.d.*	*n.d.*	*n.d.*	0.02	Kasimir-Klemedtsson *et al*. (1997)
	Corn, wheat	USA	41°0'N, 82°84'W	**TS**	56 to 58	15	0.52 to 0.57	0.02 to 0.2	Elder and Lal (2008)

[1] Data drawn from a review paper. [2] As described in the introduction to the review paper; not described for individual studies. 3 Calculated from soil C%, by multiplying by 1.724, after Howard (1965). Climatic zones: BS = boreal subcontinental; B = boreal; TO = temperate oceanic; TS = temperate subcontinental. Classified according to Fischer et al. (2012). Measurement approaches: ^{UR} Unknown (assumed soil respiration only); ^{R} Soil respiration only; ^{NEE} Net ecosystem exchange.

1.4. Controls on greenhouse gas emissions from cropped peat soils

Greenhouse gas production, transport and consumption processes are regulated by complex spatial and temporal interactions between soil properties, and by additional environmental properties at the soil-atmosphere interface (Li, *et al*., 1992a; Li, 2007). Key soil characteristics influencing the magnitude and pattern of GHG fluxes include redox potential (Eh), moisture and aeration, temperature, pH, and substrate concentration gradients (e.g. Dalal *et al*., 2003; 2008; Le Mer and Roger, 2001; Li, 1992a; 2007; Smith *et al*., 2003). Soil properties are in turn affected by climatic factors (primarily rainfall, air temperature, solar radiation, and wind speed), vegetation properties, and farm management operations (e.g. soil tillage, fertiliser application) (Li, 2007). The following sections discuss the principal controls on the relative magnitudes and patterns of CO_2, N_2O and CH_4 production and consumption, where possible drawing from literature specific to cropped peatlands.

1.4.1. Environmental factors

1.4.1.1. Soil redox potential

Soil redox potential (Eh), in tandem with electron acceptor and C and N source availability, acts as the primary determinant of GHG production ratios from soils (Li, 2007). Generation and consumption of each of the GHGs is favoured within a specific soil Eh range, corresponding to different levels of soil aerobicity. The production of CO_2 is favoured in fully aerobic soil (Eh > +350 to +500 mV), with N_2O being generated in increasingly anaerobic microsites (Eh = -200 to +500 mV), and CH_4 emission occurring under fully anaerobic conditions (Eh < -200 to -100 mV; Li, 2007; Mitsch and Gosselink, 2000). Oxidation of CH_4 can also occur under low Eh conditions, according to the relative prevalence of electron acceptors.

1.4.1.2. Soil moisture and aeration

Soil moisture and aeration are inversely proportional to each other, and are closely related to soil Eh. Hence, they are key determinants in the pattern and magnitude of CO_2, N_2O and CH_4 fluxes.

Soil respiration rate is known to be responsive to soil water content, and particularly to water-filled pore space (WFPS) (Smith *et al*., 2003). Emissions of CO_2 decline when soil microbial activity is inhibited, that is, under very dry (< 20% WFPS) or very wet (anaerobic) conditions (Freeman *et al*., 1993; Smith *et al*., 2003). Soil moisture content interacts strongly with temperature, with CO_2 emission showing a distinct suppressed response to temperature in saturated soil, including in peats (e.g. Kechavarzi *et al*., 2007; Lloyd, 2006). Response to temperature also declines when soil WFPS is < 20% (Smith *et al*., 2003).

Methane production and consumption often occur simultaneously in different soil microsites, with the net measured efflux dependent on the balance between these two processes (Dalal *et al*., 2008). Continuous anaerobic conditions lasting more than one year are sometimes required to produce a net emission (Dalal *et al*., 2008; Oomes *et al*., 1997). Conversely, CH_4 oxidation is more common when soil is drier; although even in wet soils, absorption of CH_4 can occur in localised drier (aerobic) pockets, or pockets of anaerobic oxidisers (Dalal *et al*., 2008). Consequently, drained arable peat soils tend to yield considerably lower CH_4 emissions than semi-natural peatlands (Kasimir-Klemedtsson *et al*., 1997). Soil moisture also affects the dominant CH_4 transport processes occurring at a given time, with diffusion more common under drier conditions, and ebullition and plant-mediated transport more usual in saturated soils (Smith *et al*., 2003).

The influence of soil moisture and aeration on N_2O emission is well defined in the literature, and principally relates to its effects on the dominance of nitrification or denitrification activity (Dalal *et al*., 2003; Linn and Doran, 1984). In well-aerated soil microsites (WFPS < 40%), nitrification NO production dominates, and N_2O production is low. Between 55 to 60% WFPS, N_2O emission from nitrification rapidly increases, with denitrification taking over as the dominant process at 60 to 70% WFPS. As soils become increasingly anaerobic (> 80 to 90% WFPS), denitrification tends to follow through to completion, and N_2O production is replaced by N_2 emission. Thus, N_2O is optimally produced at 60 to 70% WFPS, although the exact threshold at which this occurs depend on soil texture and structure (Linn and Doran, 1984).

1.4.1.3. Soil temperature

Emissions of CO_2, N_2O and CH_4 all show a general positive correlation with soil temperature. For CO_2, this relationship is frequently described exponentially, by application of the Q_{10} value to soil microbial and root respiration (Smith et al., 2003). Considerable variability in Q_{10} values have been observed between sites (estimated at between 2.3 and 3.0 on peat soils), and depend upon the depth of temperature measurement, soil type, and interactions with soil nutrient status including fertiliser inputs (e.g. Jauhiainen et al., 2014; Smith et al, 2003). For example, the lower thermal conductivities and diffusivities found in peat soils compared to mineral soils, result in peat soils holding heat for longer. Temperature-driven CH_4 production also shows considerable variability, ranging from a Q_{10} of 1.1 to 28.0 (averaging 4.0), and optimum production at c. 35°C (Dalal et al., 2008). Conversely, CH_4 consumption shows a much lower sensitivity to temperature, with an effect on consumption only observable at temperatures of < 10°C. The temperature response of N_2O production varies according to soil type, microbial communities, land use, and climatic zone (Dessureault-Rompre et al., 2010; Farquharson and Baldock, 2008). A recent study by Jauhiainen et al. (2014) showed that the negative correlation between soil temperature and N_2O emission produces a stronger response shown on fertilised compared to unfertilised sites, possibly reflecting the total nitrification and denitrification activity in the soil. Soil temperature influences the ratio of nitrification to denitrification end products (N_2O and N_2), with maximum N_2O emission rates occurring at 20-35°C; at higher temperatures, N_2 emission becomes dominant (e.g. Braker et al., 2010).

1.4.1.4. Soil pH and salinity

The optimum pH range for CO_2, CH_4, and N_2O production all approximate to a neutral pH (Dalal *et al*., 2003; 2008). Methanogenesis, methanotrophy, nitrification and denitrification are all possible outside of this range, with CH_4 production and oxidation tending to decline, and N_2O tending to increase, with an increase in acidity. Increased salinity generally inhibits microbial activity, reducing emissions of GHGs (e.g. Dalal *et al*., 2003; 2008). As cultivated fen peats tend to have a pH range of c. 5 to 7, and a relatively low salinity, these factors are unlikely to be limiting to GHG production in horticultural systems on peat soils.

1.4.1.5. Soil substrate availability and quality

The greater emissions of CO_2 and N_2O associated with cultivated peatlands, compared with mineral soils, are largely a function of the availability of the SOM substrate, with emissions tending to increase with SOM content (e.g. Bellamy *et al*., 2005; Kasimir-Klemedtsson *et al*., 1997). Freshly drained and cultivated peats commonly yield higher emissions than historically drained peats, reflecting an increased availability of labile SOM after initial soil disturbance (Dawson and Smith, 2007). Emissions of N_2O appear to be enhanced by increasing SOC, but only up to a point, after which an increase in soil C/N ratio supresses denitrification N_2O efflux, shifting towards dominant N_2 production instead (Dalal *et al*., 2003).

Other soil nutrients may additionally affect GHG emissions, although interactions with soil microbes are not always straightforward. A deficiency of essential plant nutrients (e.g. K, P, Mg) prevents plant uptake of ammonium (NH_4^+) and nitrate (NO_3^-), leaving a greater soil N reserve to provision N_2O emissions (Dalal *et al*., 2003). The relationship between soil nutrient status and CH_4 production is more complex. For example, adding Ni, Co and Fe to deficient soils increases CH_4 production, but adding these where levels are sufficient reduces emissions, and adding Cu increases CH_4 oxidation. The response of CH_4 emission or absorption is related to whether the micronutrients present act as alternative electron acceptors, for example, where Fe^{3+} is present, CO_2 is produced instead of CH_4 (Dalal *et al*., 2008; Le Mer and Roger, 2001).

1.4.1.6. Soil diffusivity

Soil diffusivity is affected by soil texture, structure, and moisture content. High soil diffusivity (e.g. in well-aerated soil), results in lower CH_4 emissions, because anaerobic conditions are prevented and some CH_4 is oxidised in transit to the soil surface from deeper layers. Diffusivity has a greater influence on CH_4 emission in well-aerated unvegetated systems, since in vegetated anaerobic soils, transport by plant aerenchyma tends to dominate (Dalal *et al*., 2008; Le Mer and Roger, 2001). In contrast, N_2O emission is enhanced where soil diffusivity is high, since nitrification tends to dominate, and N_2O dominates over N_2 denitrification products (Dalal *et al*., 2003).

1.4.2. Farm management operations

1.4.2.1. Soil tillage

Soil tillage affects GHG emissions through its influence on soil aeration and substrate availability. The effects of tillage on soil compaction vary according to the tillage approach used. For example, while conventionally tilled (CT) areas of soil show increased soil aeration and reduced soil moisture retention, soil compaction can be increased along farm traffic routes (Bessou *et al*., 2010; Mummey *et al*., 1998). Controlled-traffic farming (CTF) can reduce overall field compaction compared with random traffic farming (RTF) (Vermeulen and Mosquera, 2009). Tillage increases substrate availability, by exposing SOM to decomposer microorganisms, and turning any residues into the ground where they might be more readily decomposed. As a result, SOM content generally declines over time (Elder and Lal, 2008; Cerri *et al*., 2007).

Soil CO_2 emissions are usually greater from tilled soils than from no-till systems, being substantially related to the degree of soil disturbance (Elder and Lal, 2008). Soil C storage is promoted in NT systems (Cerri *et al*., 2007), particularly where direct mulching or incorporation of crop residues are also employed (Ahmad *et al*., 2009). Despite this, sequestration may take some time to achieve, and is on average slower in drier climatic areas than in humid zones. Further, sequestration in restored peat soils rarely replicates the rate of C accumulation observed in virgin peat (Parish *et al*., 2008).

Tillage simultaneously increases uptake from, and efflux to, the atmosphere of CH_4, by increasing soil surface diffusivity (Dalal *et al*., 2008), and in the long term, reduces oxidation by disrupting soil microbes (Elder and Lal, 2008). Conversely, where no-till (NT) is employed, soil surface diffusivity is likely to be reduced, resulting in suppressed CH_4 emission and oxidation (Elder and Lal, 2008; Six *et al*., 2004). Elevated soil moisture enhances this effect, for example with subtropical rice paddies showing 28% higher emissions in CT than NT (Ahmad *et al*., 2009), compared to a temperate corn-wheat rotation (no significant difference, with a low flux in both NT and CT treatments; Elder and Lal, 2008).

The observed effects of tillage on N_2O fluxes vary greatly, with high N_2O emissions reported from both tilled and untilled soils. Many authors report comparatively higher peaks and duration of emissions in untilled cropped soil, owing to elevated soil moisiture levels associated with compaction (e.g. Bessou *et al*., 2010; Vermeulen and Mosquera, 2009). In contrast, Chapuis-Lardy *et al*. (2007) suggested that deep ploughing could release N_2O generated and stored in deeper soil layers. While Six *et al*. (2004) suggested that over a 20-year period in temperate zones, the difference may become negligible, this may not be the case for peat soils, where disturbance results in sustained high mineralisation rates compared to undisturbed peats (Bussell *et al*., 2010).

1.4.2.2. Irrigation

Irrigation application tends to reduce net CO_2 emission, and encourages SOC retention (e.g. of 0.79 ± 0.75 t C ha^{-1}, a threefold increase in irrigated compared to rain fed crops; Liebig *et al*., 2005; Mosier *et al*., 2005).

Methanotrophs remain viable once activated under anaerobic conditions, even if subject to dry periods thereafter (Le Mer and Roger, 2001). Despite this, re-activation often requires a prolonged period of re-wetting (Oomes *et al*., 1997). This is supported by the observations of

Scheer *et al*. (2008), who found that CH_4 emissions were significant from flooded rice fields, but not from other irrigated crops; and Mosier *et al*. (2005) who found that even when applied irrigation increased soil WFPS to almost constant field capacity, soils were still only a small net source of methane.

Episodic N_2O pulses are commonly associated with rapid increases in soil moisture, for example through rainfall or irrigation events (Li *et al*., 1992a). Peak emissions of 1100-1900 µg N m^{-2} h^{-1}, 1-2 days after dry soil irrigation were observed by He *et al*. (2007; 2009) and Adviento-Borbe *et al*. (2007), corresponding to a soil WFPS increase from 38-55% to 100%. Irrigated crops can yield annual emissions of up to five times that of rain-fed crops (Liebig *et al*., 2005; Mosier *et al*., 2005), but the effect is less pronounced in wetter years (Horvath *et al*., 2010). Irrigation method is an important influence on fluxes: for example, using drip irrigation can substantially reduce emissions (by 70-75%), when compared to furrow irrigation (Sanchez-Martin *et al*., 2008; 2010).

1.4.2.3. Water table raising and drainage

Emissions of CO_2 generally increase with water table lowering (i.e. drainage), with organic soils becoming a net CO_2 source after drainage (Lal, 2011), mainly through the removal of limits on aerobic decomposer activity (Rochette *et al*., 2010). This is particularly the case in peatlands, where a ready supply of SOM substrate for decomposition exists (Smith *et al*., 2003). For example, Furukawa *et al*. (2005) found CO_2 hourly emissions changed from 30 to 266 mg C m^{-2} h^{-1} with a land use gradient from submerged tropical crops to a deeper water table, and predicted a 50% increase in CO_2 emission with a 10 cm lowering of the water table.

While submerged soils tend to generate CH_4 (David *et al*., 2009), a review of 165 papers by Mander *et al*. (2010) indicated that drained peatlands emit considerably less CH_4 than natural peatlands (averaging 3.7 kg ha^{-1} yr^{-1} and 106 kg ha^{-1} yr^{-1} respectively). Nevertheless, methanotroph reactivation in saturated soil microsites may result in emission of CH_4 pulses following temporary elevations in water table.

In a review of natural and modified peat lands, Mander *et al*. (2010) cited primary N_2O sources as drained (partly restored) peatlands (7.2 kg ha^{-1} yr^{-1}), conventional arable lands on fen soils (4.5 kg ha^{-1} yr^{-1}); and drained fens (2.1 kg ha^{-1} yr^{-1}). Tile-drained, sub-irrigated soils may emit double the quantity of N_2O than from free-draining soils, both in the short term (4.4 kg ha^{-1} d^{-1} and 2.3 kg ha^{-1} d^{-1} respectively), and in the longer term (12.9 kg ha^{-1} $season^{-1}$ and 5.8 kg ha^{-1}

season^{-1} respectively; David *et al*., 2009). Effluxes of up to 60 kg ha^{-1} during a growing season have been reported for sub-irrigated land, possibly as a cumulative effect of wet-dry cycling.

1.4.2.4. Fertiliser application

The effect of fertiliser addition on emissions depends on rate and timing of application, and the type of fertiliser applied (whether organic or mineral), as well as the growth stage of the crop, soil moisture content, and temperature at the time of and following application.

Adding organic fertiliser to soils (e.g. sewage sludge or slurry) generates CO_2; an increase in application rate positively correlates with total emission (Huang and Tang, 2010). This response can be suppressed under flooded conditions (e.g. in a rice paddy system; Ahmad *et al*., 2009). While Adviento-Borbe *et al*. (2007) suggest that incorporating inorganic N fertiliser into soils with plant residues during the autumn may increase SOC sequestration over the winter period in a mineral soil, this response may differ in peat soils. Addition of surplus N to soils with high SOM content, may enhance decomposition rates and deplete SOC, a phenomenon known as ‘priming’ (Bingeman *et al*., 1953).

Commonly, adding fertiliser to soils has no effect on CH_4 emissions (e.g. Dunmola *et al*., 2010), except under flooded conditions, where emissions may be enhanced (e.g. Ahmad *et al*., 2009). Fertiliser composition influences the duration and magnitude of enhanced CH_4 emissions, including determining whether increasing the application rate affects emissions (Dalal *et al*., 2008). This may be a function of competing electron acceptors (Hastings *et al*., 2010), or fertiliser instigating a decline in soil pH (Ahmad *et al*., 2009). Adding other macronutrients (e.g. P and K) to deficient soils can reduce emissions. Adding organic fertilisers with a high C/N ratio and low lignin/N ratio (e.g. as in fresh manure, slurry, or fresh plant residues), can enhance emissions up to 50 times more than when an equivalent rate of urea or ammonium sulphate is added to the soil (Dalal *et al*., 2008).

Emissions of N_2O in response to fertiliser application are highly episodic, with the magnitude and response lag determined by rainfall and temperature, and by the type of fertiliser applied (Jacinthe and Dick, 1997). Emission response time can vary between 3 and 14 days, with a shorter response time associated with higher application rates and coincidence with rainfall events. Evidence suggests that the previous year’s fertiliser application can influence emissions even from fallow plots in the current year, with sometimes prolonged periods of elevated N_2O emissions being observed (e.g. Huang and Chen, 2009; Ma *et al*., 2010).

1.4.2.5. Crop type

Growing high-yielding crops maximises crop carbon sequestration potential (Adviento-Borbe *et al*., 2007), but this advantage may be negated by such crops having short growing periods and thus increasing the frequency of soil cultivation, resulting in elevated cumulative CO_2 emissions (Predotova *et al*., 2010a). Planting perennial crops or reverting to permanent semi-natural vegetation cover generally increases SOC accumulation and reduces net emissions (e.g. Liebig *et al*., 2005).

In soils emitting CH_4, increasing vegetation cover reduces emissions, provided that the vegetation is not aerenchymous (Dalal *et al*., 2008; Le Mer and Roger, 2001). However, this relationship is not straightforward, and the advantages of crop moisture drawdown should be balanced with potentially greater root exudate production or post-harvest residue inputs (both of which provide substrates for methanogenesis).

Measured N_2O emissions are often greatest in the early stages of crop growth (e.g. He *et al*., 2007) – a possible effect of lack of nitrate absorption, or a large soil moisture surplus when plants are small (Liebig *et al*., 2005). Alternatively, it may be associated with an increase in stomatal absorbance of N_2O and NO absorbance as crop canopy cover increases (Pang *et al*., 2009). Overall N_2O emissions can be reduced by growing nitrate-demanding crops, with a high lignin and polyphenol content, and high nutrient use efficiency (Adviento-Borbe *et al*., 2007; Hellebrand *et al*., 2008; Kavdir *et al*., 2008).

1.4.2.6. Crop residue management

The overall effect of plant residues on GHG emissions depends upon residue quality (C/N ratio, decomposition rate) and management, as well as the environmental and soil conditions during the period of residue cover.

Crops leaving a higher proportion of plant residue can increase C sequestration compared to lower-biomass crops, for example maize compared to soybean (Adviento-Borbe *et al*., 1997). The resulting net reduction in NEE is not always proportional to the total residue quantity applied; residue quality (litter C/N ratio) is also important, strongly affecting soil mineralisation rate (Komatsuzaki and Ohta, 2007). Soil respiration may also be suppressed by employing surface mulching, and choosing crops with high root and residue lignin content and root volume (Liebig *et al*., 2005). Residue retention may take 5 to 20 years to reach maximum levels after first application (Komatsuzaki and Ohta, 2007), but the effect is reversed very rapidly once residue input and retention ceases (Oenema *et al*., 2001).

Retaining residues in-field can substantially increase CH_4 emission, particularly if incorporated (Le Mer and Roger, 2001). This effect may be partially attributable to soil disturbance, as retaining in situ stubble slows the rate of C loss from the soil (Howden and O'Leary, 1997), and surface mulching with straw may reduce emissions by preventing residue decomposition (Harada *et al*., 2007). A low C/N ratio, low C content, and high lignin/N ratio of residues reduce CH_4 efflux, so composting residues before incorporation can result in non-significant emissions (Le Mer and Roger, 2001; Dalal *et al*., 2008). Applying residues when soil is warm and flooded, results in significantly higher emissions than application under cool, moist conditions (Dalal *et al*., 2008).

There is some evidence that retaining N-rich residues enhances N_2O emissions, for example Dobbie and Smith (2003) found that cereal fields yielded a higher background emission than grassland (0.31 kg N ha^{-1} yr^{-1} and 0.22 kg N ha^{-1} yr^{-1} respectively), probably due to straw retention providing an N source. Litter distribution can significantly affect the magnitude of N_2O emissions, with aggregated N-rich litter yielding fluxes of up to eight times higher than evenly distributed litter (Loecke and Robertson, 2009).

1.5. Methods available for direct measurement of GHG emissions

Techniques for directly measuring fluxes of CO_2, N_2O and CH_4 may be broadly categorised into micrometeorological methods and chamber techniques. When comparing GHG measurement methods, attention should be paid to potential sources of error inherently embedded in the approach, as well as error resulting from equipment deployment and the processing and interpretation of GHG data. A number of detailed reviews and guides have been published, relating the relative utilities and drawbacks of each method (e.g. de Klein and Harvey, 2013; Denmead, 2008; McGinn, 2006). The salient points are discussed briefly here.

1.5.1. Micrometeorological methods

Micrometeorological techniques measure the dynamics of air mass movements over the ecosystem of interest, to capture gas exchange between the atmosphere and land surface. While various specific methods exist, only a few are widely used for the purpose of measuring GHGs. One of the most commonly used techniques is that of eddy covariance, which monitors the rate of vertical movement of air pockets, and the gas concentration within them, over a given period of time. Instrumentation typically consists of a mast equipped with GHG sensors and an anemometer at the required height above ground, as well as additional meteorological equipment.

The eddy covariance technique is well suited to measuring integrated, system-wide, larger-scale fluxes, with spatial scale theoretically varying from c. 0.5 ha up to several million ha. Its continuous deployment provides uninterrupted measurements incorporating the full range of temporal variability in fluxes, and usually yielding more accurate absolute estimates of fluxes over an extended period than chamber techniques may supply. Vegetation of any height can relatively easily be included in measurements. The primary disadvantage of micrometeorological techniques is that instrumentation is comparatively expensive and delicate, and technically and logistically difficult to set up – this may restrict application in some environments and circumstances. Replication is often not possible, so the technique is not suitable for identifying fine-scale controlling variables and processes influencing emissions. In the case of N_2O flux measurements, techniques are not fully developed or applicable to all circumstances.

1.5.2. Chamber techniques

Chambers are the most widely used equipment for measuring gas fluxes from soils. They work by enclosing a predetermined volume of air (chamber headspace volume) within a vessel placed over the ground, where the degree of interaction with the atmosphere is controlled for a given time period; samples of headspace gases are then drawn off at intervals for analysis. In principle, the chamber method magnifies concentration changes of trace gases and should yield a period of linear concentration increase which allows flux calculation through the use of calibration curves. Chamber design may also include installation of permanent collars into the ground to minimise soil disturbance effects on measurements, and in some cases automatic opening and closure is installed. Two general types of chamber exist: *steady-state (SS)* or open (dynamic) design, and *non-steady-state (NSS)* or closed design. The former operates by circulating gas through the headspace, creating a near-constant headspace concentration, whereas the latter induces an increasing headspace gas concentration.

Chamber approaches are well suited to measuring fine-scale distinctions between different treatments, or linking emissions to environmental variables so that processes may be understood. Chambers are highly customisable, and easy to construct, install, transport and use, so are suitable for application to a wide range of environments and circumstances. Deployment and sample analysis can be highly labour and resource-intensive. This problem may be overcome to some extent by deploying automated chambers, which are kept in-situ and facilitate reduced labour input, whilst enabling frequent sampling (useful for measuring short term order-of-magnitude flux changes in specific situations). For landscape-scale measurements, there are often problems with scaling up chamber-derived measurements, associated primarily with the degree of temporal and spatial heterogeneity within the ecosystem of interest. The accuracy of up-scaled emissions estimates can be improved by carefully incorporating representative spatial and temporal variability within the experimental design. Sources of temporal variability are usually addressed through appropriate sampling strategies incorporating measurements over cycles of diurnal and seasonal variation, and ideally on a multi-annual basis. Including inter-annual variation is important as weather and management conditions over one annual cropping cycle may be atypical of longer-term trends, potentially biasing flux estimates. Capturing spatial variation corresponds to deploying a sufficient number of chambers within each identified land use type, cropping type, or soil type.

1.6. Mitigation of greenhouse gas emissions from horticultural peat soils

While many authors have published reviews of potential GHG mitigation measures (MMs) relating to cropped agroecosystems, the evidence base for mitigating GHGs from horticultural peat soils remains minimal. Table 1.3. provides a compilation of candidate MMs from key scientific and grey literature sources (Akiyama *et al*., 2004; Asgedom and Kebreab, 2011; Baldock *et al*., 2012; Barnes *et al*., 2010; Cole *et al*., 1997; Dalal *et al*., 2003, 2008; de Klein *et al*., 2001; Fitton *et al*., 2011; Garnett, 2011; Gregorich *et al*., 2005; Henault *et al*., 2012; Jones *et al*., 2010; Jones, *pers. comm.*; Lal, 2008; Lal *et al*., 2011; Le Mer and Roger, 2001; MacLeod *et al*., 2010; Mikkelsen, 1994; Moran *et al*., 2008; Rounsevell and Reay, 2009; Smith *et al*., 2008; Synder *et al*., 2009). Mitigation options can be generally classified as one of three types of measure, according to their underlying mechanism: reducing emissions, enhancing removals, or avoiding (displacing) emissions (Smith *et al*., 2008). A further categorisation was applied in Table 1.3. to facilitate merging of similar MMs.

The UK has a relatively well established GHG mitigation policy, in global terms, as well as a framework of evolving agri-environmental schemes in place, which might be modified to accommodate a future emphasis on GHG mitigation (Norse, 2012; Reed *et al*., In Press). There are also some policies in place protecting peatlands, but not specifically against degradation under agricultural use (Bain *et al*., 2011). Greater connectivity between existing policies would facilitate delivery of GHG mitigation targets. Several recent studies have been conducted in the UK agricultural sector examining the potential efficacy, cost, and barriers to uptake of a short list of GHG MMs (Barnes *et al*., 2010; Jones *et al*., 2010; Moran *et al*., 2008; MacLeod *et al*., 2010). While none of the short listed measures related specifically to horticultural peat soils, the methods used could be easily adapted to use with the horticultural sector. The primary knowledge gap relates to a lack of information on baseline GHG emissions from peat soils under horticultural use, and the potential reduction in emissions when MMs are applied.

Table 1.3. Long list of candidate mitigation measures (MMs) drawn from the literature, with the potential to reduce or offset emissions from horticultural peat soils in the UK.

No.	Category	Intervention	Source
1	Nutrient mgt.	**Use frequent soil and crop nutrient status testing and nutrient management planning** to optimise farm N budgeting	2, 4, 6, 8, 13, 15, 20, 23
2	Nutrient mgt.	**Improve spatial fertiliser N placement** to avoid loss of excess N (e.g. use shallow injection or foliar feeds)	2, 5, 6, 8, 15, 16, 17, 18, 20, 22, 23
3	Nutrient mgt.	**Improve timing of fertiliser N application** to avoid loss of excess N (e.g. apply just before crop growth; avoid application for 5 days after heavy rain; use split timing application)	2, 3, 4, 5, 6, 8, 9, 10, 13, 16, 18, 20, 22, 23
4	Nutrient mgt.	**Reduce fertiliser N dosage** to avoid loss of excess N (e.g. avoid applying fertiliser to legumes)	2, 4, 6, 8, 11, 13, 16, 18, 20, 22, 23
5	Nutrient mgt.	**Apply nitrification inhibitors to soil when applying ammonium-based fertiliser N** (e.g. mineral N, manure)	2, 3, 5, 6, 7, 8, 9, 10, 12, 16, 18, 20, 21, 22, 23
6	Nutrient mgt.	**Apply urease inhibitors to soil with fertilisers**	2, 3, 6, 18, 20
7	Nutrient mgt.	**Avoid or delay applying pesticides that interact with fertilisers to increase GHG emissions** (e.g. bromoxynil or methomyl)	7, 17
8	Nutrient mgt.	**Apply mineral N fertiliser using drip or sprinkler irrigation**, to optimise infiltration and N uptake	3, 6, 15
9	Nutrient mgt.	**Increase reliance on nitrogen-fixing crops** to replace mineral N fertiliser	2, 3, 5, 6, 8, 9, 10, 11, 15, 18, 20, 22, 23
10	Nutrient mgt.	**When applying manure on moist soils, include ammonium sulphate or urea** with manure application, to reduce CH_4 emission	15, 17
11	Nutrient mgt.	**Use farm-yard manure as a fertiliser** to encourage soil CH_4 oxidation	17, 20, 22
12	Nutrient mgt.	**Avoid applying farm-yard manure and liquid manures to soils**, to reduce potential N_2O emissions	6, 8, 9, 11, 20, 23
13	Nutrient mgt.	**Avoid applying mineral N to soils within 5 days of applying manures**	4, 13, 18, 20
14	Nutrient mgt.	**Grow crops with a lower N required**	2, 6, 12, 18, 20, 22
15	Nutrient mgt.	**Grow crops that produce natural nitrification inhibitors**	18
16	Water mgt.	**Optimise irrigation system efficiency** to keep soil moist but not saturated, control soil N release, and reduce pump fuel use	3, 5, 6, 7, 8, 15, 17, 18, 20, 21, 23
17	Water mgt.	**Optimise ground water levels** at each stage of the annual crop cycle (e.g. maintain shallow water table for longer; maintain water table above depth of drainage pipes)	3, 6, 7, 8, 17, 18, 20, 22, 23
18	Water mgt.	**Select crops that maximise water use efficiency**	3, 21
19	Water mgt.	**Apply gypsum or ammonium sulphate when soils are flooded**, to provide alternative electron acceptors for methanogens	3, 7, 17
20	Water mgt.	**Maintain or improve field drainage network** to optimise drainage and sub-surface irrigation efficiency and reduce leaching	6, 7, 18, 20, 21, 22, 23
21	Crop mgt.	**Grow crops that maximise both crop yield efficiency and residue return to the soil**	2, 9, 11, 16, 18, 20, 22, 23
22	Crop mgt.	**Increase the number of crops planted per year**, matching cropping intensity to land quality on different parts of the farm	2, 3, 6, 10, 16, 18, 20, 21, 22, 23
23	Crop mgt.	**Grow crops with greater below-ground biomass** (roots, exudates and mycorrhizae)	2, 5, 9, 11, 15, 16, 17, 22, 23
24	Crop mgt.	**Grow crops that can convert N_2O to N_2 even at low N_2O concentrations** (e.g. some improved legume varieties)	12
25	Crop mgt.	**Choose crop varieties that minimise CH_4 emission** (e.g. with low levels of plant-mediated transport, and low C-rich residue return to soil)	7, 17, 22
26	Crop mgt.	**Change sowing and harvesting dates to optimise nutrient use efficiency**	21
27	Crop mgt.	**Diversify crop rotations** to optimise soil nutrient status and SOC return (e.g. include perennial crops or temporary pasture in rotations)	2, 3, 5, 6, 9, 10, 11, 15, 16, 21, 22
28	Fallow/ residue mgt.	**Avoid or shorten bare fallow periods** (e.g. grow fast-growing cover crops such as green manures)	2, 5, 6, 8, 15, 18, 20, 22, 23
29	Fallow/ residue mgt.	**Increase the use of perennial food crops**, to promote SOC retention	3, 5, 9, 11, 16, 18, 20, 21, 22
30	Fallow/ residue mgt.	**Retain crop residues on fields during fallow periods**, and allow to decompose before ploughing in	2, 6, 7, 8, 9, 11, 15, 16, 17, 18, 20, 22, 23

Table 1.3. Long list of candidate mitigation measures (MMs) drawn from the literature, with the potential to reduce or offset emissions from horticultural peat soils in the UK (*continued*).

No.	Category	Intervention	Source
31	Fallow/ residue mgt.	**Deep burial of crop residues after harvest**, to encourage SOC retention in deeper soil layers	11
32	Fallow/ residue mgt.	**Grow crops with lower C/N ratio or higher lignin content**, to reduce or slow down residue substrate supply for GHG production	7, 11, 17, 23
33	Tillage	**Eliminate tillage**	2, 5, 7, 8, 9, 10, 11, 15, 16, 18, 20, 21, 22, 23
34	Tillage	**Reduce tillage** (e.g. less frequent or shallower tillage; use direct drilling instead of transplanting)	2, 5, 7, 9, 10, 11, 17, 18, 20, 21, 22, 23
35	Tillage	**Avoid tillage and fertiliser application at the same time**	23
36	Soil mgt.	**Maintain optimal soil pH**, to optimise plant growth and reduce overall GHG emission	3, 6, 7, 12, 17, 18, 23
37	Soil mgt.	**Minimise soil compaction from wheeled operations**, to minimise N_2O and CH_4 emissions	3, 7, 11, 12, 18, 20, 21, 23
38	Soil mgt.	**Apply biochar to depleted SOC soils**	15,16, 21
39	Soil mgt.	**Avoid applying lime to soil** (e.g. apply oxides such as quicklime, slaked lime, CaO or MgO instead, to reduce CO_2 emissions from carbonates and manufacturing processes	23
40	Soil mgt.	**Apply polyacryamide fertiliser gels to soil** to reduce N_2O emissions and soil organic matter loss through wind erosion	14, 19
41	Soil mgt.	**Avoid applying fleece or other coverings to crops**, particularly when soil is moist, to minimise soil heating	3, 7
42	Soil mgt.	**Apply methane inhibitors** (e.g. bacterial inhibitors) to soils with high moisture and SOM content	10, 17, 21
43	Fossil fuel	**Minimise avoidable use of farm machinery** to reduce fossil fuel consumption	2, 5, 10, 21, 22
44	Fossil fuel	**Use alternative fuels to fossil fuels** (e.g. in machinery, and for on-farm processing and power and heat supply)	2, 5, 7, 10, 15, 16, 21, 22, 23
45	Fossil fuel	**Use more fuel-efficient machinery and equipment with better power ratings**	2, 10, 21, 23
46	Fossil fuel	**Avoid growing artificially heated protected crops** (heated greenhouse or heated poly-tunnel crops)	10
47	Fossil fuel	**Use integrated pest management** to reduce synthetic pesticide use and resource wastage through crop losses	10, 15
48	Sequestration	**Convert cropland to grassland**	2, 3, 6, 7, 9, 10, 15, 16, 21, 22
49	Sequestration	**Convert cropland to bio-energy or bio-fuel plantations**	2, 5, 15, 21, 22
50	Sequestration	**Restore cropland to native wetland vegetation** (e.g. wet woodland; lowland meadow; lowland heathland; reedbeds), using minimal cultivation	5, 15, 16, 21, 22
51	Sequestration	**Increase on-farm planting of trees and shrubs** (e.g. inter-row cropping; orchards; shelterbelts; buffer strips; on marginal land)	6, 9, 10, 15, 16, 21, 22
52	Sequestration	**Install buffer zones** between fields and watercourses, and within fields, to catch leached nutrients and SOC	7, 16, 21, 22
53	Sequestration	**Avoid additional draining and cultivation** of uncultivated wetlands	18, 20, 21, 22
54	Miscellaneous	**Use anaerobic digestion with energy recovery, or bio-fuel production**, to process organic wastes (e.g. crop residues)	2, 4, 5, 7, 10, 13, 15, 18, 21, 22, 23
55	Miscellaneous	**Harvest all crops in field irrespective of quality**, to reduce input resource wastage	14
56	Miscellaneous	**Use 'carbon trading' measures to offset emissions** (e.g. invest in off-farm wetland restoration or renewable energy schemes)	3, 10, 15
57	Miscellaneous	**Use precision farming** (e.g. GPS; annual machinery calibration) to apply chemicals and irrigation according to spatial variation in crop requirements, and to minimise compaction	4, 6, 13, 15, 16, 18, 20, 21, 22
58	Miscellaneous	**Use existing crop residues** (e.g. corn, wheat, barley) for producing long-lived bio-products	3, 22, 23

Sources: (1) Akiyama et al.*, 2004; (2) Asgedom and Kebreab, 2011; (3) Baldock* et al.*, 2012; (4) Barnes* et al.*, 2010; (5) Cole* et al.*, 1997; (6) Dalal* et al.*, 2003; (7) Dalal* et al.*, 2008; (8) de Klein* et al.*, 2001; (9) Fitton* et al.*, 2011; (10) Garnett, 2011; (11) Gregorich* et al.*, 2005; (12) Henault* et al.*, 2012; (13) Jones* et al.*, 2010; (14) Jones,* pers. comm.*; (15) Lal, 2008; (16) Lal* et al.*, 2011; (17) Le Mer and Roger, 2001; (18) MacLeod* et al.*, 2010; (19) Mikkelsen, 1994; (20) Moran* et al.*, 2008; (21) Rounsevell and Reay, 2009; (22) Smith* et al.*, 2008; (23) Synder* et al.*, 2009.*

1.7. Conclusions

The current global and UK knowledge base relating to GHG emissions from cultivated peat soils, and from horticultural use in particular, is poor. There is a critical need for quantification of emissions, in order to prioritise sectorial mitigation efforts with the aim of meeting overall UK emissions reduction targets. Uncertainty over the size of the UK peat soil carbon stock is further limiting estimates of current and potential future soil emissions and sequestration. The development of a coherent mitigation strategy is contingent upon bridging the knowledge gap between proposed mitigation measures, and their applicability to horticultural peat soils in terms of mitigation potential and likely adoption rates.

Chapter 2

Annual emission cycle of greenhouse gases from managed peat soils

Abstract

Greenhouse gas emissions from agriculture represented the second-largest emissions source in the UK in 2012, at 10.1% CO_2-e of total emissions, with direct soils emissions making up over half of this figure. Peat soils under intensive horticulture potentially contribute a substantial proportion of soils emissions, but measured UK estimates are currently lacking. Quantifying emissions is a crucial step in refining national inventory estimates, prioritising mitigation efforts, and facilitating sustainable planning of soil management. This study reports the first budget of greenhouse gas emissions from peat soils under horticultural production in the UK. Fluxes of CO_2, N_2O and CH_4 were measured using a chamber method, on three farms in East Anglia, UK, on three contrasting soil types (c. 20%, c. 35%, and c. 70% soil organic matter content respectively), under a number of commercially important crops in similar rotations. Cumulative annual CO_2 fluxes over the period 10th June 2011 to 9th June 2012 were 13.04 ± 2.39 to 30.85 ± 2.54 t CO_2-e ha^{-1} yr^{-1}, showing a general increase with soil organic matter content, and on cropped compared to bare soils. Annual emissions of N_2O varied from 4.94 ± 0.77 to 13.88 ± 1.91 t CO_2-e ha^{-1} yr^{-1}, and CH_4 from -0.02 ± 0.08 to 0.04 ± 0.02 t CO_2-e ha^{-1} yr^{-1}; neither showed a significant relationship with either soil organic matter content or cropping. Distinct seasonal patterns of CO_2 and N_2O were observed, corresponding to significant correlations between emissions and soil and air temperature, soil moisture content, water table depth, and soil nitrate on some soil types. No discernible seasonal pattern in CH_4 was observed, and very few significant correlations with soil environmental variables were found. Compared to emissions estimates suggested in IPCC inventory guidelines for cultivated peat soils, the observed emissions in this study were relatively high, and net annual fluxes of CO_2 and CH_4 represented an annual soil carbon loss of 0.42 to 0.94 cm soil yr^{-1} (based on a mean peat bulk density of 0.34 to 0.73 g cm^{-3}, and a mean soil C content to 1 m depth of 11.6 to 34.3%). These results indicate that UK agricultural soil emissions and soil loss could be substantially reduced by mitigating horticultural peat soil emissions.

2.1. Introduction

Peat soils (Histosols) represent a major store of carbon (C) within the terrestrial biosphere (Limpens *et al.*, 2008). While most studies of C loss have focused on non-agricultural peat soils, current evidence suggests that intensively cultivated lowland peats may also represent hotspots for greenhouse gas (GHG) emissions and related soil organic C (SOC) losses (Smith *et al.*, 2007a; Dawson *et al.*, 2010; Cannell *et al.*, 1999). Work on these arable peat systems has mainly focussed on quantifying CO_2 emission and changes in SOC storage, largely neglecting emissions of CH_4 and N_2O, whilst complete GHG budgets for organic soils under continuous arable management are entirely lacking (Evans *et al.*, 2011; Worrall *et al.*, 2011). As GHG Emission Factors (EF) for arable mineral soils, or peat under managed grassland, are not likely to reflect emissions from these arable peats (due to differences in management regime and soil properties, for example), there is an urgent need to develop accurate EFs for these agro-ecosystems. This is reinforced by their economic importance in terms of food security in many countries (Parish *et al*., 2008).

Improving emissions estimates allows more accurate inventorying at the national level, a legal requirement for emissions reduction target compliance (UK Parliament, 2008), and an important step in identifying mitigation priorities. Mitigating agricultural emissions could contribute substantially to overall reduction targets: in 2012, agriculture was estimated to be the second-largest sector contributor to emissions in the UK (9.0% CO_2-e of total UK emissions), with direct agricultural soils emissions of N_2O making up almost half of this figure (4.6% CO_2-e; adapted from Webb *et al*., 2014). Presently, agricultural peat emissions are calculated using a default EF averaged over all temperate zones (Webb et al., 2014), with little recognition of differences in UK climate, peat soil characteristics and agricultural management practices when compared to the temperate-zone average. Consequently, soil N_2O EFs in particular have been identified as a priority for refinement, accounting for 52-56% of the total uncertainty in UK GHG estimates since 1990 (Webb *et al*., 2014).

The sustainability of cropping on peat soils is an important consideration for long term food security. Rates of soil loss from UK lowland peats have been reported at 0.27 to 3.09 cm soil yr^{-1} (Richardson and Smith, 1977; Hutchinson, 1980), with a recent estimate from East Anglian arable fens of 1.10 to 1.48 cm soil yr^{-1} between 1982 and 2004 (Dawson *et al*., 2010). Emissions of CO_2 may constitute between 35% and 100% of peat subsidence C losses (Leifeld *et al*., 2011), but the literature remains unclear regarding the proportion of total SOC loss that can be attributed to other routes (principally, wind and water erosion, leaching, and crop adherence). Quantifying GHG emissions from arable peats under different management regimes can therefore contribute to estimates of future soil losses, and enable prioritisation of soil loss mitigation measures via the different routes of loss.

The factors influencing emissions from agricultural soils are numerous and interact in often complex ways; factors include soil (e.g. moisture, temperature, porosity, substrate availability), climate (rainfall, temperature), and vegetation (yield, water uptake), which in turn are driven by human activities such as farm operations (Li, 2007). Often, a change in a single factor may simultaneously increase the emission of one GHG and result in the reduction of another (Smith *et al*., 2008). Whilst individual studies have been conducted and models created that identify the relative importance of these factors in driving agricultural soils emissions (e.g. Giltrap *et al*., 2010), quantification of emissions drivers requires further attention with regard to horticultural peat soils in the UK.

The primary aim of this study was to quantify and compare emissions of CO_2, CH_4 and N_2O from soils of comparatively high organic matter content (c. 70% SOM, c. 35% SOM and c. 20% SOM to 1 m depth respectively), under a number of commercially important horticultural crops. A secondary objective was to determine which soil and crop factors most strongly influence GHG fluxes from these soils.

2.2. Methods and materials

2.2.1. Study sites

The study area was located in East Anglia, UK, and comprised drained lowland fen typified by flat topography (0-1% slope) with a long-term (1980-2013) mean annual rainfall of 621 mm, mean annual temperature of 10.2°C (winter mean 4.4°C, summer mean 16.4°C), and mean annual sunshine of 1280 h (UK MetOffice, 2014). All sites have been under long-term horticultural and arable production in rotation since c. 1940, growing primarily vegetables (e.g. celery, leeks, lettuces, potatoes, red beet) and wheat. Details of site locations and management practices implemented during the monitoring period are provided in Fig. 2.1., Chapter 3 and Appendix A.

Three farms (sites) were identified for monitoring on the basis of their contrasting soil organic matter content to 1 m depth: farm site L20, with c. 20% SOM content, farm site M35, with c. 35% SOM content, and farm site S70, with c. 70% SOM content. Monitoring sites were selected from each farm using farm records to identify fields with typical commercial cropping rotations. Crops selected for study were: celery (*Apium graveolens* L.), red beet (*Beta vulgaris* L., grown in tandem with a cover crop of barley, *Hordeum vulgare* L.), lettuce (*Lactuca sativa* L.), radish (*Raphanus sativus* L.), and potato (*Solanum tuberosum* L.). Six fields (three at farm M35 and three at farm S70) were sampled monthly from April 2011 until June 2012, with a seventh field (at farm L20) added in June 2011.

One sampling block (12 x 30 m) was randomly positioned within each field (Appendix B). Blocks were located at least 10 m from field margins and areas of heavy vehicle trafficking were avoided. Each monitoring block contained five randomised pairs of either cropped (C) or bare (B) plots 6 x 6 m in size with the long axis of the block running parallel to the crop planting line. During the growing season the bare plots were covered with black geo-textile ground cover to suppress weed growth. Each 6 x 6 m plot enclosed a 1 m buffer around its inner boundary to reduce edge effects of adjacent plots and field areas, leaving a 4 x 4 m monitoring area containing one randomly positioned GHG monitoring collar. No observable effect of the ground cover on soil temperature or moisture was noted within the sampling areas.

Figure 2.1. Location of field sites within the UK: site L20 (c. 20% SOM content); site M35 (c. 35% SOM content); and site S70 (c. 70% SOM content). Copyright Google Maps, 2015.

2.2.2. Seasonal greenhouse gas fluxes

Greenhouse gas measurements were undertaken approximately monthly at all sites. Closed, non-vented static chambers were used to monitor soil emissions of N_2O and CH_4. Briefly, cylindrical black polyethylene collars (internal dimensions d = 26.3 cm, h = 19.8 cm; PBSL, Colchester, UK) were inserted 12 cm into the soil and left *in situ* unless removed to allow tillage operations to take place. All vegetation was removed from within and surrounding the collar at installation, and at least 24 h before each sampling event.

The static chambers fitting onto the soil collars consisted of white opaque polypropylene closed cylinders (internal dimensions d_1 = 22.0 cm, d_2 = 25.0 cm, h = 26.3 cm; CJK Packaging, Derbyshire, UK), with a rubber septum sampling port 7 cm from the top of the chamber, and an internal battery powered 25 mm 12 V fan (typical flow rate = 54 L min^{-1}; CPC Ltd, Leeds, UK). Chambers were inserted 4.5 cm into the top lip of the soil collar so that the flexible seal around the chamber rim formed an air-tight seal with the collar, giving a final enclosed headspace volume of 19.8 dm^3 (Appendix B). All chambers were vented for > 5 min prior to collar attachment and GHG sampling. Individual temperature loggers (iButton DS1921G-F5 thermochrons, HomeChip Ltd., Milton Keynes, UK) were placed inside each collar to monitor within-chamber temperatures compared to ambient temperature over the enclosure period (no significant heating effect was found within the chambers relative to that outside; $p > 0.05$).

After placement of the static chamber on the collar, the chamber headspace was sampled four times at approximately 10 min intervals, with the first gas sample taken immediately after chamber enclosure. Glass sample vials (20 mL) fitted with butyl rubber septa (QUMA Electronik & Analytik GmbH, Wuppertal, Germany) were manually evacuated twice prior to sampling. Gas samples were removed from the headspace of the static chambers using a 30 ml syringe and a 21G, 2-inch needle. On insertion of the needle into the septum, the syringe was flushed twice then the sample taken and injected into the vial.

Gas samples were stored at room temperature in the dark until analysis. Sample analysis was within six weeks of collection using a gas chromatograph (Varian 450-GC, Bruker UK Ltd., Coventry, UK), equipped with a flame ionisation detector (FID, operated at 120-125°C) and electron capture detector (ECD, operated at 300°C), and attached to a QUMA QHSS®-40 Headspace Sampler (QUMA Electronik & Analytik GmbH, Wuppertal, Germany), which injected 2 mL of sample into the GC. Gas standards with certified concentrations to within 2-10% of their specification (STG, Ltd., UK) were analysed concurrently with field samples. Sample concentrations of GHGs were calculated using a linear regression of the known standards concentrations against standard chromatograph area outputs from the GC ($mV \times s$) with the origin assumed to be zero (after Levy *et al*., 2011).

Immediately following chamber measurements, soil CO_2 emissions were measured within 10-30 cm of the outside of chamber collars using an EGM-4 portable infra-red gas analyser (IRGA; PP Systems Ltd, Hitchin, UK) equipped with an automatic SRC-1 soil respiration chamber (internal dimensions d = 10.35 cm, h = 6.50 cm; total volume = 1694 mL), as described in Alm (2007). Soil temperature (0-10 cm depth) and air temperature (at c. 1 m above ground) were recorded at the time of sampling using a Checktemp1® temperature probe (accurate to ±0.3°C; Hanna, Bedfordshire, UK).

All GHG samples from a single field were taken within a 3 h period during daylight hours with all six fields sampled over a 96 h period.

2.2.3. Measurement of environmental variables

A range of soil and crop samples were taken within 24 h of chamber and IRGA measurements to support the GHG monitoring work. In each plot, a randomly placed bulk density core (h = 5 cm, V = 100 cm^3) was used to collect soil from 0-5 cm and 0-10 cm depths. Soils were stored at 4°C then homogenised before analysis for a number of physical and chemical properties. Briefly, soil moisture and bulk density were calculated after drying the soil cores at 105°C for > 24 h. A sub-sample of field-moist soil (0-5 cm depth) was used to extract plant available NH_4^+ and NO_3^- with 0.5 M K_2SO_4 or 1 M KCl for 1 h (1:5 w/v), then concentrations determined with a Powerwave XS Microplate Spectrophotometer (BioTek UK, Bedfordshire, UK) using the colorimetric methods outlined in Mulvaney (1996) and Miranda *et al*. (2001) respectively. Soil pH was measured on field-moist soil (0-10 cm depth) in 0.01 M $CaCl_2$ (1:1 w/v) following the method of Doran and Jones (1996).

To estimate above-ground crop biomass during the growing season, five randomly selected plants were taken from near the sampling block in each field, their roots removed, and shoot fresh and dry (80°C, > 48 h) weight determined. At harvest, shoots were additionally separated into ‘harvested’ and ‘residue’ portions, which were weighed and dried separately.

Water table depth (to within 10 cm) was measured at the field margin ditches perpendicular to the sampling block, and in the centre of the monitoring block using a dip well.

2.2.4. Data cleaning and statistical analysis

2.2.4.1 Data cleaning procedure: nitrous oxide and methane

Sample concentrations (C_v, in ppm) were first converted to mass unit equivalents, using equation 1:

$$C_m = \frac{C_v \cdot M \cdot P}{R \cdot T} \tag{1}$$

where C_m is the mass per volume of the GHG (in µg N_2O-N L^{-1} or µg CH_4-C L^{-1}), M is the total molecular weight of the carbon or nitrogen within the trace gas species (in g), P is the barometric pressure (in hPa), R is the gas constant (= 8.314 J K^{-1}), and T is the air temperature (in K).

Chamber size testing revealed the majority of fluxes to be approximately linear over time. Therefore we calculated fluxes (F, in µg N_2O-N m^{-2} h^{-1} or µg CH_4-C m^{-2} h^{-1}) over the chamber enclosure period from the C_m values using a linear regression of concentrations corrected for chamber volume and area, using equation 2:

$$F = V \cdot \frac{C_{rate}}{A} \tag{2}$$

where V is the chamber volume (in m^3), C_{rate} is the linear rate of change of GHG concentration over time (in µg N_2O-N h^{-1} or µg CH_4-C h^{-1}), and A is the area of soil enclosed by the chamber (in m^2).

Fluxes were visually examined and accepted for further analysis if the adjusted R^2 (R^2_{adj}, a more suitable measure of linearity that R^2 where n = small, Stevens, 2002) was ≥ 0.70 (Ford *et al*., 2012). Where R^2_{adj} of the raw flux was < 0.70, a single data point was removed to increase the R^2_{adj} past the threshold. This type of data cleaning is considered acceptable where there is a high concentration data point at t_0 (commonly caused by perturbation of headspace air on chamber insertion), a low concentration data point at t_3 (often the result of headspace saturation and subsequent tailing off of the flux curve), or a clear outlier at t_1 or t_2. Additionally, since low fluxes (defined here as fluxes ≤ 0.01 mg GHG-x m^{-2} h^{-1}, or where the mean concentration of the data points was less than ambient concentration) often display a low R^2_{adj}, they were also included in the analysis even if they had an R^2_{adj} value of < 0.70; this avoided a high flux bias in the data (Alm *et al*., 2007; Ford *et al*., 2012). Data cleaning resulted in 94% of N_2O and 92% of CH_4 individual flux curves being accepted for analysis.

2.2.4.2. Data cleaning procedure: soil respiration carbon dioxide

Soil respiration values were corrected for diurnal temperature variation after Koerber *et al*. (2009), using the procedure described in Parkin & Kaspar (2003). We corrected fluxes using Met Office MIDAS air temperature averaged for available stations in the vicinity of each field site, and a Q_{10} of 2.2 for peat soils (York, 2012). No information could be found for appropriate correction for diurnal flux variation for nitrous oxide or methane, so the raw data values were used.

2.2.4.3. *Statistical analysis*

For each sampling date, the flux mean and standard error were calculated separately for each SOM% and cropping combination, to give values in mg CO_2-C m^{-2} h^{-1}, µg N_2O-N m^{-2} h^{-1} and µg CH_4-C m^{-2} h^{-1} respectively.

Mean cumulative fluxes over the whole measurement period were calculated separately for each SOM and cropping combination by multiplying hourly values by 24 to give a daily flux, followed by stepwise interpolation of the flux values on known dates, then summing the resulting values over the required period. Cumulative standard errors for each treatment were taken as the standard error of the cumulative means of individual chambers within that SOM-cropping type. Cumulative fluxes were calculated for the period 10th June 2011 to 9th June 2012 to allow inclusion of all sites for comparison. All cumulative flux estimates were converted to 100-year global warming potential (GWP_{100}) CO_2 equivalents (CO_2-e) according to the methodology outlined in IPCC (2000), using a GWP_{100} of 21 for CH_4, and 310 for N_2O. This allowed comparison between SOM-cropping types for total GWP and individual GHG GWPs. It also enabled direct comparison with emissions estimates for UK cultivated peats in the most recent UK National GHG Inventory Report (Webb *et al.*, 2014), which utilises IPCC (2000) methodology EFs.

All figures in this chapter were produced using Sigmaplot v. 12.3 (Systat Software, Inc.). Statistical analyses were performed using SPSS v. 20 (IBM, Inc.), with significance being accepted at $p \leq 0.05$ except where stated otherwise. All data values are quoted as mean ± standard error unless specified otherwise.

Statistical analyses were performed separately on each SOM-cropping combination. Normality was tested using the Shapiro-Wilk test (Field, 2005), and non-normal data were log-transformed or square-root transformed; where transformation was ineffective, or where heterogeneity of variances was observed (Levene’s or Welch’s test statistic), appropriate non-parametric tests were used to compare medians of those data groups.

Cumulative fluxes of CO_2, N_2O, and total GWP_{100} were compared using 2-WAY ANOVA and Bonferroni Post-Hoc tests, using SOM% and cropping as fixed factors. Cumulative CH_4 fluxes were compared using the Kruskall-Wallis test.

Relationships between individual GHGs and environmental variables (soil temperature, MIDAS mean air temperature, measured air temperature, daily and 5-day cumulative rainfall, soil N concentrations, soil bulk density, soil pH, and crop aerial biomass) were explored using Kendall's tau statistic (τ), a powerful correlation statistic for the non-normal distribution of these variables.

2.3. Results

2.3.1. Weather conditions

The mean air temperature for the period 14th April 2011 to 31st Dec 2011 was similar for all sites, at 13.4°C, 13.2°C and 13.1°C for L20, M35 and S70 sites respectively (Fig. 2.2.a.), with the highest mean daily temperatures (23.4 to 24.7°C) observed at all sites during June, and the lowest (-0.4 to 0.5°C) during December. Farm S70 was the wettest site during 2011, with total cumulative rainfall for this period of 283 mm, 286 mm, and 311 mm for the three sites respectively; the highest monthly rainfall (58.4 to 71.4 mm) was recorded during June at all sites. Mean soil temperatures during 2011 varied from 3.9 to 18.2°C at the L20 site, 3.5 to 20.0°C at the M35 site, and 4.3 to 16.7°C at the S70 site.

From 1st January 2012 to 21st June 2012, mean air temperature was lower than during the previous year at all sites (8.3°C, 8.1°C and 8.0°C at L20, M35 and S70 sites respectively, Fig. 2.2.a.). The highest mean daily temperatures were recorded in May (18.6 to 19.8°C), and the lowest in February (-7.2 to -6.8°C). In 2012, total cumulative rainfall was very similar for the three sites (306 mm, 301 mm and 300 mm respectively), and the wettest month was April (103 to 104 mm total rainfall). Mean soil temperatures were lower than during 2011 at the L20 and M35 sites (3.9 to 13.4°C and 3.9 to 17.5°C respectively), and higher at the S70 site (4.6 to 17.1°C).

Over the whole monitoring period, S70 was the coolest and wettest site; while M35 and L20 had similar higher daily average temperature and lower cumulative rainfall. Peak rainfall events over the measurement period were moderate and similar across sites, with peak daily rainfall events of 19 to 23 mm observed.

2.3.2. Patterns of GHG emissions and related environmental variables over time

2.3.2.1. Seasonal fluxes of carbon dioxide

Mean carbon dioxide fluxes varied from -2.24 ± 11.45 to 201.45 ± 26.81 mg CO_2-C m^{-2} h^{-1} across the whole monitoring period (14th April 2011 to 21st June 2012; Fig. 2.2.b.), representing the largest GHG flux from all sites on most sampling occasions.

A pronounced seasonal pattern of soil respiration was observed at all sites, with the highest fluxes (157.08 ± 29.45 to 201.45 ± 26.81 mg CO_2-C m^{-2} h^{-1} on cropped soils and 127.46 ±

49.91 to 201.07 ± 20.66 mg CO_2-C m^{-2} h^{-1} on bare soils) recorded between June and September 2011, declining relatively steadily to the lowest fluxes (6.28 ± 15.27 to 21.70 ± 5.71 mg CO_2-C m^{-2} h^{-1} on cropped soils and -2.24 ± 11.45 to 20.01 ± 2.66 mg CO_2-C m^{-2} h^{-1} on bare soils) which were measured between October 2011 and February 2012. Net carbon uptake was only observed briefly, at the L20 site during October 2011.

The overall mean CO_2 emission was greater from cropped soils than bare soils at all sites, varying from 65.16 ± 9.64 to 102.60 ± 6.27 mg CO_2-C m^{-2} h^{-1} on cropped soils and from 46.46 ± 8.79 to 80.56 ± 5.28 mg CO_2-C m^{-2} h^{-1} on soils without crops. However, marked differences were not observed on all sampling dates between cropped and bare soil plots (Fig. 2.2.b.). Mean CO_2 flux correlated clearly with the amount of SOM on bare soils only, increasing from 46.46 ± 8.79 mg CO_2-C m^{-2} h^{-1} at the L20 site to 72.87 ± 4.16 mg CO_2-C m^{-2} h^{-1} at the M35 site and 80.56 ± 5.28 mg CO_2-C m^{-2} h^{-1} at the S70 site. On cropped soils, an increase in mean CO_2 flux was observed between the L20 and L35 sites (from 65.16 ± 9.64 to 102.60 ± 6.27 mg CO_2-C m^{-2} h^{-1}), but mean emission declined to 89.59 ± 5.46 mg CO_2-C m^{-2} h^{-1} at the S70 site.

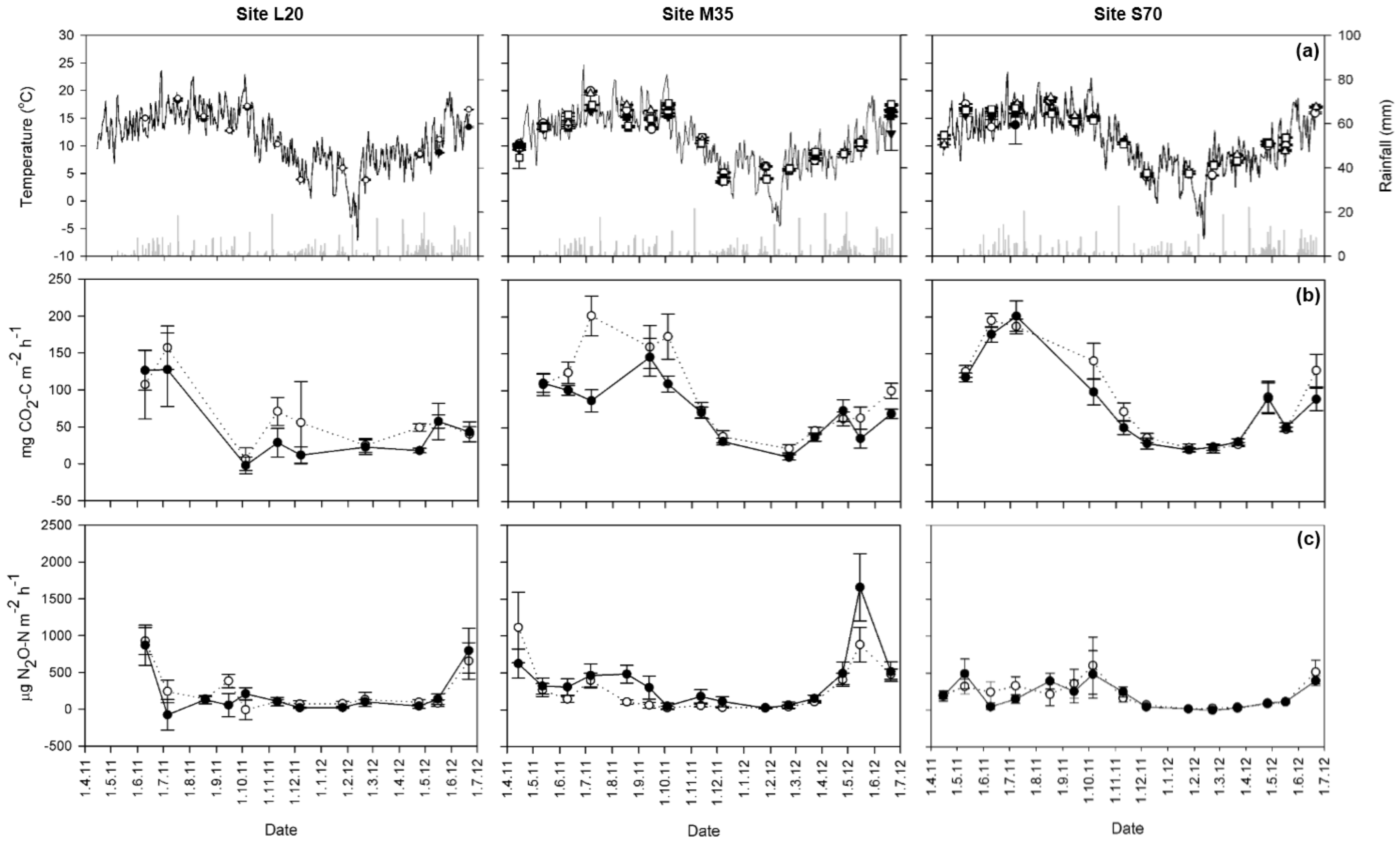

Figure 2.2. Daily rainfall, air temperature and soil temperature (a); and fluxes of CO_2 (b), and N_2O (c); 14th April 2011 to 21st June 2012: cropped and bare soils at site L20 (c. 20% SOM content), site M35 (c. 35% SOM content), and site S70 (c. 70% SOM content). Error bars represent ± 1 S. E. of the mean value.

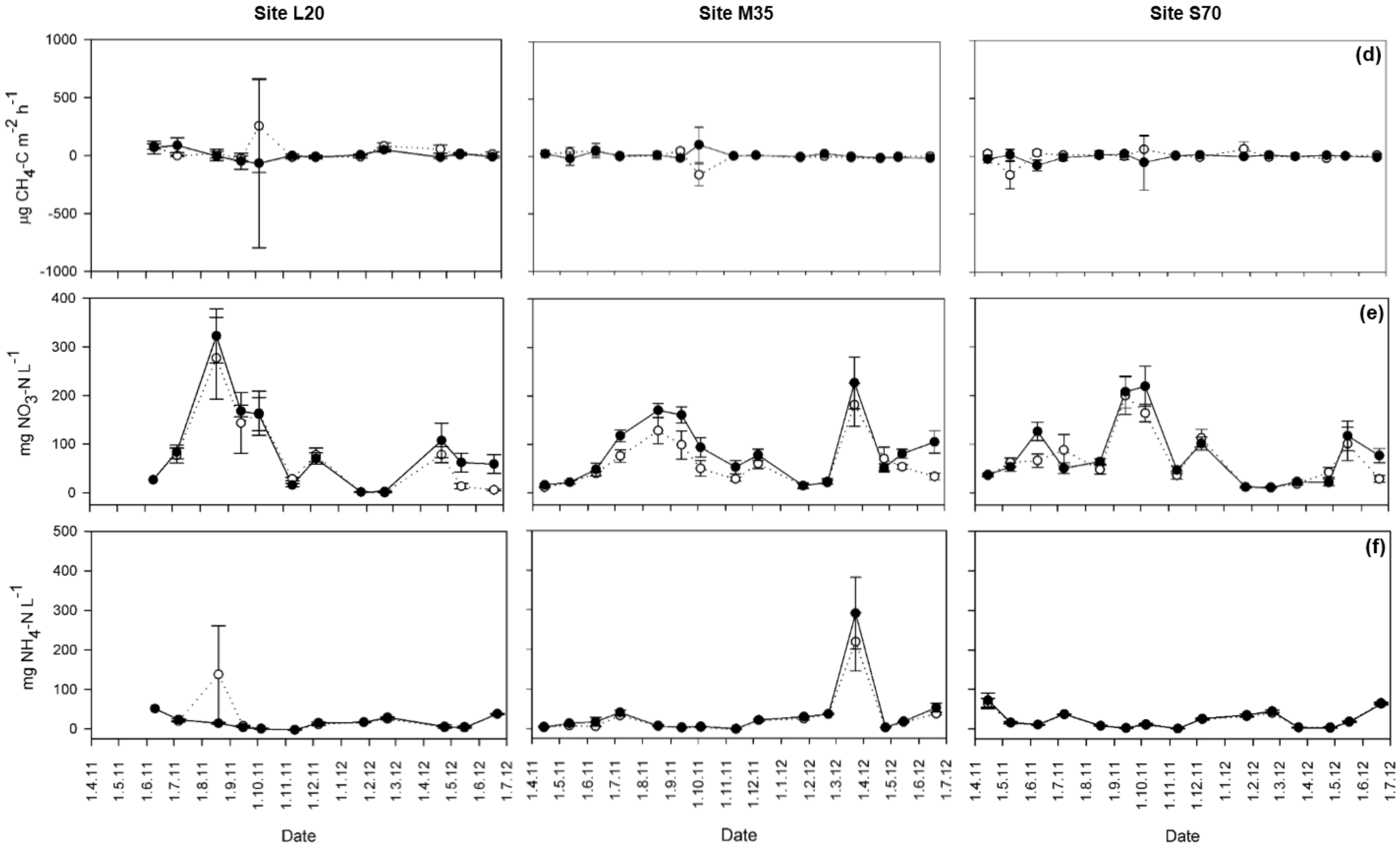

Figure 2.2. Fluxes of CH_4 (d); and soil NO_3^- (e) and NH_4^+ (f); 14th April 2011 to 21st June 2012: cropped and bare soils at site L20 (c. 20% SOM content), site M35 (c. 35% SOM content), and site S70 (c. 70% SOM content). Error bars represent ± 1 S. E. of the mean value.

—— MIDAS mean air temperature (°C); ▨ Rainfall (mm); Mean soil temperature (°C): ● Cropped soil, field L1, M1, S1, ○ Bare soil, field L1, M1, S1, ▼ Cropped soil, field L2, M2, S2, △ Bare soil, field L2, M2, S2, ■ Cropped soil, field M3, S3, □ Bare soil, field M3, S3; CO_2, N_2O, CH_4 emission, and NO_3-N and NH_4-N concentration: ● Cropped soil, ○ Bare soil.

2.3.2.2. *Seasonal fluxes of nitrous oxide*

During most months, mean N_2O emissions were considerably smaller in magnitude than CO_2 emissions, even when corrected for GWP, but showed greater variability. Nitrous oxide fluxes fell within the range -75.24 ± 206.82 to 1660.82 ± 455.31 µg N_2O-N m^{-2} h^{-1} over the monitoring period (Fig. 2.2.c.).

Similarly to soil respiration, N_2O emission tended to be lower in the winter and higher during the summer. Peak mean fluxes (600.89 ± 387.05 to 1115.69 ± 478.43 µg N_2O-N m^{-2} h^{-1} on cropped soils and 489.41 ± 201.40 to 1660.82 ± 455.31 µg N_2O-N m^{-2} h^{-1} on bare soils) were observed during the months of April to June, with the exception of peak N_2O from cropped S70 soils in October 2011. Minimum mean fluxes (-6.58 ± 135.11 to 27.57 ± 22.91 µg N_2O-N m^{-2} h^{-1} on cropped soils and -75.24 ± 206.82 to 23.95 ± 11.26 µg N_2O-N m^{-2} h^{-1} on bare soils) were recorded between October 2011 and February 2012, except for on bare L20 soils, where minimum fluxes were observed in July 2011. Mean negative fluxes were recorded on only a few occasions, at the L20 site (July and October 2011) and S70 site (February 2012).

The differences in mean N_2O emission over the whole monitoring period between cropped and bare soil plots were less consistent than for soil respiration. Mean fluxes were higher from the cropped than bare plots at the L20 and S70 sites (224.75 ± 46.17 compared to 206.06 ± 56.00 µg N_2O-N m^{-2} h^{-1}, and 219.00 ± 36.47 compared to 198.65 ± 31.85 µg N_2O-N m^{-2} h^{-1} respectively), but not at the M35 site (281.90 ± 42.71 compared to 391.40 ± 50.09 µg N_2O-N m^{-2} h^{-1}). Mean fluxes on individual sampling occasions were for the most part very similar, with only a few showing marked differences between cropped and bare soil plots (Fig. 2.2.c.). In relation to SOM content, mean nitrous oxide emission followed the same pattern on both cropped and bare soils: highest mean fluxes from the M35 soils, followed by the L20 soils, with the lowest mean fluxes from the S70 soils.

2.3.2.3. *Seasonal fluxes of methane*

Mean CH_4 fluxes were very small when compared to the other GHGs, both in absolute terms and when corrected for GWP_{100}. Mean methane fluxes varied from -161.91 ± 118.53 to 255.30 ± 398.95 µg CH_4-C m^{-2} h^{-1} over the period 14th April 2011 to 21st June 2012 (Fig. 2.2.d.).

Methane fluxes oscillated around zero for the whole of the measurement period, with no apparent seasonal trend in emissions. Maximum emissions (53.59 ± 62.12 to 255.30 ± 398.95

µg CH_4-C m^{-2} h^{-1} on cropped soils and 21.89 ± 18.07 to 103.10 ± 152.89 µg CH_4-C m^{-2} h^{-1} on bare soils) were observed during September-October 2011 at most sites, except for on cropped M35 soils (June 2011) and on L20 soils without crops (July 2011). Minimum fluxes of -161.91 ± 118.53 to -23.82 ± 19.69 µg CH_4-C m^{-2} h^{-1} on cropped soils and -78.55 ± 46.52 to -17.76 ± 59.16 µg CH_4-C m^{-2} h^{-1} on bare soils were measured at similar times, during September-October 2011 on cropped M35 and on L20 plots, and during May-June 2011 on cropped S70 and bare M35 plots. Net methane uptake was observed on 39% of sampling occasions.

In terms of mean methane emission over the whole measurement period, neither SOM content nor cropping appeared to strongly influence fluxes. Mean CH_4 fluxes more than doubled between the L20 and M35 bare soil sites (5.00 ± 48.38 compared to 11.37 ± 11.46 µg CH_4-C m^{-2} h^{-1}), then fell again to a methane uptake of -5.60 ± 14.57 µg CH_4-C m^{-2} h^{-1} from the S70 site. On cropped soils, the opposite pattern was observed, with mean fluxes of 37.04 ± 36.06, -0.34 ± 9.72, and 2.35 ± 11.61 µg CH_4-C m^{-2} h^{-1} from L20, M35 and S70 soils respectively.

2.3.2.4. Seasonal patterns of soil N availability

Mean available NO_3-N content (< 0.5 to 322.34 ± 55.82 mg NO_3-N kg^{-1} dry soil) was higher on most dates throughout the monitoring period than mean available NH_4-N content (< 0.5 to 291.28 ± 90.53 mg NH_4-N kg^{-1} dry soil, Fig. 2.2.e., f.).

The pattern of soil nitrate concentration over time differed between sites, with peak NO_3-N levels observed in August 2011 at the L20 site (322.34 ± 55.82 mg NO_3-N kg^{-1} dry soil), March 2012 at the M35 site (227.19 ± 53.49 mg NO_3-N kg^{-1} dry soil), and October 2011 at the S70 site (219.11 ± 41.38 mg NO_3-N kg^{-1} dry soil). A peak in available nitrate was recorded at all sites from March to May 2012 at all sites (107.45 ± 35.36, 227.19 ± 53.49 and 117.27 ± 30.40 mg NO_3-N kg^{-1} dry soil respectively). Differences in soil NO_3-N between cropped and bare soil plots were observed on very few sampling dates and tended to be greater during the summer, although mean cropped NO_3-N was lower overall than mean bare soil NO_3-N (66.05 ± 3.92 and 82.93 ± 4.22 mg NO_3-N kg^{-1} dry soil respectively). Mean soil nitrate over the whole observation period was highest at the L20 site (86.57 ± 10.08 mg NO_3-N kg^{-1} dry soil), followed by the S70 site (73.89 ± 4.34 mg NO_3-N kg^{-1} dry soil), and lowest at the M35 site (72.01 ± 4.13 mg NO_3-N kg^{-1} dry soil).

Soil available NH_4-N remained at a relatively low level throughout the observation period, only rising above 100 mg NH_4-N kg^{-1} dry soil twice, on L20 cropped soils in August 2011 (peaking at 138.33 ± 122.89 mg NH_4-N kg^{-1} dry soil, Fig. 2.2.f.), and on M35 cropped and bare soils in March 2012 (peaking at 219.19 ± 73.17 and 291.28 ± 90.53 mg NH_4-N kg^{-1} dry soil respectively). During periods of baseline ammonium levels, the pattern of available NH_4-N concentration was very similar at all sites, with no discernible differences between cropped and bare soils, apart from at the M35 site where cropped NH_4-N was slightly lower during June in both years. Overall mean soil ammonium concentrations were almost identical at the L20 and S70 sites (19.41 ± 5.73 and 22.93 ± 1.32 mg NH_4-N kg^{-1} dry soil respectively), and higher at the M35 site (32.26 ± 4.80 mg NH_4-N kg^{-1} dry soil).

2.3.2.5. Effect of environmental variables on GHG emissions

Soil temperature, mean MIDAS air temperature, and measured air temperature showed a highly significantly positive correlation with soil respiration at both the M35 and S70 sites, but did not significantly correlate with soil respiration at the L20 site (Table 2.1.). In terms of the amount of variability explained by temperature, soil temperature was the best predictor of CO_2 emission (15-27% of variability), followed by mean air temperature (15-22% of variability), with measured air temperature being the weakest predictor (13-19% of variability). This relationship was consistent at both M35 and S70 sites on both cropped and bare soils, and for the most part stronger on cropped than bare soils.

Indicators of soil moisture were generally negatively associated with CO_2 emission, with water table depth explaining the greatest proportion of variability in fluxes, but only at the cropped M35 and the S70 sites (8% and 10-11% of variability respectively). Soil moisture content (H_2O_{DW}) accounted for a further 3-5% of variability in soil respiration on cropped M35 and S70 soils and 5-8% of variability on M35 and S70 soils without crops. Daily rainfall only significantly predicted CO_2 emission at the S70 site, explaining 4% of variability in emission, while 5-day cumulative rainfall explained 1-5% of variation in soil respiration on the S70 and M35 cropped soils.

Soil nitrate, ammonium and total N (nitrate + ammonium) were less consistent predictors of soil respiration, with the relationship varying between sites. At the M35 site, soil N variables were negatively associated with soil respiration, with soil NO_3-N and total N explaining 2% and 3% of variability in cropped soil CO_2 emission, and soil NH_4-N explaining 2% of bare soil CO_2 emission. Soil N variables were positively associated with CO_2 emission at the L20 and S70 sites. A significant correlation on the L20 site was only found between NH_4-N and bare soil emission, but explained the highest proportion of variability (7%). At the S70 site, CO_2 emission was positively associated with NO_3-N and soil N, explaining 3% and 2% of emission variability respectively.

Soil bulk density was a poor predictor of CO_2 emission, only demonstrating a significant negative association with emission on M35 cropped soils, and rather weakly (2% of variability). Soil pH was positively correlated with M35 cropped CO_2 emission (3% of variability), and negatively correlated with S70 emission (5-6% of variability). Crop aerial biomass was only significantly correlated with soil respiration at the S70 site, but accounted for a large proportion (30%) of variability in emissions.

Temperature variables were significantly positively correlated with N_2O emission in many of the SOM%-cropping categories, although more weakly than for soil respiration. Soil temperature significantly predicted nitrous oxide emission at all sites, explaining 4%, 2-3%, and 5-9% of emissions from L20, M35 and S70 sites respectively. Mean MIDAS air temperature only significantly correlated with N_2O emission at the S70 site, accounting for 3-5% of variability in fluxes. Measured air temperature was a significant predictor on all but the L20 cropped soils, but only explained 1-2% of variability at the M35 and S70 sites and 4% of variability on L20 bare soils.

Soil moisture variables were negatively associated with N_2O emission, but only in a few categories, and only weakly compared to CO_2 emission predictors. Water table depth accounted for 4-5% of variability on S70 soils, while soil moisture content explained 1-2% of variability in fluxes on the M35 and S70 bare soils, and daily rainfall explained 1% of variability on fluxes on cropped S70 soils. Soil N variables were also weaker predictors of N_2O than CO_2 emission, explaining 1-2% of variability in fluxes. Soil NO_3-N concentration was positively associated with N_2O emission at the S70 site and M35 bare soil site (2% of variability), while total soil N was positively associated with N_2O emission at the S70 cropped site. Soil bulk density was a positive predictor of N_2O emission on S70 bare soils (2% of variability). Soil pH was significantly negatively correlated with N_2O flux on S70 soils. Crop aerial biomass was only significantly associated with fluxes at the L20 site, but accounted for 44% of variability in emissions.

Table 2.1. Significant linear correlations between measured environmental variables and emissions of CO_2, N_2O and CH_4.

	Crop	SOM	Soil t (°C)	Mean air t (°C)	Air t (°C)	D rain (mm)	5d rain (mm)	H_2O_{DW} (%)	WT (m)	NO_3-N (mg L^{-1})	NH_4-N (mg L^{-1})	N (mg L^{-1})	Soil BD (g cm^{-3})	Soil pH	$Crop_{AB}$ (g m^{-2})
CO_2	**Crop**	**20%**													
		35%	0.460***	0.472***	0.427***			-0.229***	-0.289***	-0.153**		-0.180***	-0.144**	0.167**	
		70%	0.517***	0.449***	0.385***	-0.189**	0.111*	-0.274***	-0.324***	0.176**		0.125*		-0.214***	0.547***
	Bare	**20%**					0.223*				0.258*				
		35%	0.390***	0.387***	0.365***			-0.215***			-0.144**				
		70%	0.517***	0.426***	0.434***	-0.212***	0.139*	-0.187**	-0.334***	0.179**		0.125*		-0.244***	
N_2O	**Crop**	**20%**	0.197*												-0.661*
		35%	0.129**		0.113*										
		70%	0.216***	0.184***	0.139**	-0.105*			-0.193**	0.137**		0.134**	0.147**	-0.272***	
	Bare	**20%**	0.191*		0.192*										
		35%	0.180***		0.107*			-0.116*		0.094*					
		70%	0.292***	0.232***	0.126**			-0.134**	-0.219***	0.135**				-0.240***	
CH_4	**Crop**	**20%**													
		35%													
		70%													
	Bare	**20%**								0.220*					
		35%			-0.093*					-0.096*		-0.092*			
		70%													

*Values are presented as Kendall's tau statistic (τ), with significance levels presented as * ($p < 0.05$), ** ($p < 0.01$), or *** ($p < 0.001$). Environmental variables presented are: Soil t (soil temperature); Mean air t (mean MIDAS daily air temperature); Air t (measured air temperature); D rain (cumulative MIDAS daily rainfall); 5d rain (cumulative MIDAS 5-day rainfall); H_2O_{DW} (gravimetric soil water content, % of dry weight); Water table (water table depth); NO_3-N (soil extractable nitrate); NH_4-N (soil extractable ammonium); N (soil extractable nitrate + ammonium); Soil BD (soil bulk density); Soil pH (soil pH); and $Crop_{AB}$ (crop aerial biomass).*

Only a small number of environmental variables measured here were significantly associated with methane flux, and only weakly so. Soil nitrate was positively correlated with CH_4 efflux on bare L20 soils, explaining 5% of flux variability. Soil NO_3-N and total N, and measured air temperature, were all negatively correlated with CH_4 emission on bare M35 soils, each accounting for 1% of variability.

2.3.3. Cumulative GHG emissions

Table 2.2. and Fig. 2.3. show annual cumulative emissions of CO_2, N_2O, CH_4 and overall GWP_{100} over the period 10th June 2011 to 9th June 2012.

Similarly to seasonal emissions, CO_2 represented the largest annual flux from all soil types on both cropped and bare soil plots, ranging from 73.7 to 79.5% of total GWP on cropped soils to 60.8 to 79.5% of total GWP on soils without crops (Table 2.2.). N_2O represented a smaller but still substantial annual emission, of between 20.4 and 26.4% of GWP on cropped soils and 20.3 and 39.2% of emissions on bare soils. On all sites, cumulative methane emission was negligible, comprising < 0.5% of annual emissions.

Cumulative annual CO_2 emission varied from 13.04 ± 2.39 to 30.85 ± 2.54 t CO_2-e ha^{-1} yr^{-1}. Annual emission significantly increased as SOM content increased from 20% to 35%, from 19.29 ± 2.7 to 30.85 ± 2.54 t CO_2-e ha^{-1} yr^{-1} on cropped soils and from 13.04 ± 2.39 to 21.52 ± 1.13 t CO_2-e ha^{-1} yr^{-1} on bare soils ($p < 0.01$), but not between L35 and S70 sites (Table 2.2.). Cropping also had a significant main effect on cumulative CO_2 emission ($p < 0.01$), with higher mean fluxes from cropped plots at all three sites. No significant interaction effect on cumulative CO_2 emission was found between cropping and SOM content, but the response of cumulative emission to higher SOM content differed according to cropping: on bare soils, total CO_2 emission was higher at the S70 than the M35 site, but on cropped soils, emission was slightly lower at the S70 site.

Annual nitrous oxide emission fell within the range 4.94 ± 0.77 to 13.88 ± 1.91 t CO_2-e ha^{-1} yr^{-1} (Table 2.2.). Soil organic matter content significantly influenced cumulative N_2O emission ($F = 5.601$, $p < 0.01$), but only between the 35% and 70% SOM sites (significantly decreasing, $p < 0.05$), although the increase in N_2O emission observed between 20% SOM and 35% SOM sites was almost significant ($p = 0.06$). Cropping had no main effect on N_2O emission, while the interaction effect between SOM content and cropping was almost significant ($p = 0.09$).

Table 2.2. Cumulative annual fluxes of CO_2, N_2O and CH_4, and total cumulative GHG emissions (GWP_{100}) in t CO_2-e ha^{-1} yr^{-1} (± S.E.), for cropped and bare soils at L20 site (c. 20% SOM content), M35 (c. 35% SOM content), and S70 site (c. 70% SOM content). Totals are reported for the period 10th June 2011 to 9th June 2012.

	Cropped soil, t CO_2-e ha^{-1} yr^{-1}				Bare soil, t CO_2-e ha^{-1} yr^{-1}				% heterotrophic respiration
	CO_2	**N_2O**	**CH_4**	**GWP_{100}**	**CO_2**	**N_2O**	**CH_4**	**GWP_{100}**	
L20	19.29^{ac}	6.9	-0.01	26.19^{h}	13.04^{bc}	4.94	-0.02	17.96^{i}	67.6
	± 2.76	± 2.07	± 0.09	± 3.35	± 2.39	± 0.77	± 0.08	± 3.10	
M35	30.85^{ad}	7.93^{f}	0.04	38.82^{h}	21.52^{bd}	13.88^{g}	-0.004	35.39^{i}	70.0
	± 2.54	± 0.78	± 0.02	± 2.38	± 1.13	± 1.91	± 0.01	± 1.96	
S70	28.26^{e}	7.9^{f}	0.01	36.17	26.03^{e}	6.66^{g}	0.04	32.73	92.1
	± 2.25	± 2.23	± 0.04	± 2.14	± 2.50	± 1.49	± 0.05	± 2.30	

Statistically significant results between groups (2-way ANOVA) are signified by the superfixes a-i.

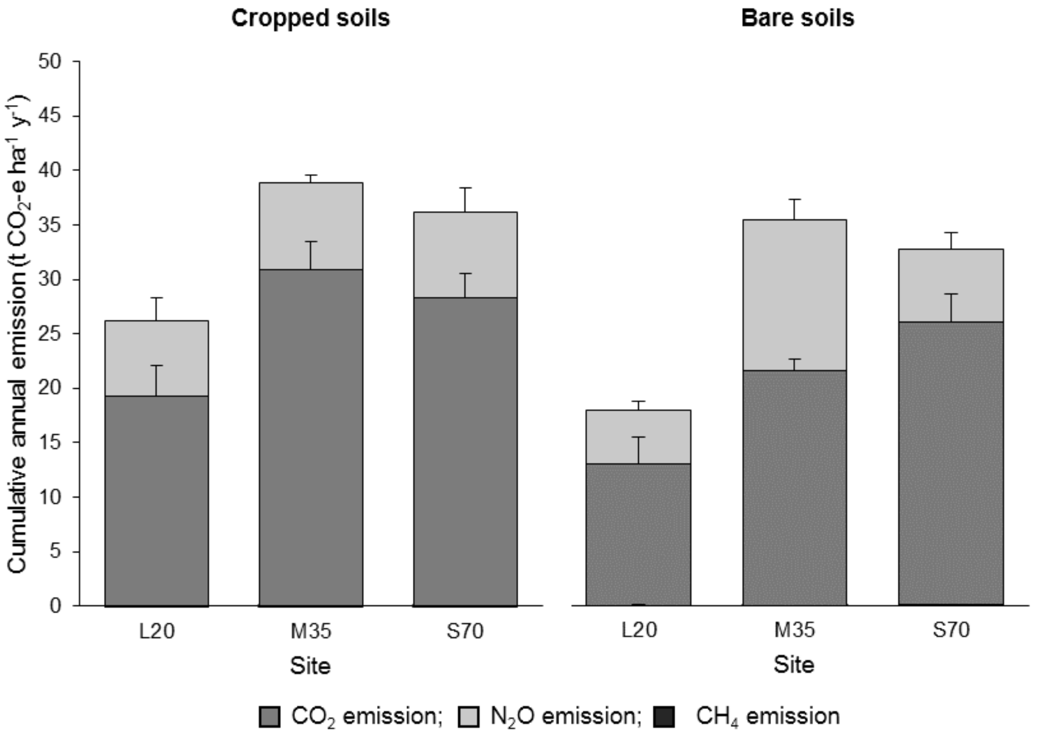

Figure 2.3. Cumulative annual emission, cropped and bare soils: site L20 (c. 20% SOM content); site M35 (c. 35% SOM content); and site S70 (c. 70% SOM content). Error bars represent ± 1 S. E. of the mean value.

Cumulative methane emission (-0.02 ± 0.08 to 0.04 ± 0.02 t CO_2-e ha^{-1} yr^{-1}, Table 2.2.) was not significantly affected by either SOM% or cropping. A non-significant trend of increasing emission was seen on the bare soil plots as SOM content increased, but was only observed on cropped plots between the L20 and L35 soils (Table 2.2.). Annual CH_4 emission was lower on bare soil than cropped plots except for at the 70% SOM site.

On both cropped and bare soils, cumulative GWP_{100} (17.96 ± 3.10 to 38.82 ± 2.38 t CO_2-e ha^{-1} yr^{-1}) was lowest at the L20 site, highest at the M35 site, then slightly lower at the S70 site (Table 2.2.). On all sites, total cropped emission was greater than bare soil emission. In agreement with these observations, two-way ANOVA showed a significant main effect of both SOM content ($F = 12.254$, $p < 0.001$) and cropping ($F = 4.882$, $p < 0.05$); the increased overall emission between the L20 and M35 sites was highly significant ($p < 0.001$), but the decline in GWP_{100} between M35 and S70 sites was not significant. No significant interaction of SOM content and cropping was found.

2.4. Discussion

2.4.1. Seasonal fluxes of carbon dioxide and environmental predictors

To the author's knowledge, this is the first study conducted in the UK which estimates a full annual cycle of CO_2, N_2O and CH_4 from a horticultural peat soil. The range of CO_2 emissions recorded in this study (-2.24 ± 11.45 to 201.45 ± 26.81 mg CO_2-C m^{-2} h^{-1}; section 2.3.2.1., Fig. 2.2.b.) is similar to the magnitude of mean fluxes observed in other studies of temperate and boreal cropped Histosols (2.91 to 356.67 mg CO_2-C m^{-2} h^{-1}, e.g. Elder and Lal, 2008; Kechavarzi *et al.*, 2007; Maljanen *et al.*, 2002; Morrison *et al.*, 2013). Additionally, the pronounced seasonal pattern in emissions observed here (highest in summer and lowest in winter) is typical of that found from other agricultural peat soils (e.g. Kechavarzi *et al.*, 2007; Lloyd, 2006; Maljanen *et al.*, 2004). Observed minimum winter fluxes (6.28 ± 15.27 to 21.70 ± 5.71 mg CO_2-C m^{-2} h^{-1} on cropped soils and -2.24 ± 11.45 to 20.01 ± 2.66 mg CO_2-C m^{-2} h^{-1} on bare soils) are comparable to those found by Kechavarzi *et al.* (2007) on bare soils in East Anglia. While peak summer fluxes (157.08 ± 29.45 to 201.45 ± 26.81 mg CO_2-C m^{-2} h^{-1} on cropped soils and 127.46 ± 49.91 to 201.07 ± 20.66 mg CO_2-C m^{-2} h^{-1} on bare soils) are also similar to those previously recorded in East Anglia (Kechavarzi *et al.*, 2007; Morrison *et al.*, 2013), they are considerably smaller than those recorded under cereals by Elder and Lal (2008) (1747.50 mg CO_2-C m^{-2} h^{-1}) and Maljanen *et al.* (2002) (583.33 mg CO_2-C m^{-2} h^{-1}). This may be attributable to the higher mean annual soil temperatures (5 cm depth) of 15.9 to 16.9°C recorded by Elder and Lal (2008) compared to this study's mean of 11.0 to 12.2°C, and similarly, the maximum air temperature of 33°C recorded by Maljanen *et al.* (2002), compared to a recorded summer maximum of 23.9°C here.

In terms of predictors of CO_2 flux rates, this study found a significant positive association between CO_2 emission and temperature variables (13-27% of variability in emissions) at the M35 and S70 sites, and a significant negative association with water table depth (8-11% of variability) at the cropped M35 and the S70 sites. The relationship between temperature and SOM oxidation is well documented (Dawson and Smith, 2007), and many other authors have found soil and air temperature to be amongst the strongest predictors of soil respiration rate on cropped peat soils (e.g. Maljanen *et al.*, 2001; Morrison *et al.*, 2013). Likewise, increasing water table depth is commonly associated with higher CO_2 flux from cultivated peat soils (e.g. Kechavarzi *et al.*, 2007; Maljanen *et al.*, 2001). Other authors have also noted a negative association between CO_2 emission and soil moisture content (e.g. Maljanen *et al.*, 2001;

Morrison *et al*., 2013), and a positive association with crop residue inputs after harvesting (e.g. Elder and Lal, 2008), while soil bulk density and pH are also known to influence emissions from both organic and mineral soils (e.g. Dawson and Smith, 2007; Ĉuhell *et al*., 2010). The strongest predictor of CO_2 emission found in this study was crop aerial biomass (accounting for 30% of variability), but only on S70 soils. This is consistent with the results of Morrison *et al*. (2013) on a nearby site under a similar crop, and is probably a reflection of the increasing root biomass and resultant root respiration as crop growth progresses (Koerber *et al*., 2010). It should be noted that the majority of available crop biomass data related to the S70 site, so stronger correlations might be expected than for sites with fewer data points.

2.4.2. Seasonal fluxes of nitrous oxide and environmental predictors

Nitrous oxide emissions observed over the monitoring period in this study (-75.24 ± 206.82 to 1660.82 ± 455.31 µg N_2O-N m^{-2} h^{-1}, section 2.3.2.2., Fig. 2.2.c.) were somewhat lower than vegetable crop emissions in the literature, although minimum fluxes were similar to those found elsewhere. Other studies of temperate and boreal arable Histosols have measured N_2O fluxes of 0 to 7083.33 µg N_2O-N m^{-2} h^{-1} (Elder and Lal, 2008; Flessa *et al*., 1998; Maljanen *et al*., 2002; Regina *et al*., 2004; Rochette *et al*., 2010; Weslien *et al*., 2012), with maximum mean vegetable crop fluxes within the lower end of that range (31.82 to 3182.48 µg N_2O-N m^{-2} h^{-1}, Regina *et al*., 2004; Rochette *et al*., 2010; Weslien *et al*., 2012). The pattern of seasonal N_2O emissions observed (peaks in spring and late summer or early autumn; lower emissions during winter) has also been noted in other studies, with peaks usually relating to mineral N application, cultivation, or post-harvest residue input (e.g. Elder and Lal, 2008; Rochette *et al*., 2010). At the study sites, crop establishment operations (tillage, mineral N application, irrigation, planting) are often practiced in tandem over a few days (G's Fresh, *pers. comm.*), so it is difficult to allocate N_2O peaks to specific management operations here. Peaks of N_2O associated with specific management operations (and rainfall events) can be highly episodic, and consequently may be difficult to capture using infrequent sampling strategies (De Klein and Harvey, 2013). It is possible that the relatively low peak emissions observed in this study compared to others, were a result of sampling at a time before or after the peak flux response to management events. Conversely, both Rochette *et al*. (2010) and Weslien *et al*. (2012) measured N_2O on a weekly basis, and may have captured a greater proportion of peak N_2O flux as a result.

The occurrence of N_2O maxima at these sites did largely coincide with crop management event clusters, and this seemed to be related principally to tillage events at the L20 site and to irrigation and fertiliser application events at S70. At the M35 site, N_2O emissions remained relatively high for the duration of the red beet crop grown in two out of three fields between May and October 2011, despite only sporadic management events coinciding with GHG sampling during that time (Fig. 2.2.c.). A similar seasonal pattern in a carrot crop was observed by Weslien *et al*. (2012), although the reason for elevated growing-season emissions was not determined.

Emissions of N_2O were significantly correlated with similar predictors to CO_2 emissions, although in general, the association was weaker for many variables. Despite crop aerial biomass being a significant predictor at the L20 site, its association with N_2O efflux was very strong, explaining 44% of variability in emission. The mechanism for N_2O production in this case may be related to basal leaf decomposition during the growing season, with larger plants providing a greater source of labile residue OM than smaller plants (Regina *et al*., 2004). This relationship may be weaker on the higher OM soils because the SOM provides a more consistent source of N for mineralisation, so leaf decomposition within the cropping season has a negligible effect on emissions. Other correlations, although weak, are consistent with the relationship between N_2O and environmental variables found in other studies (e.g. soil and air temperature, Maljanen *et al*., 2002; water table depth, Rochette *et al*., 2010; soil moisture content, Elder and Lal, 2008; soil N content, Flessa *et al*., 1998). The weak linear relationship between individual explanatory variables and N_2O emission has been observed by other authors (e.g. Regina *et al*., 2004), and in this case perhaps reflects the complex relationship between N_2O emission and crop management events, which each may influence a number of different soil physical and chemical variables concurrently. This makes it difficult to attribute effluxes to any one particular predictor.

2.4.3. Seasonal fluxes of methane and environmental predictors

Mean methane fluxes at the study sites (-161.91 ± 118.53 to 255.30 ± 398.95 µg CH_4-C m^{-2} h^{-1} over the monitoring period, section 2.3.2.3., Fig. 2.2.d.) were similar to those found on other cropped temperate and boreal peat soils (-97.33 to 116.67 µg CH_4-C m^{-2} h^{-1}, Elder and Lal, 2008; Flessa *et al*., 1998; Maljanen *et al*., 2002; 2004), but peak emissions in this study were considerably smaller than the 883.33 to 1411.25 µg CH_4-C m^{-2} h^{-1} recorded by Elder and Lal (2008). The difference in magnitude between maximum CH_4 emissions here and in Elder and

Lal's study may signify missed emissions peaks here, or they may reflect a response to the comparatively higher mean soil temperature found by those authors. No strong seasonal trend in methane emissions was found at our sites, with both maximum and minimum emissions being measured at similar times of year, during summer and autumn. The scarcity of strong predictors of methane emission found in this study was also observed by Elder and Lal (2008), and may be indicative of the high variability of fluxes compared to environmental predictors obscuring significant associations.

2.4.4. Cumulative GHG emissions

2.4.4.1. Comparison with other studies

Annual emissions in this study were dominated by CO_2 emission, followed by N_2O emission, whilst CH_4 emissions were small (section 2.3.3., Table 2.2.), similar to the pattern of emissions found in other studies of cropped peat soils (e.g. Elder & Lal, 2008; Maljanen *et al*., 2004).

Cumulative annual soil respiration rates from the soils in this study (13.04 ± 2.39 to 38.59 ± 1.56 t CO_2-e ha^{-1} yr^{-1} on a per-field basis, data not shown) are high compared to rates observed at other temperate or boreal cropped and bare soil sites (e.g. 6.9 to 20.0 t CO_2-e ha^{-1} yr^{-1}, Kasimir-Klemedtsson *et al*., 1997; Maljanen *et al*., 2004), but considerably lower than those found by Elder and Lal (2008) (69.25 to 82.44 t CO_2-e ha^{-1} yr^{-1}). The difference in annual CO_2 emission between this study and these others may be partially accounted for by differences in mean annual soil temperatures associated with the respective latitudes of each study's sampling locations. While the author is not aware of any studies that have monitored a full annual cycle of CO_2 emission from horticultural Histosols, the results in this study are similar to those estimated by Morrison *et al*. (2013) from a partial annual NEE budget made at the S70 site (10.63 ± 3.33 to 28.25 ± 3.81 t CO_2-e ha^{-1} yr^{-1}). Although gross primary productivity (GPP) was not accounted for in calculating net CO_2 emissions in this study, the difference in cumulative CO_2 emission between this study's results and those of Morrison *et al*. (2013) could be plausibly accounted for with photosynthetic activity.

The mean cropped CO_2 emissions observed here were greater than mean bare soil emissions (65.16 ± 9.64 to 102.60 ± 6.27 mg CO_2-C m^{-2} h^{-1}, compared with 46.46 ± 8.79 to 80.56 ± 5.28 mg CO_2-C m^{-2} h^{-1}, section 2.3.2.1.), but only by a relatively small amount, with the majority of cumulative annual soil respiration (58.1 to 95.9% on a per-field basis) a result of microbial rather than root respiration. The proportion of respiration, as root respiration, found here was very similar here at sites L20 and M35 (32.4 and 30.2% respectively; Table 2.2.), in keeping with the 35-45% reported by by Kasimir-Klemedtsson *et al*. (1997). Root respiration was considerably lower at the S70 site (7.9%), which may be attributable to high rates of microbial respiration at the S70 site (related to higher OM content) minimising the relative importance of root respiration within the total soil respiration budget.

In this study, both mean hourly CO_2 fluxes and cumulative annual fluxes related comparably to SOM content. On both bare and cropped soils, emissions increased by 57% as SOM increased from 20% to 35% (section 2.3.3., Table 2.2.). As SOM increased from 35% to 70%, a smaller increase was observed on bare soils (11%), and a decrease in emissions on cropped soils (13%). The author is not aware of any other published studies that have simultaneously measured emissions from soils of different SOM content within a small geographical area, but the comparable emissions from the M35 and S70 sites may be attributable to their similar mean topsoil (0-10 cm) SOM content (c. 58% and c. 67% respectively), compared to the greater difference in SOM content to 1 m depth (section 2.2.1). The slightly higher emissions from the M35 site may be a result of differences in cropping regimes, with more invasive tillage and harvesting operations involved in growing the dominant crops at the M35 site (red beet) than at the S70 site (lettuce, celery). Additionally, the cooler and wetter average weather conditions at the S70 site compared to the M35 site during 2011-2012 may have reduced its mean rate of CO_2 emission.

Cumulative N_2O emissions estimated in this study (1.51 ± 0.29 to 15.79 ± 4.30 t CO_2-e ha^{-1} yr^{-1} on a per-field basis) are within the lower range of arable peat soils emissions found in the literature (1.8 to 48.9 t CO_2-e ha^{-1} yr^{-1}, e.g. Kasimir-Klemedtsson *et al*., 1997; Weslien *et al*., 2012). This may reflect a difference in cropping, with the cereal crops on average yielding a higher N_2O efflux than vegetable crops. In comparison to other studies of vegetable cropping on Histosols, this study's annual emissions are very similar to those found under lettuce, celery and onion crops by Rochette *et al*. (2010) (1.75 to 19.58 t CO_2-e ha^{-1} yr^{-1}) and under potato by Regina *et al*. (2004) (5.07 ± 0.93 t CO_2-e ha^{-1} yr^{-1}) but lower on average than those observed under carrots by Weslien *et al*. (2010) (48.9 ± 3.34 t CO_2-e ha^{-1} yr^{-1}).

Although differences in cumulative N_2O fluxes between cropped and bare soils did not follow a consistent pattern in this study, higher annual emissions were observed from cropped soils in five out of seven fields (data not shown). Higher rates of soil respiration are known to relate to reduced soil aerobicity through increased oxygen removal (Rochette *et al*., 2010), as well as facilitating plant-mediated transport of N_2O (Maljanen *et al*., 2002); these factors potentially contribute towards higher annual N_2O fluxes when sustained over the cropping season.

Soil organic matter content did not appear to predict mean rates of N_2O emission *per se*, with the M35 site yielding the highest mean fluxes, and S70 the lowest. This may indicate an interaction between the similar top soil OM content at the M35 and S70 sites, and mineral N application rate and timing relative to sampling dates. All sites experienced a period of wet then warm weather in spring 2012 (Fig. 2.2.a.), followed by an N_2O peak. This peak accounted for a large proportion of the annual flux at the M35 site compared to the other sites, reflected in the relatively high mineral N application rate during this period of optimal weather conditions (c. 50-120 kg N ha^{-1}, compared with c. 60-70 kg N ha^{-1} at the S70 site and c. 80 kg N ha^{-1} at the L20 site). The high root crop-induced emissions during summer 2011 further contributed to the M35 N_2O budget compared with the other sites.

Methane emissions from temperate and boreal cropped and bare peat soils tend to be low (-0.15 to 0.25 t CO_2-e ha^{-1} yr^{-1}, e.g. Elder and Lal, 2008; Kasimir-Klemedtsson *et al*., 1997; Maljanen *et al*., 2002). Whilst these emissions are from cereal crops rather than vegetable crops, the emissions in this study (-0.02 ± 0.08 to 0.04 ± 0.05 t CO_2-e ha^{-1} yr^{-1}) are of a comparable magnitude.

The soil conditions experienced at these study sites (well-mixed, moist but unsaturated top-soils) indicated predominantly aerobic conditions, which are not conducive to significant CH_4 emissions (Le Mer and Roger, 2001). Given the lack of clear pattern in annual emissions with increasing SOM content (Table 2.2.), soil physical conditions appear to be dominant over SOM% in determining emissions at these sites. Cropping had a non-significant, but discernable effect on cumulative fluxes, being slightly higher on cropped soils than on bare soils, in five out of seven fields (data not shown). This may support the premise of soil moisture controlling CH_4 generation: crop roots accessing deeper, moister soil layers are known to conduct gases such as methane through their tissues to the atmosphere, bypassing aerobic upper soil layers where methane would otherwise be consumed in transit to the surface (e.g. Le Mer and Roger, 2001).

2.4.4.2. Comparison with IPCC default EFs

The most recently published UK National Greenhouse Gas Inventory (Webb *et al*., 2014) uses the IPCC (2000) Tier 1 default emission factors (EFs) to estimate N_2O and CH_4 emissions from temperate Histosols drained and cultivated for cropland, to give annual estimates equivalent to 3.90 t CO_2-e ha^{-1} yr^{-1} nitrous oxide, and “negligible” methane emissions. Carbon dioxide emissions from UK cultivated Histosols are estimated at 3.99 to 46.90 t CO_2-e ha^{-1} yr^{-1} (corresponding to peat of < 1 m depth and < 12% SOC content, and peats of > 1 m depth and > 12% SOC content, respectively), using Tier 3 modelled data based on English fen peats drained for agriculture before 1990.

Mean cumulative annual CO_2 emissions in this study were within the upper range estimated using the IPCC method, varying from 13.04 ± 2.39 to 38.59 ± 1.56 t CO_2-e ha^{-1} yr^{-1} on a per-field basis. The estimates provided from this study may be considered to be relatively robust in terms of capturing spatial within-field heterogeneity, having being sampled from well-mixed peats that have been repeatedly cultivated for at least 50 years (Morrison *et al*., 2013), and showing similar variation in soil characteristics to an independent study in a nearby location by Dawson *et al*. (2010). Given that well-aerated, homogenised peats tend to be less reactive than relatively undisturbed Histosols (e.g. Kasimir-Klemedtsson *et al*., 1997), and that the depth and SOC contents of the peats are intermediate between those corresponding to the IPCC upper and lower EFs, emissions from these field sites might be expected to be towards the lower range of the IPCC-predicted emissions. Consequently, the IPCC Tier 3 EFs for UK cultivated Histosols might be slightly under-predicting emissions, particularly from the L20 site with its low SOM content of 20%. Further refinement of Tier 3 CO_2 EFs may be required to reflect a gradient between high and low OM content arable peats, rather than using the lower resolution ‘high’ and ‘low’ categorisation currently recommended.

Estimated annual N_2O emissions in this study varied from 1.51 ± 0.29 to 15.79 ± 4.30 t CO_2-e ha^{-1} yr^{-1} on a per-field basis, and in all but one field were two to four times greater than the IPCC-estimated Tier 1 default value. At a national scale, the IPCC (2000) method sums the emissions from peat cultivation with emissions from mineral fertiliser application and crop residue application (each 1.25% of applied N). While insufficient crop residue data is available to allow calculation of residue EFs from the study sites, mineral N EFs at the study sites after deducting the Histosol EF were estimated at between 0% and 42% of N applied (assuming all N_2O emissions were directly caused by mineral N application). Given the typically low total N content of the salad crops that are typically grown on these sites, it is unlikely that residue N would account for a substantial portion of annual emissions. While some uncertainty may exist around the exact EF appropriate for fertiliser N application, it is likely that the underestimation of emissions at some sites is a result of the IPCC (2000) default EF for cultivated temperate Histosols being too low. Changes to the IPCC methods since 2000 support this argument – for example, the EF for fertiliser and crop residue application has been reduced to 1% of applied N (IPCC, 2006), implying a larger EF for cultivated Histosols, and more recently, the Tier 1 EF relating to temperate cultivated peats has been increased to 6.33 ± 2.34 t CO_2-e ha^{-1} yr^{-1} (IPCC, 2013). It is possible that even this value is an underestimate for the sites considered in this study, if significant emission peaks related to management events were not captured during field measurements (Section 2.4.4.2.). The inclusion or omission of such peaks in calculations of cumulative annual flux estimates can significantly affect the quality of derived EFs (De Klein and Harvey, 2013).

Observed CH_4 emissions (-0.02 ± 0.08 to 0.04 ± 0.05 t CO_2-e ha^{-1} yr^{-1}) may be considered to be within the “negligible” range described in the IPCC (2000) method, and the more recent quantitative estimate of 0 ± 0.06 t CO_2-e ha^{-1} yr^{-1} (IPCC, 2013). On the basis of this single year’s data, further modification of methane EFs from the soil surface is therefore unnecessary when predicting emissions from these sites.

2.4.5. Soil loss rates

Using annual C budget calculations and mean soil bulk density and SOC values from the three study sites, cumulative net gaseous carbon losses between 10th June 2011 and 9th June 2012 represent a soil loss rate of 0.42 to 0.63 cm soil yr^{-1} at the L20 site (based on a mean soil bulk density of 0.73 g cm^{-3} and a mean SOC content to 1 m of 11.6%), 0.51 to 0.94 cm soil yr^{-1} at the M35 site (based on a mean soil BD of 0.45 g cm^{-3} and a mean SOC content to 1 m of 24.7%), and 0.45 to 0.91 cm soil yr^{-1} at the S70 site (based on a mean soil BD of 0.34 g cm^{-3} and a mean SOC content to 1 m of 34.3%). While these values are low compared with the estimates of 1.10-1.48 cm soil yr^{-1} between 1982 and 2004 made by Dawson *et al.* (2010), they do fall within the range of 0.27-3.09 cm soil yr^{-1} computed by other authors surveying the East Anglia region over a longer time period (Richardson & Smith, 1977; Hutchinson, 1980). With mean peat depths in the region estimated at 1-2 m (Morrison *et al*., 2013), current rates of gaseous carbon losses may allow for c. 100-250 years of horticultural cropping under the current management and cropping rotations. However, it should be noted that this does not allow for losses via other routes (wind and water erosion, leaching of DOC, crop adherence, etc.), or for inter-annual or regional variation in weather conditions.

Variation in weather conditions can significantly affect estimates of mean annual GHG budgets, owing to the frequently significant relationships between emissions and temperature and rainfall, as well as responses to frost and snowmelt events (e.g. Rochette *et al*., 2010). The relatively low carbon emissions observed in this study compared to emissions from similar sites recorded by other authors (Section 2.4.4.1), may in part be explained by the period of prolonged drought in the east of England during most of 2011 (Morrison *et al*., 2013). Comparison of air temperature and rainfall averages over the 2011-2012 monitoring period in this study, and longer-term averages (1981-2010) for the East Anglia region (Met Office, 2014), show that both mean air temperature and rainfall were more typical for the region during January to June 2012 than between April to December 2011. If the predicted increase in air temperature commensurate with climate change predictions is also taken into account (Solomon *et al*., 2007), it is possible that the rate of soil loss will exceed that predicted from the net oxidative carbon losses observed in this study.

2.5. Conclusions

The results of this study suggest that the default IPCC (2000) EFs for annual CO_2 and N_2O emissions for cultivated Histosols may underestimate fluxes from intensively produced vegetable cropping systems, while CH_4 emissions were within a comparable range to the IPCC default values. When calculating losses of soil organic matter via GHG efflux to the atmosphere, attention should be given to the variability in emissions experienced under different weather conditions and different cropping regimes, which both vary inter-annually and geographically. Progressing towards Tier 2 (country-specific) and Tier 3 (modelling) approaches to N_2O emissions estimation, and improving on the existing Tier 3 method for CO_2 flux estimation, would aid progress towards more accurate quantification of variability between sites and years. Tier 2 estimates could be stratified according to site soil C content, nutrient status and broad cropping types. Tier 3 modelled estimates would ideally provide a detailed account of the influence of different cropping rotations, fertiliser management regimes, and variability in regional weather conditions.

The lower rates of GHG emission-related soil loss estimated from soils of lower organic matter content present a potential problem for policy-makers in incentivising growers to reduce GHG emissions from peat soils of higher organic matter content – if emissions slow as the SOM stock diminishes, while at the same time allowing continued production, growers may not be motivated to reduce emissions. Determining suitable cost-neutral or cost-negative mitigation measures, and investigating grower motivation to implement changes to farming practices which reduce net emissions, merit further attention.

Chapter 3

Estimating greenhouse gases emissions from horticultural peat soils using a DNDC modelling approach

Abstract

Peat soils represent an important global carbon sink, but can also provide a highly fertile medium for growing commercially important horticultural crops. Sustainable crop production on peat soils involves a trade-off between ensuring food security and minimising potentially substantial greenhouse gas (GHG) emissions and soil organic carbon (SOC) loss. There is an urgent need for quantification of GHG emissions from horticultural peat soils in the UK, as estimates are currently lacking. Quantifying the entire national range of net GHG fluxes through field data collection would be labour-intensive, costly and time-consuming. Utilising a process-based model driven by existing field data to estimate emissions provides a cost-effective alternative. This study evaluated the suitability of the Denitrification-Decomposition (DNDC) model for estimating emissions of CO_2, N_2O and CH_4 from horticultural peat soils in the UK. The model was parameterised using climatic, soil, and crop management data from two intensively cultivated sites on soils of contrasting organic matter contents (c. 35% and c. 70% SOM content). Simulated emissions of CO_2, N_2O and CH_4, and simulated soil physical and crop output values, were compared to measured emissions and soils estimates, for the 15-month period from April 2011 to June 2012. The performance of DNDC was assessed using baseline parameterisation, then calibrated using pre-simulation and sensitivity analysis processes. Data from two additional fields from each site was then used in a model validation exercise. Under baseline parameterisation conditions, DNDC was unsuitable for predicting GHG emissions and soil physical and crop variables from horticultural peat soils. Limited success was achieved even after attempts at model calibration and validation, in terms of both annual and episodic emissions prediction. Key constraints on model functioning appear to be its ability to model soil moisture, and some aspects of carbon and nitrogen dynamics, as well as the quality of input data relating to water table dynamics.

3.1. Introduction

Peat soils (Histosols) represent the second-most globally important carbon (C) stock after oceans (c. 3% of total land area, containing up to c. 30% of the total soil carbon stock). They also constitute some of the world's most agriculturally productive soils, so are important for food security (Parish *et al*., 2008). Managing peat soils for horticultural use is problematic because many farm operations facilitate conditions favourable for microbial mineralisation of the peat substrate (Dawson and Smith, 2007), frequently resulting in substantial rates of soil loss and greenhouse gas (GHG) emissions (Leifeld *et al*., 2011). While estimates of total GHG emissions from arable peat soils are scarce (Evans *et al*. 2011; Worrall *et al*. 2011), a recent study has estimated a combined annual emission of carbon dioxide (CO_2), nitrous oxide (N_2O) and methane (CH_4) at between c. 26 and 36 t CO_2-e ha^{-1} yr^{-1}, corresponding to a soil carbon loss of 0.42 to 0.94 cm soil yr^{-1} on cropped peat soils (Taft *et al*., 2013). Based on current soil depths, this equates to between c. 100 and 250 years before UK horticultural peatlands are lost completely. If agricultural production is to continue on peat soils, a balance needs to be achieved to preserve the remaining peat resource.

Quantifying rates of soil loss and finding ways to mitigate GHG emissions from horticultural peatlands requires careful consideration of the factors which regulate microbial production of GHGs (Li *et al*., 1992a). Accurate measurement of GHG emissions is also important for national level inventorying to comply with emissions reduction targets under the Kyoto Protocol (UK Parliament, 2008) and prioritising sectorial mitigation efforts. Emissions estimates are commonly derived from a combination of field-based measurements and computer modelling, with country-specific modelled emissions recommended for robust national inventorying purposes using the most detailed (Tier 3) IPCC calculation methods (IPCC, 2006). Although direct measurements provide estimates of GHG emissions under particular weather conditions and farm management systems, extrapolating such estimates to other scenarios may not provide robust predictions of emissions. This is because the factors driving soils' emissions are numerous, often interact in complex ways, and may be highly spatially and temporally variable (Li, 2007; Giltrap *et al*., 2010). Since a substantial research effort would be required to capture a representative proportion of variability to allow estimation of emissions under all conceivable combinations of field conditions, an alternative approach is required. Simulating emissions using mathematical models in tandem with direct measurement campaigns helps to overcome this problem, provided that, (1) The model chosen simulates the system it represents within acceptable error margins over a sufficient time scale, i.e. it is scientifically robust (Giltrap *et al*. 2010); (2) There are sufficient input data to represent a substantial proportion of the expected variability found under field conditions, so that the model may be calibrated at the correct spatial scale (Peltoniemi *et al*., 2007), and (3) the model can be validated.

A number of models are available for simulating emissions from soil-crop systems (Chen *et al*., 2008; Thomas *et al*., 2013), or from peatlands (Farmer *et al*., 2011), and can be broadly categorised as empirical models or mechanistic (process-based) models. Empirical models, utilising simple correlations between variables, tend to require a smaller input effort but often produce unreliable outputs when there is a great deal of heterogeneity or complex feedbacks in the system, as is the case with soil-crop GHG emissions. Process-based models generally require a greater input effort, but since they incorporate a representation of the complex processes and interactions within a system, they are best equipped for reliably predicting and understanding site-scale GHG fluxes under a range of conditions (Bell *et al*. 2012). Two such models which have been used to simulate emissions from agro-ecosystems are the ECOSSE (Estimating Carbon in Organic Soils – Sequestration and Emissions) and DNDC (DeNitrification-DeComposition) models.

ECOSSE is a recently developed process-based model, designed to simulate carbon and nitrogen dynamics specifically in organic and organo-mineral soils (Smith *et al*. 2010b). While it has been applied to a number of land-use types including arable crops (e.g. Khalil *et al*. 2013; Bell *et al*. 2012), its input module does not currently allow for parameterisation of multiple crops within an annual cycle (Smith et al., 2010c). For this reason, it is currently unsuitable for application to the field sites in this study. Conversely, DNDC is a highly customisable model, but was originally developed to model N_2O, CO_2, and N_2 efflux (and later, CH_4 fluxes) from US agricultural mineral soils (Li *et al*., 1992a; Li, 2000). The small suite of model versions now in circulation, have been applied to a range of ecosystems (e.g. arable, wetlands, forestry), in a range of climatic zones (Gilhespy *et al.*, 2014; Giltrap *et al*., 2010). The model has been used to compile national agricultural emissions estimates, including in the UK, so offers a precedent which might be further explored (e.g. Brown *et al*., 2002). Although a number of model versions possess the potential to yield useful emissions estimates for UK horticultural peat soils (Gilhespy *et al*., 2014), the specific application of DNDC to eutrophic Histosols has been very limited to date (e.g. Li *et al*., 1992a, b, 1994, 2010; Maljanen *et al*., 2012; Webster *et al.*, 2013). Of the sites modelled in those papers, only one was under current arable use (sugar cane production in a subtropical region of Florida; Li *et al.*, 1992a; b; 1994); the remaining sites were fen peats under native or semi-natural vegetation in boreo-temperate zones. To the author's knowledge, no site-specific DNDC simulations of UK horticultural peats have been conducted. Since peat soil biogeochemical processes and responses to agricultural operations differ from those of mineral soils, sometimes unreliable results are observed from models originally designed for application to mineral soils (Smith *et al.*, 2010b). Given its limited prior application to temperate horticultural peatlands, and the uncertainty surrounding its suitability for modelling Histosol C and N dynamics, there is scope for further investigation into the applicability of DNDC for predicting GHG emissions from these agroecosystems.

This study aimed to evaluate the performance of DNDC, in estimating emissions of CO_2, CH_4 and N_2O from a number of rotations of commercially important crops on soils of differing, high organic matter content (c. 70% SOM and c. 35% SOM respectively). Model performance was evaluated by comparing measured and modelled emissions in terms of (1) total annual GHG emissions and (2) seasonal patterns of emissions, and (3) by comparing measured environmental variables (soil and crop characteristics) with modelled environmental characteristics. Primarily, this work aimed to identify the model parameters which most strongly influenced emissions, and to assess whether the model could be adequately calibrated for predicting emissions.

3.2. Methods and materials

3.2.1. Model description: DNDC

A detailed description of the model structure, functioning and parameterisation, is given in Li *et al*. (1992a, b), Li (2000), and ISEOS (2012). The basic model structure is outlined here.

The DNDC model simulates GHG fluxes within an agro-ecosystem by determining the collective effect of integrated input parameters on the soil environment, and in turn, the effect of the soil environment on carbon (C) and nitrogen (N) cycling and GHG emissions. The simulation incorporates two main levels or components (Figure 3.1.). Elements of component one, *ecological drivers*, are inputted by the user, and comprise characteristics of climate, soil, vegetation (crops), and anthropogenic activity (farming operations); the combined effect of these inputs drives the equations in three sub-modules (*soil climate*, *crop growth*, and *decomposition*), to predict values for each aspect of component two. Component two comprises a range of *soil environmental variables*, namely temperature, moisture content, pH, redox potential (Eh), and C and N substrate concentrations. Soil variable values are combined through a further three sub-modules (*nitrification*, *denitrification*, and *fermentation*), resulting in the prediction of emissions of CO_2, N_2O and CH_4, as well as NO, N_2 and NH_3 gases. Equations linking model sub-components were derived from appropriate combinations of the classical laws of physics, chemistry and biology, and empirical values drawn from laboratory observations. Many of the input values driving these equations may be user-modified (described in section 3.2.3 and Tables 3.1., 3.2. and 3.3.).

The six sub-modules operate in the following way:

1. The *soil-climate* sub-module calculates soil heat and moisture transfers through the soil profile layers (c. 5 cm horizontal layers), on an hourly and daily time step. This module also computes O_2 diffusion through the soil, and Eh. Transfers of heat and water through each layer are determined by soil texture, and soil temperature and moisture potential gradients. Soil texture values (clay content, soil density) influence the soil hydraulic properties, wilting point (WP), and field capacity (FC), and heat transfer rate. Daily climatic data (air temperature, rainfall) directly affect soil moisture and temperature, and indirectly affect soil moisture via simulated evapotranspiration. Soil moisture status determines soil O_2 concentration and Eh, which regulate the activity of GHG generating microbes, and therefore the magnitude and proportion of each GHG

emitted. Following rainfall events, soil moisture is simulated by sequentially filling each layer down the soil profile until the point of saturation is reached.

2. The *crop growth* sub-module simulates crop growth on a daily time step. Biomass accumulation is driven by crop management input values, and outputs from the *soil-climate* sub-module: solar radiation (determined by latitude), temperature, and water and N availability within the rooting zone. Crop biomass C and C/N ratios are partitioned into four fractions: root, leaf, stem, and grain (harvested fraction, section 3.2.3.). Upon crop harvest, the whole root fraction and a user-specified portion of the leaf and stem residue fraction are transferred to the C and N pools in the *decomposition* sub-module. The *crop growth* sub-module feeds back into the *soil-climate* sub-module as the crop grows and uses C, N and water, affecting soil temperature, moisture, pH, Eh, DOC, and available N concentrations.
3. Under simulated aerobic conditions, the *decomposition* sub-module computes rates of decomposition, and resultant NH_3 and CO_2 production, on a daily time interval. Simulated organic matter (OM) is allocated to litter, microbial, humad, and humus pools. Humads are defined as *materials partially stabilised by humification and adsorption*, while humus is described as the stable portion of SOM. Decomposition occurs in the decomposable litter, microbial biomass, and humad pools. With the exception of humus, each pool is split into labile and resistant fractions, which are assigned different decomposition rates and C/N ratios. Rates of decomposition may be modified by the user, and are also limited by soil texture, temperature, and moisture properties from the *soil-climate* sub-module. Simulated mineralisation in the *decomposition* sub-module generates substrates (DOC, NH_4^+, NO_3^-), which feed back into the *crop growth* sub-module when assimilated by plants, or are lost through leaching or volatilisation to NH_3, adsorbed onto clay particles, re-assimilated by microbial biomass, or provide input substrates for the *nitrification*, *denitrification* and *fermentation* sub-modules.
4. The *nitrification* sub-module predicts the rate of nitrification under aerobic conditions, and the proportion of NO and N_2O gases produced. Nitrification rates are determined by growth and death rates of nitrifying bacteria, and soil properties (temperature, moisture, pH, and availability of DOC and NH_4^+).

5. The *denitrification* sub-module predicts the rate of denitrification in anaerobic soil conditions, as well as the proportion of NO, N_2O and N_2 gases produced. Denitrification rates are determined by growth and death rates of denitrifying bacteria, and soil properties (temperature, moisture, soil aeration, pH, and availability of the substrates DOC, NH_4^+, NO_3^-, NO_2^-, NO, and N_2O). Aerobic and anaerobic microsites in the soil matrix are simulated as a function of shifting soil O_2 concentration, and denitrification is assumed to occur only within anaerobic microsites. This sub-module is activated after each simulated rainfall event.

   The magnitude of N_2O generation from both *nitrification* and *denitrification* sub-modules is determined by the rate of gas diffusion from the soil to the atmosphere, in turn determined by soil temperature, moisture status, density, and clay content. Variation in these factors can result in transformation of a portion of generated N_2O, prior to reaching the soil surface.

6. The *fermentation* sub-module computes the flux (production, transport and consumption) of CH_4, and is activated under persistent anaerobic conditions. The balance between CH_4 production and consumption is a function of DOC concentration, soil temperature, degree of plant-mediated transport (from the *crop growth* sub-module), soil porosity, and the proportion of oxygenated microsites in the soil (in which CH_4 is consumed by methanotrophs before reaching the soil surface). The model allows for simultaneous CH_4 production and consumption.

Simulated DNDC outputs include daily profiles of soil variables (e.g. temperature, moisture, Eh, pH, and concentrations of total SOC, NO_3^-, and NH_4^+), and daily fluxes of CO_2, N_2O and CH_4. Annual estimates of each GHG flux are also provided.

A detailed description of field sampling methods is given in Chapter 2, and is summarised here. Emissions of CO_2, CH_4 and N_2O were measured using the closed chamber method, approximately once per month between April 2011 and June 2012 inclusive. In tandem with GHG measurements, soil temperature and water table depth were measured, and soil samples were extracted for analysis of moisture content, bulk density, and extractable NO_3^- and NH_4^+ content. Where practicable, crop samples were removed at harvest, for estimation of above-ground harvested crop biomass. Soil cores were extracted to a depth of 1 m in January 2012 for analysis of soil carbon and nitrogen stocks. Two sites with soils of different organic matter (SOM) contents were included in this study: S70 (c. 70% SOM; 3 fields sampled: S1, S2, and

S3), and M35 (c. 35% SOM; 3 fields sampled: M1, M2 and M3). The study fields contained a number of commercially important crops including lettuce, celery, potato, radish and red beet (beetroot), in various differing rotations.

3.2.2. DNDC simulations and comparison with field data

Model version DNDC 9.5 (December 2013 release) was used for all simulations. Only cropped soils from the field study were included in this study; bare soils were omitted from simulation. Modelled data were compared against measured data using two types of metrics, where appropriate:

1. Cumulative annual emissions of CO_2, N_2O and CH_4 over the period 10th June 2011 to 9th June 2012, and total annual harvested biomass, were compared on a percentage basis to give a Relative Deviation value (*RD*, after Abdalla *et al.*, 2011):

   $$RD = \frac{(P - O)}{O} \times 100 \tag{3}$$

   where *P* is the annual predicted (simulated) cumulative total, and *O* is the annual observed cumulative total.

2. Episodic daily emissions of CO_2, N_2O and CH_4, and environmental variables (soil temperature, soil water-filled pore-space (WFPS), soil NO_3^-, soil available NH_4^+, and pH), were compared using the procedure and equations detailed in Smith *et al.* (1997). All compared measured variables had been replicated in the field, so the following statistics were used to assess model fit, at a significance level of 0.05 where appropriate:

   a. The *F* value of the lack-of-fit statistic (*LOFIT*), for overall model error relative to measurement error;
   b. The *t* value of the root mean squared error (*RMSE*), for collective error in the prediction of individual data points;
   c. The relative error statistic (*E*), for total model bias;
   d. The sample correlation coefficient (*r*), for linear correlation between modelled and measured values.

> Episodic daily water table depths were not replicated, so simulated values could only be compared to observed values in terms of *RMSE* (without a $t_{0.05}$ value) and *r*, and the mean difference statistic (*M*) to evaluate model bias.

Where simulated output variables did not exactly match available measured data, the closest modelled equivalent was selected for comparison. Thus, measured soil temperature (0-10 cm) and WFPS (0-5 cm) were compared to modelled soil temperature and WFPS at 5 cm depth, and measured soil NO_3^- and exchangeable NH_4^+ (both measured at 0-5 cm) were compared to modelled soil NO_3^- and exchangeable NH_4^+ at 0-10 cm depth.

All comparisons were performed in Microsoft Excel 2010. Episodic data were compared using the MS Excel-based MODEVAL evaluation model, after Smith *et al*. (1997).

3.2.3. Baseline model run

One field from each of the S70 and M35 sites was chosen for the baseline and subsequent calibration model runs, using DNDC in site mode. The fields (M1 and S3) were chosen with the aim of including as much variation in cropping as possible. Each cropping year was simulated separately for each field, rather than as a continuous cropping system, to allow for the inclusion of pre-simulation years (section 3.2.4.).

An initial baseline simulation was conducted on fields M1 and S3, to test the suitability of the model for predicting emissions ‘off the shelf’. The simulation was run for a single year. As far as possible, default model parameters were used, with the exception of model variables that are normally adjusted by the operator to incorporate site- (in this case, field-) specific data. Adjusted parameters are described below.

3.2.3.1. Climatic data

Weather data were obtained from the UK Met Office via the British Atmospheric Data Centre (BADC, 2014). Default data requirements were inputted into the model, comprising daily minimum and maximum air temperatures (°C), and daily rainfall (cm). Mean weather variables were computed separately for each site, from Denver and Marham weather stations for site S70, and Brooms Barn and Chettisham weather stations for site M35.

Atmospheric parameters were assigned the same values for both sites, and were based on mean values for the region, separated by year where possible. Total N concentration in rainfall (NO_3^-

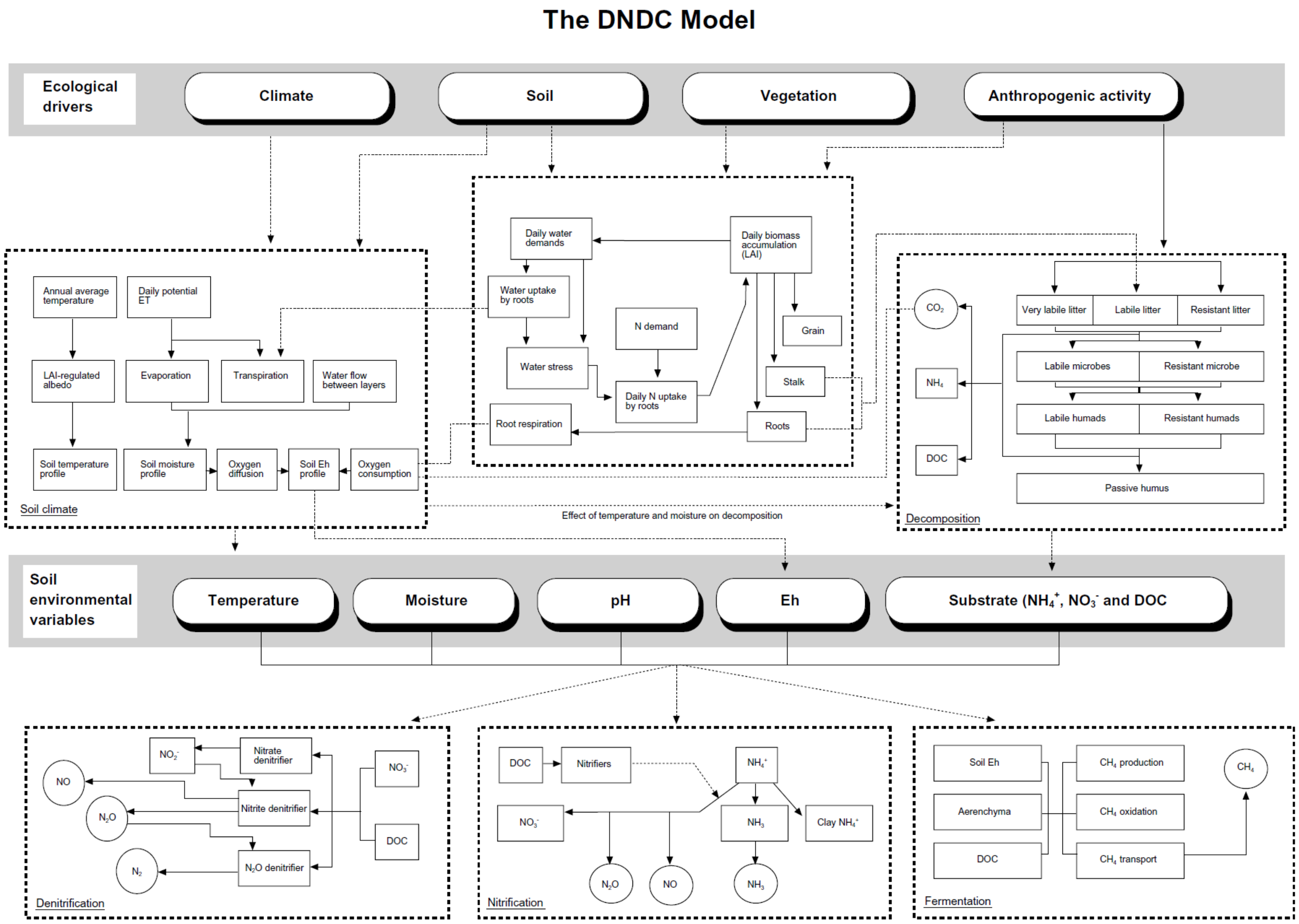

Figure 3.1. Schematic representation of the DNDC model structure, comprising two components of (1) Ecological drivers, and (2) Soil environmental variables (Li, 2000).

+ NH_4^+) was 0.92 mg N L^{-1} (estimated from DEFRA, 2011). Atmospheric background concentration of CO_2 was 390 ppm (IPCC, 2013a), and NH_3 concentrations were 1.704 and 1.273 µg N m^{-3} for 2011 and 2012 respectively (estimated from DEFRA, 2013). For the purposes of this study, atmospheric CO_2 concentration was assumed to be stable over time.

3.2.3.2. Soils data

From the available arable land use types (*Upland crop field*, *rice paddy field*, and *wetland*), *rice paddy field* and *wetland* were considered unsuitable for simulations here, as their water table dynamics are based on responses to frequent site flooding (ISEOS, 2012). As crop flooding did not occur at the sites in this study, *Upland crop field* was selected for all simulations. The remaining soil parameters inputted into the *Soils* sub-module, were based on the selection *Define soil texture by specifying Top soil (0-10 cm) texture*. Default model values corresponding to soil texture *Pristine peat soil* (site S70) and *Cultivated/Drained peat soil* (site M35) respectively, were combined with field-specific values for some variables (Table 3.1.).

Default values were retained for soil WFPS at field capacity (0.55) and wilting point (0.26), and for clay fraction (0.06), hydraulic conductivity (0.015 m h^{-1}), and soil porosity (0.701). Soil structure variables remained at a bypass flow rate of 0, and a drainage efficiency of 1; the depth of water retention layer was changed to 1.25 m for both sites, based on the mean estimated depth of transition from peat to underlying material (own data; Dawson *et al.*, 2010). Microbial activity index, slope, soil salinity index, and rain water collection index, were all maintained at their default values (1, 0, 0 and 1 respectively). Functions relating to water erosion estimation (SCS and MUSLE) were not used in this study. Default values relating to the decomposition of SOC pools (litter, humads, and humus; all with a value of 1) were retained.

Soil bulk density, pH, and initial NO_3^- and NH_4^+ concentrations were inputted as mean values measured in January 2012 (Table 3.1.). In addition to these measurements, measured soil C and N data were used to re-define the SOC within the 0-10 cm soil layer, and SOC profile parameters in the *Soils* sub-module (Table 3.1.). Soil cores were extracted from each farm in January 2012 using a bulk density ring (d = 5 cm, V = 100 cm^3), for the full depth of the top soil (0-10 cm), and every 10 cm below this, to 1 m depth. Total C and N analysis was conducted on oven-dried (105°C) soil, using a TruSpec CN analyser (Leco Corp., St Joseph, MI, USA). When details of SOC and soil N are inputted, DNDC automatically adjusts SOC fractions and

C/N ratios. Inputted SOC partitioning values, and corresponding model-adjusted values, are given in Table 3.1.

3.2.3.3. Cropping data

None of the crops in the calibration test fields S3 and M1 were overwintered, so all cropping systems were treated as one cropping system per year (Table 3.2.).

Crops were selected from the existing model database, and default values initially retained for all physiological metrics with the exception of thermal degree-days (TDD) for maturity, and maximum biomass production (in terms of carbon). Thermal degree-days were estimated separately for each crop, by multiplying the number of days between crop planting and harvest, by the cumulative mean daily temperature over the cropped period. Grain yield was calculated using estimated yields from farm records, field measurements and values from the literature, with carbon content assumed to be 0.4 kg C kg^{-1} DW (ISEOS, 2012). Changing both the TDD and biomass production values caused an automatic change in the values for maximum biomass production of leaf, stem and root, and crop N demand. Default values for N fixation index (1), vascularity (0), and optimum temperature for growth (25ºC, except for radish, 15ºC) were retained. Tree maturity age was ignored in all simulations, remaining at the default value of -1.07×10^8. Default and adjusted crop parameters used in the calibration are given in Table 3.2.

Crop management values relating to tillage, fertilisation, irrigation, flooding, and plastic application were taken from farm records unless otherwise stated, and are detailed in Table 3.3. The *Manure Amendment* and *Grazing or cutting* tabs were ignored, since no fields underwent these types of management at any point during the simulated period.

The harvested fraction of any crop is modelled as “grain” by DNDC, regardless of which part of the plant is harvested (Li *et al.*, 1997). Therefore, it is assumed here that the fraction of leaves and stems left in the field after harvest was 1 (i.e. 100%), since any harvested leaves would have been included in the modelled grain fraction. Efforts were made to match as closely as possible the stated tillage method with the available methods in DNDC, based on estimated depth of tillage. Manual fertiliser application was selected to allow separate input of each mineral N fertilisation event. Quantities of applied fertiliser were calculated from the amount of applied fertiliser stated in farm records, taking into account the % N in the formulation, and whether it was solid or liquid fertiliser. Information relating to the N form in each fertiliser was

Table 3.1. Soils input data used for DNDC baseline, calibration and validation simulations.

Input category	**Input variable**	**Units**	**Field, (mean baseline value [1]), mean calibrated or validated value [1]**					
			M1	**M2**	**M3**	**S1**	**S2**	**S3**
Climate	Latitude		(52.445)	52.445	52.452	52.522	52.531	(52.524)
Soil	Bulk density	g cm^{-3}	(0.39)	0.44	0.49	0.34	0.34	(0.33)
	pH		(6.96)	6.87	6.04	5.95	6.04	(6.66)
	SOC at surface soil (0-10 cm) [2]	kg C kg^{-1} soil	(0.335)	0.335	0.280	0.349	0.349	(0.349)
	Depth of top soil with uniform SOC content	m	(0.2)	0.2	0.2	0.2	0.2	(0.2)
	SOC decrease rate below top soil	Rate, 0.5 to 5.0	(0.55), 0.86	0.86	0.86	0.8	0.8	(0.5), 0.8
	Soil partitioning: re-define	Yes / no	(Yes)	Yes	Yes	Yes	Yes	(Yes)
	Bulk C/N [3]	Ratio	(10.09)	10.09	10.09	10.09	10.09	(10.09)
	V. l. litter (fraction, C/N)	Fraction 0-1, Ratio	(**0, 5**)	**0, 5**	**0, 5**	**0, 5**	**0, 5**	(**0, 5**)
	L. litter (fraction, C/N)	Fraction 0-1, Ratio	(**0, 25**)	**0, 25**	**0, 25**	**0, 25**	**0, 25**	(**0, 25**)
	R. litter (fraction, C/N)	Fraction 0-1, Ratio	(**0.01, 100**)	**0.01, 100**	**0.01, 100**	**0.01, 100**	**0.01, 100**	(**0.01, 100**)
	Humads (fraction, C/N) [3]	Fraction 0-1, Ratio	(0.17, 10)	0.17, 10	0.13, 10	0.17, 10	0.17, 10	(0.17, 10)
	Humus (fraction, C/N) [3]	Fraction 0-1, Ratio	(0.82, 10)	0.82, 10	0.86, 10	0.82, 10	0.82, 10	(0.82, 10)
	IOC (fraction, C/N)	Fraction 0-1, Ratio	(**0, 500**)	**0, 500**	**0, 500**	**0, 500**	**0, 500**	(**0, 500**)
	Initial NO$_3^-$ conc. at surface soil	mg N kg^{-1} soil	(16.71)	4.53	24.09	14.20	11.29	(10.02)
	Initial NH$_4^+$ conc. at surface soil	mg N kg^{-1} soil	(24.57)	16.84	26.02	34.65	28.65	(27.06)

[1] *Measured in January 2012, unless stated otherwise. Bold text indicates model default values.* [2] *Measured mean SOC concentration, 0-20 cm depth; n= 3.* [3] *Automatically adjusts when surface soil SOC is modified.*

unavailable, so the N component of each fertiliser was assumed to be 50% ammonium and 50% nitrate, unless otherwise stated in farm records. It was assumed that none of the fertilisers were controlled-release, and were always surface applied ($d = 2$ mm), unless otherwise stated in farm records. Irrigation events were drawn from farm records, and volumes of water applied converted to cm of water applied when necessary. All irrigation was applied using overhead sprinkler systems. Measured changes in water table level (including use of sub-surface irrigation on the S70 site) were modelled by loading pre-prepared files into the *Observed water table data* option within the *Flooding* tab. Early crops on the farms in this study were covered with horticultural fleece, and the radish crop in field S2 was covered with fine mesh to protect against cabbage root fly; both of these covers were selected as *Film mulch* in the *Plastic* tab. Since all experimental plots were entirely covered with fleece or mesh, the modelled fraction of covered ground was set at 1 (i.e. 100%).

Particularly in larger fields, a single management operation may have been applied over a number of days; where this was the case, the first date of the entire application was taken as the event date.

3.2.4. Optimising model stability using pre-simulation

Commonly, modelled output values may take a number of years after initialisation to stabilise (Peltoniemi *et al*., 2007). In order to minimise simulation error caused by instability in initial modelled soil conditions, the second stage of calibration in this study involved repeat-running each baseline data file for fifty simulated years. This allowed visual identification of a point of sufficient stability within this period, of SOC and soil N pools and soil WFPS (A. Hastings and J. Yeluripati, *pers. comm.*; Appendix C). All subsequent stages of simulation used the relatively unstable period as a pre-simulation period, and the final (relatively stable) year as the simulated year for comparison with observed data.

3.2.5. Sensitivity analysis

Following baseline model parameterisation and pre-simulation, the sensitivity of annual cumulative emissions of CO_2 (soil respiration), N_2O and CH_4 to all relevant model input variables, was tested using the baseline pre-simulated data. Only one variable was altered for each sensitivity test, while all others were kept constant. The only exception to this was in testing the sensitivity of model outputs to crop parameters – in this case, default values for all crop fractions together were compared to modified values. Where possible, the range of values tested for each variable (and the steps within that range), was based on realistic bounds suggested by field measurements over the annual cycle, farm records, or values from suitable sources of scientific and grey literature. Where potential values could not be found in this way, attempts were made to gauge sensitivity by including some extreme values in the analysis (for example, with regards to SOC pool partitioning and decomposition rates). The results of the sensitivity analysis were used to inform the subsequent model calibration procedure.

3.2.6. Model calibration and validation

Model calibration was attempted for the two trial fields M1 and S3, by modifying the parameters that showed greatest absolute or percentage response to adjustment during the sensitivity analysis. This was done using a stepwise approach, and aimed to calibrate the simulated data to measured data by varying as few model parameters as possible. At each stage, the magnitudes of annual emissions and crop biomass fractions were assessed using the *RD* statistic, and the seasonal pattern of emissions and selected soil variables were examined visually, before further adjustment. During the final stages of calibration, the fit statistics described in section 3.2.2. were additionally used to augment visual assessments of seasonal patterns of emissions and soil variables.

The model validation procedure was conducted on the two remaining fields from site M35 (fields M2 and M3), and the two remaining fields from site S70 (fields S1 and S2). Years 2011 and 2012 were again modelled separately for each field, with each simulation incorporating a pre-simulation period defined in section 3.2.4. Field-specific data relating to soils, crops, and management operations were used where possible (Tables 3.1., 3.2. and 3.3.). Input variables modified during the calibration stage were then adjusted accordingly, for each field. Output variables were compared to measured data visually, and using the annual *RD* statistic and episodic data fit statistics.

Table 3.2. Crop input data used for DNDC baseline, calibration and validation simulations.

Input variable		**Units**	**Crop, (mean baseline value), [1] mean calibrated or validated value [1]**						
			Lettuce, Iceberg	**Lettuce, Romaine**	**Celery**	**Red Beet**	**Potato**	**Radish**	**Mustard cover crop**
Crop type			(34, Lettuce)	34, Lettuce	(22, Celery)	(19, Beet)	(18, Potato)	60, Radish	57, Mustard
Perennial crop?		Yes / No	(**No**)	**No**	(**No**)	(**No**)	(**No**)	**No**	**No**
Cover crop?		Yes / No	(**No**)	**No**	(**No**)	(**No**)	(**No**)	**No**	Yes
Fraction of leaves and stems left in field after harvest		Fraction 0-1	(1)	1	(1)	(1)	(1)	1	0.1 [18]
Max. biomass production	Grain	kg C ha^{-1} yr^{-1}	(650), 2600	1000	(1400), 5600	(2200), 8800	(4500), 18000	1600	80
	Leaf [2]		(81), 1544	1409	(47600), 4738	(293), 5225	(836), 3682	756	2760
	Stem [2]		(81), 1544	773	(47600), 4738	(293), 5225	(836), 6955	756	2760
	Root [2]		(203), 2438	1364	(43400), 6462	(147), 8250	(257), 12273	1333	2400
Biomass fraction	Grain	Fraction 0-1	(**0.64**), 0.32	0.22	(**0.01**), 0.26	(**0.75**), 0.32	(**0.7**), 0.44	0.36	0.01
	Leaf		(**0.08**), 0.19	0.31	(**0.34**), 0.22	(**0.1**), 0.19	(**0.13**), 0.09	0.17	0.35
	Stem		(**0.08**), 0.19	0.17	(**0.34**), 0.22	(**0.1**), 0.19	(**0.13**), 0.17	0.17	0.35
	Root		(**0.20**), 0.30	0.30	(**0.31**), 0.30	(**0.05**), 0.30	(**0.04**), 0.30	0.30	0.30
Biomass C/N ratio	Grain	Ratio	(**11.5**)	**11.5**	(**12**)	(**58**)	(**60**)	**19**	**15**
	Leaf		(**20**)	**20**	(**12**)	(**50**)	(**60**)	**45**	**25**
	Stem		(**20**)	**20**	(**12**)	(**50**)	(**60**)	**45**	**25**
	Root		(**30**)	**30**	(**21**)	(**80**)	(**60**)	**44**	**45**
Annual N demand [2]		kg N ha^{-1} yr^{-1}	(71), 462	242	(10117), 1564	(52), 464	(144), 682	148	280
TDD for maturity			(690 [3], 566 [4]), 712 [5], 748 [6], 739 [7], 713 [8], 669 [9], 868 [10]	738 [11], 802 [12]	(1213 [14]), 1623 [13]	(2772 [15,16]), 2574 [17]	(2316)	365	2127
Water demand		g H_2O g DM^{-1}	(**800**)	**800**	(**500**)	(**318**)	(**415**)	**508**	**150**

Where calibration or validation values are unmodified from the baseline values, no additional values are given. Identical crop variety biomass values were used across all fields and positions in the annual rotation.

[1] Bold text indicates model default values. [2] Automatically adjusts when crop C/N ratios or biomass is modified. [3] M1 2012, 1st crop. [4] M1 2012, 2nd crop. [5] M2 2012, 1st crop. [6] M2 2012, 2nd crop. [7] M3 2011 crop. [8] S2 2011 crop. [9] S2 2012, 1st crop. [10] S3 2012, 2nd crop. [11] S1 2011, 1st crop. [12] S1 2011, 2nd crop. [13] S1 2012 crop. [14] S3 2011 crop. [15] M1 2011 crop. [16] M2 2011 crop. [17] M3 2012 crop. [18] Assumed harvested with leaves on.

Table 3.3. Farm management data used for DNDC baseline, calibration and validation simulations.

Field, year	Crop	Growth period [1] dd/mm, (days)	Tillage dd/mm, method [2], (depth)	Fertiliser dd/mm, type, (N applied, kg N ha^{-1})	Ground cover dd/mm, (days), type [3]	Irrigation dd/mm, (rate, cm)
M1, 2011	Red beet	22/04 - 20/10 (181 d)	22/04, P, (20 cm) 22/04, PH, (10 cm) 22/04, BF, (20 cm) 04/05, MH, (5 cm) 28/05, MH, (5 cm) 31/10, SS, (30 cm)	08/06, CAN, (50.0) 30/06, CAN, (50.0) 20/07, CAN, (50.0)		05/05, (2.5) 27/05, (2.5) 27/06, (2.5)
M1, 2012	Lettuce (Iceberg)	24/03 - 31/05 (68 d)	20/03, P, (20 cm) 24/03, PH, (5 cm) 24/03, PH, (5 cm)	20/03, NPK, (33.2) 21/03, N37, (116.9) 27/03, NPK, (35.3) 07/04, NPK, (0.1) 02/05, NPK, (0.01) 23/05, NPK, (0.14)	24/03 - 17/04 (24 d), HF	23/03, (2.0) 28/03, (1.5) 28/05, (3.0)
	Lettuce (Iceberg)	17/07 - 22/08 (36 d)	17/07, DP, (30 cm) 17/07, P, (20 cm) 17/07, PH, (5 cm) 17/07, PH, (5 cm)	17/07, NPK, (35.3) 19/07, NPK, (0.12) 25/07, NPK, (0.16) 03/08, NPK, (0.16) 07/08, NPK, (0.12)		12/07, (0.5) 22/07, (2.5) 20/08, (0.5)
M2, 2011	Red beet	22/04 - 20/10 (181 d)	22/04, P, (20 cm) 22/04, PH, (10 cm) 22/04, BF, (30 cm) 04/05, MH, (5 cm) 28/05, MH, (5 cm) 31/10, SS, (30 cm)	08/06, CAN, (50) 30/06, CAN, (50) 20/07, CAN, (50)		04/05, (2.5) 25/05, (2.5) 27/06, (2.5) 15/07, (2.5)
M2, 2012	Lettuce (Iceberg)	28/03 - 05/06 (69 d)	20/03, P, (20 cm) 28/03, PH, (5 cm) 28/03, PH, (5 cm)	20/03, NPK, (28.0) 26/03, N37, (124.1) 28/03, NPK, (35.3) 07/04, NPK, (0.1) 04/05, NPK, (0.1) 23/05, NPK, (0.14)	28/03 - 17/04, (28 d), HF	27/03, (1.5) 02/04, (1.5) 30/05, (2.5)
	Lettuce (Iceberg)	11/07 - 22/08 (42 d)	11/07, DP, (30 cm) 11/07, P, (20 cm) 11/07, PH, (5 cm) 11/07, PH, (5 cm)	11/07, NPK, (35.3) 19/07, NPK, (0.12) 25/07, NPK, (0.16) 02/08, NPK, (0.16) 07/08, NPK, (0.12)		23/07, (2.0) 04/08, (2.5)
M3, 2011	Lettuce (Iceberg)	23/06 - 06/08 (44 d)	18/06, P, (20 cm) 23/06, PH, (10 cm) 11/07, MH, (5 cm)	23/06, NPK, (32.6) 24/06, N37, (59.7) 24/06, NPK, (27.7) [4] 11/07, N37, (48.7) [4]		23/06, (1.5) 27/06, (1.5) 28/07, (2.5)
	Mustard (CC)	31/08 – 28/04 (241 d)	07/08, D, (10 cm)			
M3, 2012	Red beet	01/05 - 28/10 [5] (180 d)	30/04, P, (20 cm) [5] 30/04, PH, (10 cm) [5] 30/04, BF, (20 cm) [5] 12/05, MH, (5 cm) [5] 04/06, MH, (5 cm) [5] 08/11, SS, (30 cm) [5]	16/06, CAN, (50) [5] 08/07, CAN, (50) [5] 28/07, CAN, (50) [5]		13/05, 2.5 [5] 03/06, 2.5 [5] 04/07, 2.5 [5]

Table 3.3. Farm management data used for DNDC baseline, calibration and validation simulations (*cont.*)

Field, year	**Crop**	**Growth period** ^{1}	**Tillage**	**Fertiliser**	**Ground cover**	**Irrigation**
		dd/mm, (days)	**dd/mm, method** ^{2}**, (depth)**	**dd/mm, type, (N applied, kg N ha**$^{-1}$**)**	**dd/mm, (days), type** ^{3}	**dd/mm, (rate, cm)**
S1, 2011	Lettuce (Romaine)	12/04 - 06/06 (55 d)	10/04, P, (20 cm) 12/04, PH, (10 cm)	12/04, NPK, (59.5)	12/04 - 12/05 ^{5} (30 d), HF	16/04, (2.5) 24/05, (2.5)
	Lettuce (Romaine)	18/07 - 05/09 (49 d)	16/07, P, (20 cm) 18/07, PH, (10 cm)	18/07, NPK, (22.3) 16/08, NPK, (29.8)		22/07, (2.5) 29/07, (2.5)
S1, 2012	Celery	27/04 - 16/08 (111 d)	31/03, P, (20 cm) 27/04, P, (20 cm) 27/04, PH, (10 cm)	27/04, NPK, (59.5)		28/04, (2.5) ^{5} 14/05, (2.5) 02/08, (2.5) ^{5}
S2, 2011	Lettuce (Iceberg)	13/04 - 05/06 (53 d)	11/04, P, (20 cm) 13/04, PH, (10 cm) 08/05, MH, (5 cm)	13/04, NPK, (59.5)	13/04 - 13/05 ^{5} (30 d), HF	17/04, (2.5) 18/05, (2.5)
	Radish	12/07 - 03/08 (22 d)	12/07, P, (20 cm) 12/07, PH, (10 cm)		12/07 - 03/08 (22 d), IM	12/07, (2.5)
S2, 2012	Lettuce (Iceberg)	23/03 - 30/05 ^{5} (68 d)	21/03, P, (20 cm) ^{5} 23/03, PH, (10 cm) ^{5}	23/03, NPK, (59.5)	23/03 - 23/04 ^{5} (31 d), HF	24/03, (2.5) ^{5} 22/04, (2.5) ^{5} 25/05, (2.5) ^{5} 27/07, (2.5) ^{5}
	Lettuce (Iceberg)	26/07 - 14/09 ^{5} (50 d)	25/07, P, (20 cm) ^{5} 26/07, PH, (10 cm) ^{5}			
S3, 2011	Celery	11/04 - 06/07 (86 d)	07/04, P, (20 cm) 11/04, PH, (10 cm) 08/05, MH, (5 cm)	11/04, NPK, (59.5)	11/04 - 11/05 ^{5} (30 d), HF	21/04, (2.5) 05/05, (2.5) 09/05, (2.5) 26/06, (2.5)
S3, 2012	Potato	18/04 - 25/09 (160 d)	12/04, SS, (30 cm) 13/04, P, (20 cm) 18/04, BF, (30 cm)	18/04, NPK, (74.4)		18/06, (2.0)

[1] Growth period is calculated from date of planting or drilling, to date of harvest. [2] BF = bed-forming; D = disked; DP = deep ploughing; MH = mechanical hoeing; P = ploughing; PH = power harrow; SS = sub-soiling. [3] HF = horticultural fleece; IM = insect mesh. [4] Injected, to a depth of 5 cm. [5] Author's estimate (farm records not available).

3.3. Results

3.3.1. Baseline model run

Overall, DNDC performance was poor when baseline input parameters were used. Running DNDC using single-year baseline inputs resulted in considerable underestimates of annual emissions of CO_2 and N_2O from both M1 and S3 fields, and an underestimate of CH_4 for field M1 (Table 3.4.). For field S3, CH_4 was overestimated. Simulated S3 CH_4 emission was the only value that fell within one S.E. of mean annual emission. At both sites, harvested crop biomass was underestimated.

Visual inspection of seasonal patterns and magnitudes of episodic emissions and soil variables suggested a poor match between most simulated output and measured values, with the exception of patterns in water table depth, and soil temperature at 5 cm depth (Figure 3.2.). This was partially verified when fit statistics were applied to each data set. Total model error in predicting outputs was within a similar range to the total measured data error, for all variables tested (using *LOFIT*, all values of *F* were lower than the critical $F_{0.05}$ value). However, *RMSE* revealed significant error between individual measured and modelled M1 and S3 values (i.e. $RMSE > RMSE_{95\%}$), for all variables, with the exception of N_2O and M1 CH_4 simulation. No model bias was observed in predicting N_2O or CH_4 emission, or soil WFPS ($E < E_{0.05}$), or M1 water table depth ($M < t_{0.05}$). Bias was evident when simulating CO_2 emission, S3 water table depth, and the remaining soils outputs, in both fields. Bias tended towards underestimates for most episodic values, apart from soil temperature and WFPS (Figure 3.2.). Significant correlations between simulated and modelled data were only found with regard to soil temperature ($r = 0.89$ and 0.88, for M1 and S3 respectively), water table depth ($r = 0.79$ and 0.99, respectively), and M1 soil NO_3^- ($r = 0.69$).

In addition to the variables presented in Figure 3.2., simulated soil pH was also monitored. It was found that changing input parameters through baseline, pre-simulation and sensitivity analysis exercises, had no effect on modelled pH at any soil depth – soil pH remained at the mean input value given in Table 3.1. For soil pH, there was no significant overall model error (*LOFIT* $F < F_{0.05}$), no model bias ($E < E_{95\%}$), and no model error as indicated by *RMSE* for field M1. For field S3, significant error was found between some modelled and measured pH values ($RMSE = 0.04$; $RMSE_{95\%} = 0.03$). A significant association between observed and simulated values was not evident.

Table 3.4. Performance of DNDC for predicting cumulative annual soil respiration, fluxes of N_2O and CH_4, and total annual harvested crop biomass at maturity; 10th June 2011 to 9th June 2012: cropped soils on fields M1, M2 and M3 (c. 35% SOM content) and S1, S2 and S3 (c. 70% SOM content).

Field	**Data set**	**Annual GHG emission**			**Harvested crop biomass (kg C ha^{-1}) ^{1}**
		CO_2 (t C ha^{-1} yr^{-1})	**N_2O (kg N ha^{-1} yr^{-1})**	**CH_4 (kg C ha^{-1} yr^{-1})**	
M1	Measured	10.28 ± 0.49	18.56 ± 3.35	0.43 ± 0.78	3500
	Baseline, year 1	0.52 *(-94.97)*	1.32 *(-92.87)*	**0.00**	2384 *(-31.88)*
	Baseline, year 26	0.60 *(-94.14)*	2.18 *(-88.25)*	**0.00**	1540 *(-56.00)*
	Calibration, year 25+26	9.29 *(-9.54)*	7.88 *(-57.54)*	**0.00**	7248 *(107.10)*
M2	Measured	9.81 ± 0.73	11.59 ± 1.81	2.13 ± 1.56	3500
	Validation, year 25+26	10.76 *(9.70)*	13.87 *(19.68)*	0.00	7079 *(102.26)*
M3	Measured	5.18 ± 0.53	9.10 ± 1.01	0.05 ± 0.04	2930
	Validation, year 25+26	9.04 *(74.48)*	24.82 *(172.81)*	0.00	6253 *(113.42)*
S1	Measured	6.15 ± 0.74	21.25 ± 5.13	3.49 ± 3.51	1900
	Validation, year 25+26	8.96 *(45.60)*	**16.88** *(-20.56)*	0.00	1898 *(-0.09)*
S2	Measured	6.45 ± 0.48	24.30 ± 11.35	1.52 ± 0.46	2350
	Validation, year 25+26	4.90 *(-24.04)*	2.38 *(-90.22)*	0.00	2397 *(2.02)*
S3	Measured	10.53 ± 0.43	3.11 ± 0.60	-3.82 ± 1.57	5900
	Baseline, year 1	0.42 *(-95.98)*	0.64 *(-62.21)*	0.00	3370 *(-42.89)*
	Baseline, year 26	0.64 *(-93.93)*	1.17 *(-62.21)*	0.00	3111 *(-47.28)*
	Calibration, year 25+26	**10.40** *(-1.24)*	**3.34** *(7.48)*	0.00	6754 *(14.48)*

Values in normal text indicate measured or modelled values. Values in italics indicate relative deviation (RD) of simulated values from measured mean values. Emboldened text indicates modelled values falling within one standard error of the mean measured value.

[1] For the whole 2011-2012 period; in the case of crops harvested after 9th June 2012, this extends beyond the measurement period.

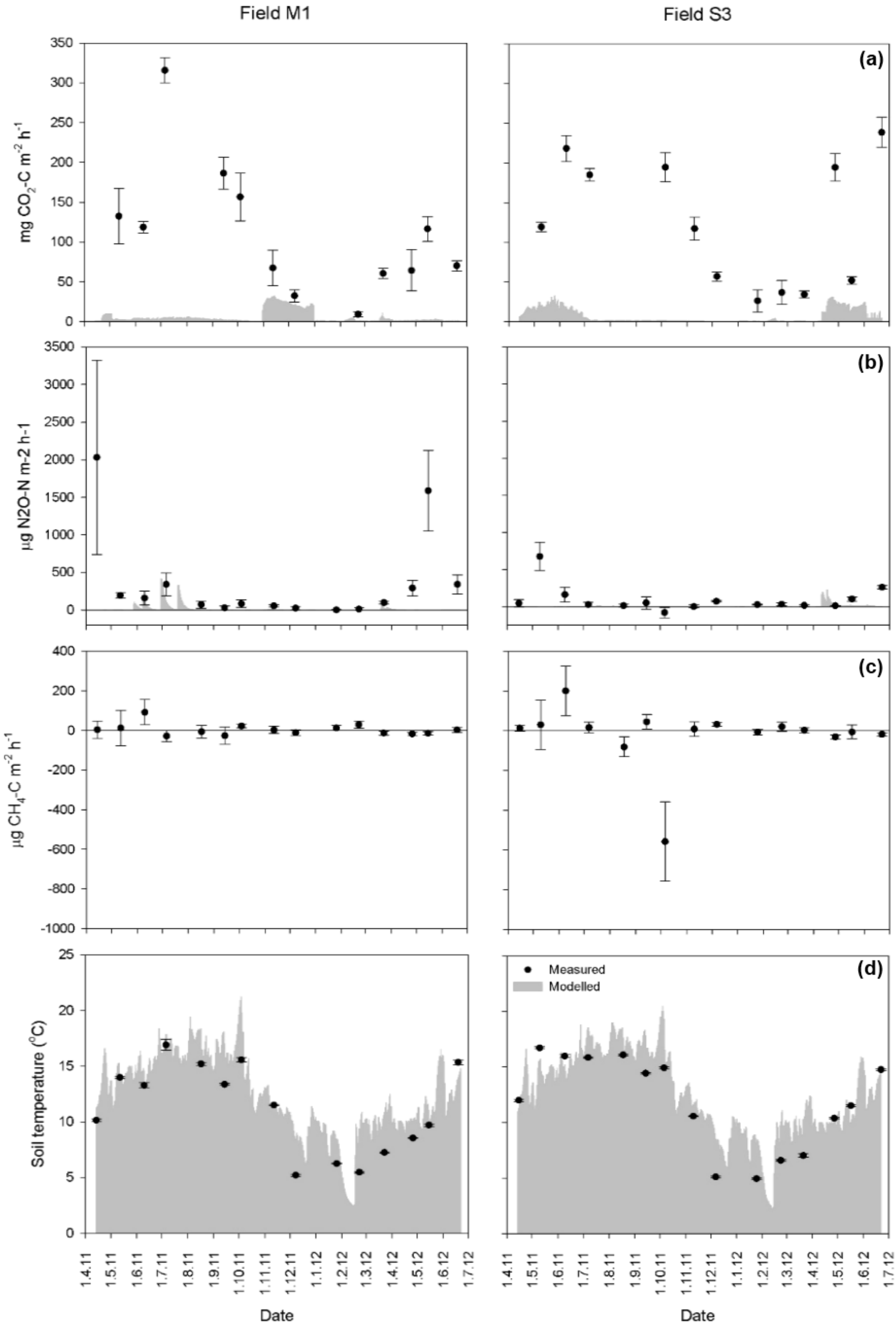

Figure. 3.2. Baseline input performance of the DNDC model at predicting episodic soil respiration (a); fluxes of N_2O (b) and CH_4 (c); and soil temperature (d); 14th April 2011 to 22nd June 2012: cropped soils – on fields M1 (c. 35% SOM content) and S3 (c. 70% SOM content). Error bars represent ± 1 S.E. of the mean value.

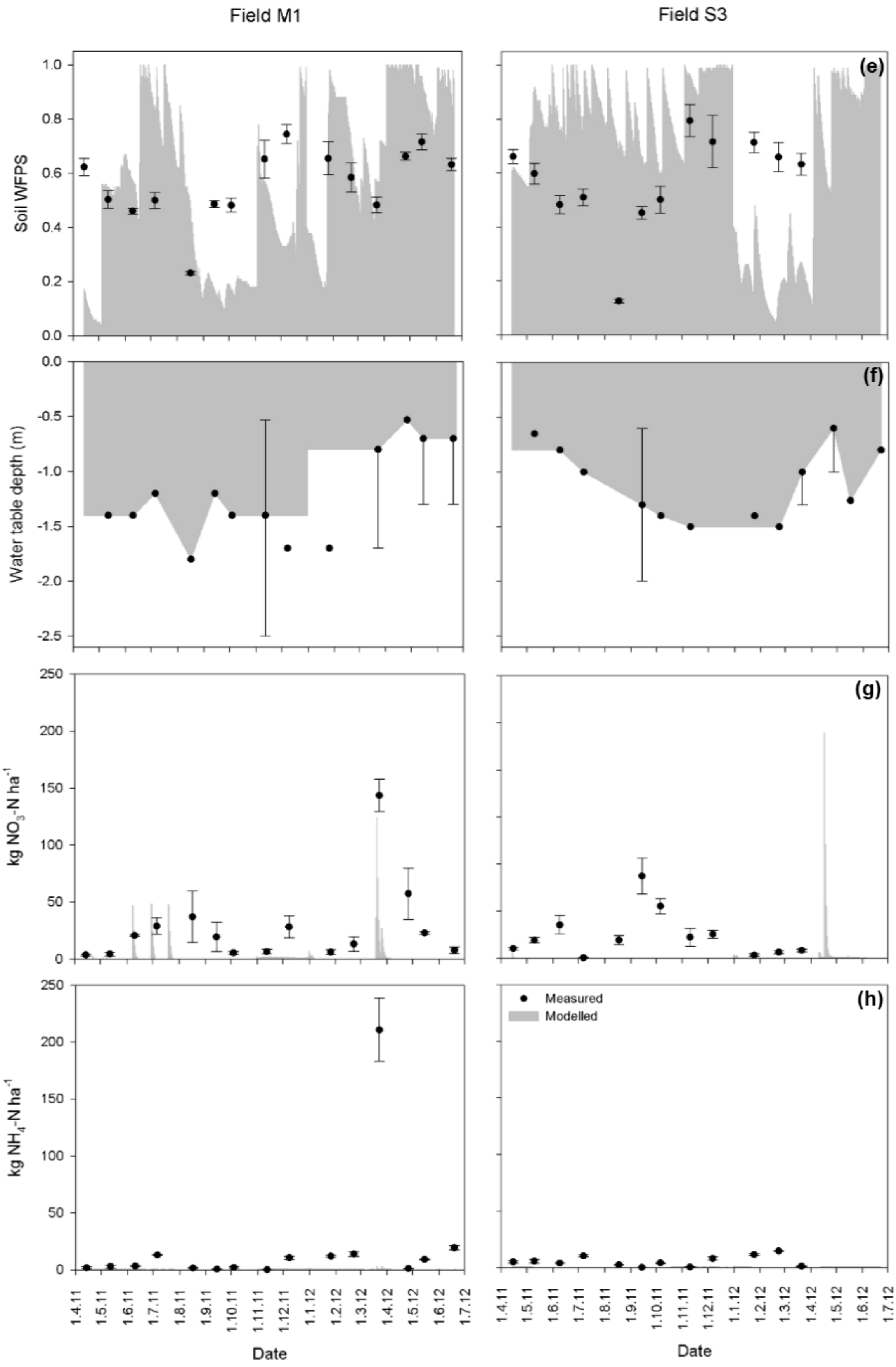

Figure. 3.2. ***(continued)*** Baseline input performance of the DNDC model at predicting episodic soil WFPS (e); water table depth (f); and soil NO_3^- (g) and NH_4^+ (h); 14th April 2011 to 22nd June 2012: cropped soils – on fields M1 (c. 35% SOM content) and S3 (c. 70% SOM content). Error bars represent ± 1 S.E. of the mean value.

3.3.2. Optimising model stability using pre-simulation model runs

The effect of pre-simulation was tested on the mean and range of daily output values over the period 10th June 2011 to 9th June 2012, for soil respiration, N_2O and CH_4 emission, soil NO_3^- and exchangeable NH_4^+, WFPS, temperature and pH, and water table depth, SOC, *d*SOC, and crop harvested biomass and residual aerial and root biomass (Appendix C). Running the DNDC model for 25 years prior to accepting the simulation year results was sufficient to eliminate the possibility of accepting stochastic output data for the majority of field-year combinations, for all of the tested variables. Varying the period of pre-simulation had no effect on daily outputs of CH_4 emission, soil pH, and water table depth, so these are excluded from Appendix C, and from further discussion here. On the basis of these results, year 26 output data was accepted for use in subsequent calibration, sensitivity analysis and validation exercises.

Using year 26 of the baseline data resulted in a slight improvement in the prediction of annual soil respiration from both M1 and S3 fields (*RD* declined by 1% and 2% respectively), and a greater improvement in the prediction of annual N_2O emission (*RD* declined by 5% and 17% respectively), but no change in annual predicted CH_4 emission (Table 3.4.). Conversely, prediction of annual harvested crop biomass was less accurate than when year 1 data were used, and was reduced by 24% and 4% for fields M1 and S3 respectively.

The magnitude and distribution of year 26 data daily modelled output values were predominantly very similar to those using year 1 data, both visually and according to fit statistics (data not shown). Visually, an overall increase in soil WFPS was observed at both sites, and an increase in M1 soil NO_3^- during 2011. Significant changes were observed in four variables. Model performance significantly decreased when predicting S3 N_2O emission (*LOFIT F* increased by a factor of 28, *RMSE* by a factor of 8, and *E* by a factor of 3 with a corresponding change to over-prediction of N_2O emission). For M1 and S3 WFPS, the correlation coefficient became significant (r = 0.68 and 0.85), but model bias increased, resulting in a significant overestimate of WFPS. A significant correlation between M1 observed and simulated NH_4^+ values was also found (r = 0.53). Model bias decreased in M1 soil NO_3^- prediction (*E* decreased by just under half, and changed to a tendency for over-prediction), but the previous significant correlation between M1 soil NO_3^- observed and simulated data points was lost (r = -0.01).

3.3.3. Sensitivity analysis: effect on modelled annual GHG fluxes

The effects of varying climatic, soil and crop management input parameters on predicted soil respiration CO_2, and N_2O, are shown in Table 3.5.; predicted CH_4 was only affected by changes in water table depth, so is only discussed briefly here. Adjusting the input values of some variables had no noticeable effect on simulated GHG emissions, within the range tested (drainage efficiency, initial NO_3^- concentration at soil surface, plant vascularity, and flooded soil water leakage rate). Adjusting several others had a negligible effect on emissions (depth of water retention layer, initial soil NH_4^+ concentration, microbial activity index, and kg N in flood water). These variables are excluded from Table 3.5.

For over half of the remaining variables tested in the sensitivity analysis, GHG emissions showed considerable sensitivity to changes in input values. Here, sensitivity is defined as yielding a change in emissions ($\Delta_{CO2\%}$ or $\Delta_{N2O\%}$) of greater than ± 50% of the baseline emission value. For continuous input variables, relative sensitivity (the % change in emissions for every 1% change in the input variable, $S_{\%}$) was also calculated. In this case, sensitivity was defined as yielding greater than a ± 0.5% change compared to the baseline emission, for every 1% change in the variable value.

Within the range tested, the sensitivity of CO_2 and N_2O emission to altered values of inputs differed between fields M1 and S3. Relating to field M1, CO_2 emission was most sensitive to soil hydraulic conductivity, soil humads and resistant litter percentage, humus decomposition rate, and maximum biomass production (Table 3.5.). Field S3 CO_2 emission was also highly sensitive to humus decomposition rate, as well as litter decomposition rate, type of post-harvest tillage, application of precision fertilisation, and water table depth. Field M1 N_2O emission was most sensitive to depth of uniform SOC content, rainwater collection index (i.e. lateral influx index of water entering the soil profile), soil humads fraction, fertiliser type, and water table depth. Again, S3 N_2O emission was most strongly influenced by different variables – air temperature, litter and humus decomposition rates, use of precision fertilisation, and water table depth.

When considering sensitivity in terms of relative sensitivity ($S_{\%}$), the greatest effect on M1 CO_2 emission was observed in changing mean daily air temperature, mean daily rainfall, depth of soil with uniform SOC content, soil hydraulic conductivity, and water table depth. Air temperature, rainfall, and water table depth were also strong influencing factors for S3, as well as litter decomposition rate, and crop residue fraction retained in-field after harvest. Similarly to $S_{\%(CO2)}$, $S_{\%(N2O)}$ relating to both fields was strongly influenced by air temperature, rainfall, and water table depth, as well as soil pH. Additionally, SOC humads fraction was important for M1, and soil porosity for field S3 $S_{\%}$ N_2O emission.

Methane emission was only sensitive to variation in simulated water table depth, and only if a water table shallower than the default measured depth was imposed (data not shown). Raising the water table depth by up to 50 cm resulted in exponential CH_4 emission increases from 0.17 to 71.9 kg CH_4-C ha^{-1} yr^{-1} for field M1, and from zero to 16.52 kg CH_4-C ha^{-1} yr^{-1} for field S3. It was not possible to calculate percentage change in emission compared to the baseline, as baseline simulated annual CH_4 emission was zero.

Table 3.5. Summary results: sensitivity of soil respiration CO_2, and N_2O emission, to relevant DNDC input variables.

		M1					**S3**				
Input	**Units**	**Values**	**$\Delta_{CO2\%}$**	**$\Delta_{N2O\%}$**	**$S_{\%}$ CO_2**	**$S_{\%}$ N_2O**	**Values**	**$\Delta_{CO2\%}$**	**$\Delta_{N2O\%}$**	**$S_{\%}$ CO_2**	**$S_{\%}$ N_2O**
Air t	°C	-6 to +4;	-0.4	***-68.5***	**-4.2**	**-3.3**	-4 to +4;	-40.8	***-72.0***	**1.1**	**1.9**
		EA t ^{1}, EA ^{2}	***82.5***	***56.7***	0.4	**36.5**	EA t ^{1}, EA ^{2}	**116.6**	**1957.5**	**557.2**	**14751.9**
Precipitation	cm	-0.8 to +1;	***-68.5***	***-80.6***	**-1.9**	0.1	-0.8 to +1;	***-51.4***	***-83.9***	**-3.3**	**-88.51**
		EA rain ^{3}	***82.2***	26.0	**1.2**	**2.2**	EA rain ^{3}	40.8	-33.4	-0.1	**1.5**
BD	g cm^{-3}	0.28	-23.8	-2.3	***0.8***	-0.1	0.23 to	-16.0	-40.1	0.4	**1.2**
		to 0.50	31.5	6.4	**1.1**	0.2	to 0.57	31.9	***86.6***	***0.6***	**1.3**
pH		5.76	0.1	-11.9	-0.1	**-2.9**	5.71	-0.1	-17.3	-0.02	**-2.0**
		to 7.38	1.6	30.6	0.01	**-1.8**	to 7.23	0.2	8.1	-0.01	***-0.5***
Clay fraction	Fraction	0.001	***-51.6***	-11.1	-0.1	-0.01	0.001	-47.1	-9.7	-0.2	-0.1
		to 0.6	12.3	4.4	0.1	0.2	to 0.6	20.0	6.9	-0.1	-0.01
SOC 0-10 cm	Fraction	0.325	-5.8	-2.2	**1.5**	***0.6***	0.273	-9.7	-32.0	0.4	**1.4**
		to 0.344	4.8	1.9	**3.5**	***0.8***	to 0.432	11.9	36.6	***0.5***	**1.5**
Uniform SOC	m	0.1	***-61.2***	-21.8	**-1.8**	**-1.5**	0.1	-12.4	-29.1	-0.2	**-1.0**
		to 1	***91.4***	***76.4***	-0.2	-0.1	to 1	4.4	49.4	-0.03	-0.1
ΔSOC / depth		0.55 to 1	***-61.2***	-21.8	***-0.9***	***-0.7***	0.5 to 1	-12.4	-29.1	-0.1	-0.3
			-8.3	-6.0	***-0.6***	-0.3		-2.1	-5.5		
Hyd. cond.	m h^{-1}	0.008	0.0	-6.6	**-2.5**	***-0.8***	0.008	0.0	0.0	0.0	0.0
		to 0.03	**116.8**	38.9	0.0	*0.2*	to 0.03				
Porosity	Fraction	0.701	-9.8	-13.5	***-0.7***	***-0.7***	0.701	-3.0	25.4	-0.2	**1.8**
		to 0.870	-9.1	9.2	-0.4	***-0.6***	to 0.870	2.2	***59.8***	-0.1	**2.5**
FC (WFPS)	Fraction	0.3	***-51.5***	-20.4	**-1.6**	-0.2	0.3	-8.9	-0.5	-0.2	-0.02
		to 0.9	44.3	3.3	**1.1**	***0.5***	to 0.9	-0.1	0.8	0.1	0.0
Rain collection index		0	***-56.1***	0.0	***-0.5***	0.0	0	-41.1	***0.0***	-0.4	0.0
		to 5	0.0	***69.5***	0.0	0.2	to 5	0.0	***107.7***	0.0	0.3
R. litter, fraction	Fraction	0.001	-7.1	-10.7	0.1	0.1	0.001	-13.0	-3.6	-0.03	-0.02
		to 0.05	**128.6**	45.1	0.3		to 0.05	2.2	1.5	-0.02	-0.01
Humads,fraction	Fraction	0.17	***-55.5***	-8.7	-0.3	-0.4	0.17	16.1	12.0	0.2	0.2
		to 0.5	**146.7**	**264.2**	**1.4**	**1.4**	to 0.5	44.6	***63.1***		0.3

Table 3.5. Summary results: sensitivity of soil respiration CO_2, and N_2O emission, to relevant DNDC input variables (*continued*).

		M1					**S3**				
Input	**Units**	**Values**	**Δ_{CO2}**	**Δ_{N2O}**	**$S_{\%}$ CO_2**	**$S_{\%}$ N_2O**	**Values**	**Δ_{CO2}**	**Δ_{N2O}**	**$S_{\%}$ CO_2**	**$S_{\%}$ N_2O**
Litter DR		1	-7.3	-22.0	-0.01	-0.08	1	***56.7***	-34.6	0.1	-0.03
		to 50	-0.8	-7.6	0.0	0.0	to 50	**269.6**	**567.1**	***0.6***	0.1
Humus DR		1	18.9	1.1	0.04	0.01	1	8.3	5.1	0.03	0.03
		to 50	**339.6**	28.7	0.3	0.02	to 50	**168.5**	**164.0**	0.1	0.1
Residue fraction	Fraction	0	-17.0	-17.2	0.2	0.2	0	**-629.9**	-47.8	**-1.1**	0.2
		to 1	-4.5	-4.5			to 1	***54.0***	0.0	***0.6***	***0.5***
Max. grain C	kg C ha^{-1}	3500	2.1	-3.1	0.02	0.0	5900	0.6	2.7	0.01	0.01
		to 35000	**136.4**	14.8	0.2	0.1	to 59000	12.7	10.2		0.03
Biomass fraction	Fraction	default; modified	35.6	23.0	nc	nc	default, modified	***51.2***	-7.5	nc	nc
PH till type		0 to 30;	-4.4	-2.5	nc	nc	0 to 30;	***-56.9***	-11.1	nc	nc
		LB [4]; CT [5]	**104.1**	6.8			LB [4]; CT [5]	**430.9**	40.6		
Fertiliser type		urea, NH_3, NO_3^-, NH_4^+,	-0.3	***-88.2***	nc	nc	urea, NH_3, NO_3^-, NH_4^+,	-5.1	***-73.0***	nc	nc
		NH_4^++NO_3^-, SO_4, PO_5	0.8	***84.6***			NH_4^++NO_3^-, SO_4, PO_5	0.2	***70.7***		
SR fertiliser		1	0.1	***-53.9***	0.0	-0.01	1	44.8	26.1	0.01	0.0
		to 60	0.3	-41.0			to 60	***54.3***	27.0	0.02	0.01
Application date	days	-1 d	-0.04	-12.9	nc	nc	-1 d	-3.2	-28.6	nc	nc
		to + 1 mo	3.7	40.6			to + 1 mo	***54.7***	15.9		
Auto-fertilisation		no, yes	0.60	***-65.7***	nc	nc	no, yes	***53.7***	-33.8	nc	nc
Precision fert.		no, yes	2.5	***-67.1***	nc	nc	no, yes	**1952.2**	**1361.4**	nc	nc
Irrigation	cm	none; -1	-7.4	-21.2	nc	nc	none; -1	-14.3	-2.7	nc	nc
		to +1	***53.6***	6.4			to +1	**124.3**	0.5		
Water table	cm	- 50 to +50;	14.6	0.0	**-1.9**	**-1.1**	- 50 to +50;	-13.3	-0.1	-0.01	0.0
		crop=50 [6]	**105.9**	**147.9**	**2.6**	**14.8**	crop=50 [6]	**396.8**	**329.3**	**9.8**	**8.2**

Values presented represent minimum and maximum percentage changes in annual emissions relative to baseline annual emissions. Figures in bold signify substantial positive or negative percentage changes: italicised bold signify a Δ_{CO2} value of 50 to 100% or a $S_{\%}$ CO_2 value of 0.5 to 1%, while emboldened plain text indicate a Δ_{CO2} value of > 100% or a $S_{\%}$ CO_2 value of > 1%. [1] East Anglia mean temperature. [2] East Anglia mean temperature and rainfall. [3] East Anglia mean rainfall. [4] Litter-burying till (depth = 20 cm). [5] Crop-terminating till (depth = 0 cm). [6] Water table maintained at 50 cm.

3.3.4. Model calibration

Reasonably good improvements in model performance were achieved by implementing a number of changes in model parameterisation. Early in the calibration exercise, it became evident that improved model performance could be achieved by combining 2011 and 2012 files into a single pre-simulation run, that is, the two years' climate and management data were alternated over a period of 26 years, and years 25 (corresponding to 2011) and 26 (corresponding to 2012) were respectively taken as the simulated years. Although this was slightly more complex than simulating the years separately, combined pre-simulation overcame the issue of the disparate SOC and WFPS distributions, observed when the two years were simulated separately.

Calibration for field M1 was accomplished by first optimising SOC distribution down the soil profile to approximate that found during soil depth core analysis (section 3.2.3.2.); this resulted in a modified SOC decrease rate of 0.86 (Table 3.1.). Despite not being specified in many farm records, some verbal reports of post-harvest tills were given by farm managers. Adding a simulated post-harvest litter-burying till one week after each crop, improved simulated magnitudes of CO_2 and N_2O. Further improvements in N_2O emission pattern and magnitude were provided by changing all fertiliser applications to ammonium-based fertilisers, and by assigning all irrigation events as additional fertilisation events (based on an average rate of 0.802 kg mm^{-1} irrigation applied, after ADAS, 2006 and Nix, 2014). The magnitude of CO_2 and N_2O emissions were simultaneously adjusted by increasing crop grain production by four times, changing all crop root fractions to 30% of total crop biomass, and increasing litter, humads, and humus decomposition rates to 5, 25 and 100 respectively (Tables 3.1., 3.2. and 3.3.). Further, water table depth was adjusted to an estimated shallowest in-field depth, based on a comparison of the mean difference (c. 60 cm) between ditch and in-field water table depth during spring 2012.

After parameterising DNDC using calibrated values, only CH_4 annual emission fell within 1 S.E. of the mean measured annual estimate, and simulated annual CO_2 emission was only slightly outside of the lower bound (Table 3.4.). Conversely, annual N_2O was substantially underestimated by 58%. Simulated harvested crop biomass was overestimated by 107%, comprising an overestimation in 2011 (176%) but an underestimation in 2012 (9%). Visually, there was very little difference between baseline year 26 data and calibrated data for CH_4 emission, or soil temperature, pH, or exchangeable NH_4^+ content (Figure 3.3.). Improvements can be seen in the magnitude and pattern of CO_2 emission and soil NO_3^- content, and in the magnitude of N_2O emissions. Calibration produced a substantial change in soil WFPS, but overall no particular improvement in the accuracy of GHG prediction. Fit statistics largely concurred with visual observations. The fit of most variables improved after calibration, when compared to year 26 baseline data. This resulted in a significant reduction of bias and increase in correlation for simulated CO_2 data (E = 27.85; r = 0.73), and a significant reduction in individual value errors and increase in correlation for simulated soil NO_3^- data ($RMSE$ = 1.15; r = 0.84). Only soil WFPS and exchangeable NH_4^+ showed a worse fit between modelled and measured data after calibration (r = 0.42 and 0.31 respectively). Calibration for field M1 did not affect the fit of CH_4 emission or soil pH.

Calibration to observed S3 data was achieved by changing the same variables as for M1. The only differences in values were in SOC decrease rate below the soil surface (0.8), and litter decomposition rate (20). Two further changes were made, in accordance with the site soil characteristics: clay content was assumed to be negligible, and was therefore reduced to 0.1% of soil content; and soil porosity was increased to 0.870, as described previously for the study site by Kechavarzi *et al*. (2010). Tables 3.1., 3.2. and 3.3. show the changes made to achieve calibration.

Annual S3 simulated estimates of CO_2 and N_2O emission fell within 1 S.E. of the observed mean estimate, while annual CH_4 emission was overestimated, but only by a small margin (Table 3.4.). Harvested crop biomass was also overestimated by only a small margin (14%), but this resulted from a combined underestimate for 2011, of 61%, and a smaller overestimate for 2012, of 38%. Visual comparison with baseline year 26 data provided very similar conclusions to the effect of calibration on M1 data, with the exception of the soil WFPS pattern, which was slightly improved, and soil NO_3^- content, which was greatly overestimated (Figure 3.3.). Most measures of fit improved after calibration, but only two variables showed significant changes in fit: model bias was reduced to non-significant levels, for N_2O (E = -52.46; still tending towards over-estimation), and for water table depth (M = -0.16; changing from a tendency for under-estimation to a tendency for over-estimation). Although there were no significant changes in the fit of episodic CO_2 values, correlation, $RMSE$, and bias all slightly improved (r = 0.57; $RMSE$ = 0.67; E = 42.67). However, overall model fit error in relation to measured error increased ($LOFIT\ F$ = 0.46). The calibration process had no effect the fit of S3 CH_4 emission or soil pH simulated data.

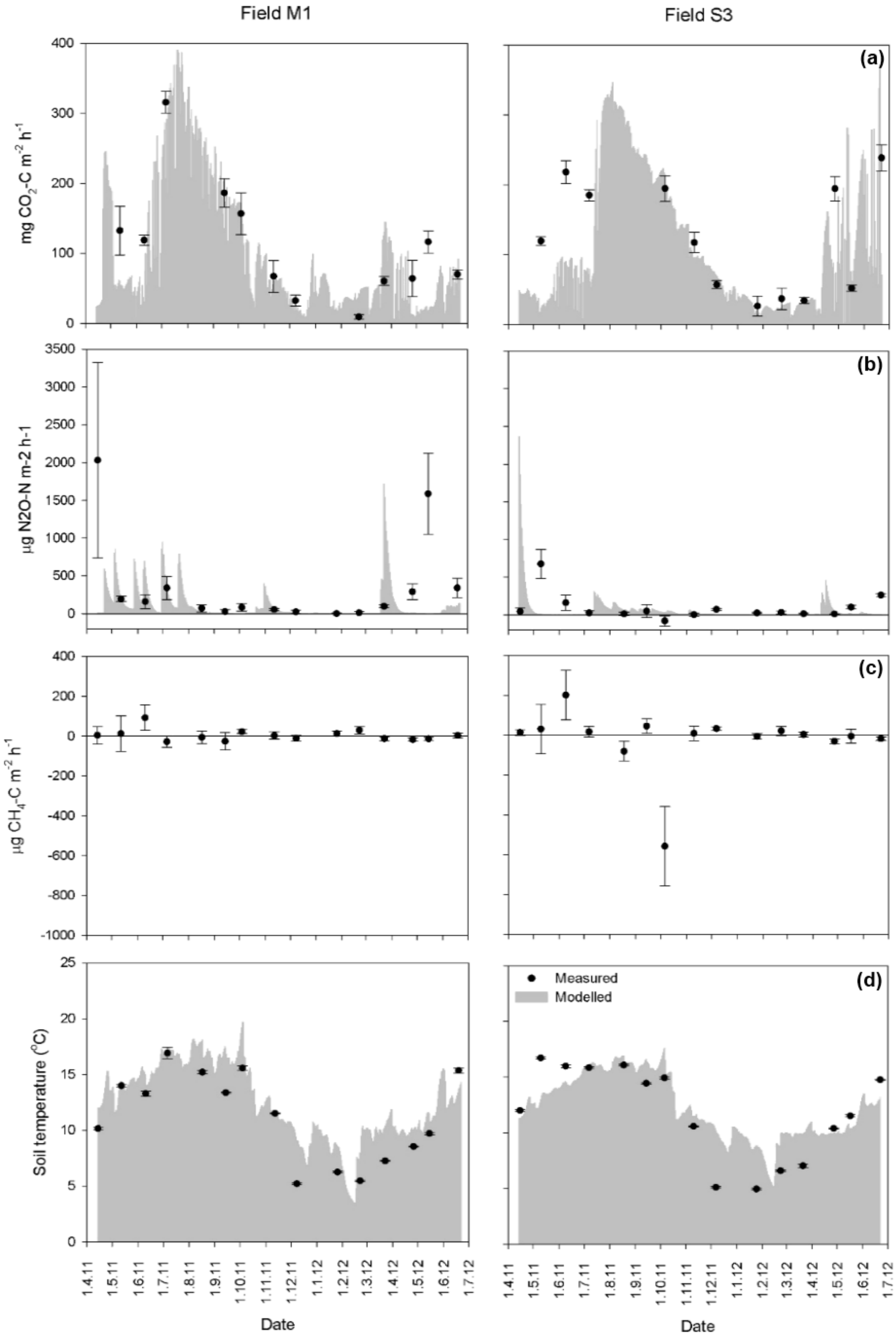

Figure. 3.3. Performance of the calibrated DNDC model at predicting episodic soil respiration (a); fluxes of N_2O (b) and CH_4 (c); and soil temperature (d); 14th April 2011 to 22nd June 2012: cropped soils – on fields M1 (c. 35% SOM content) and S3 (c. 70% SOM content). Error bars represent ± 1 S.E. of the mean value.

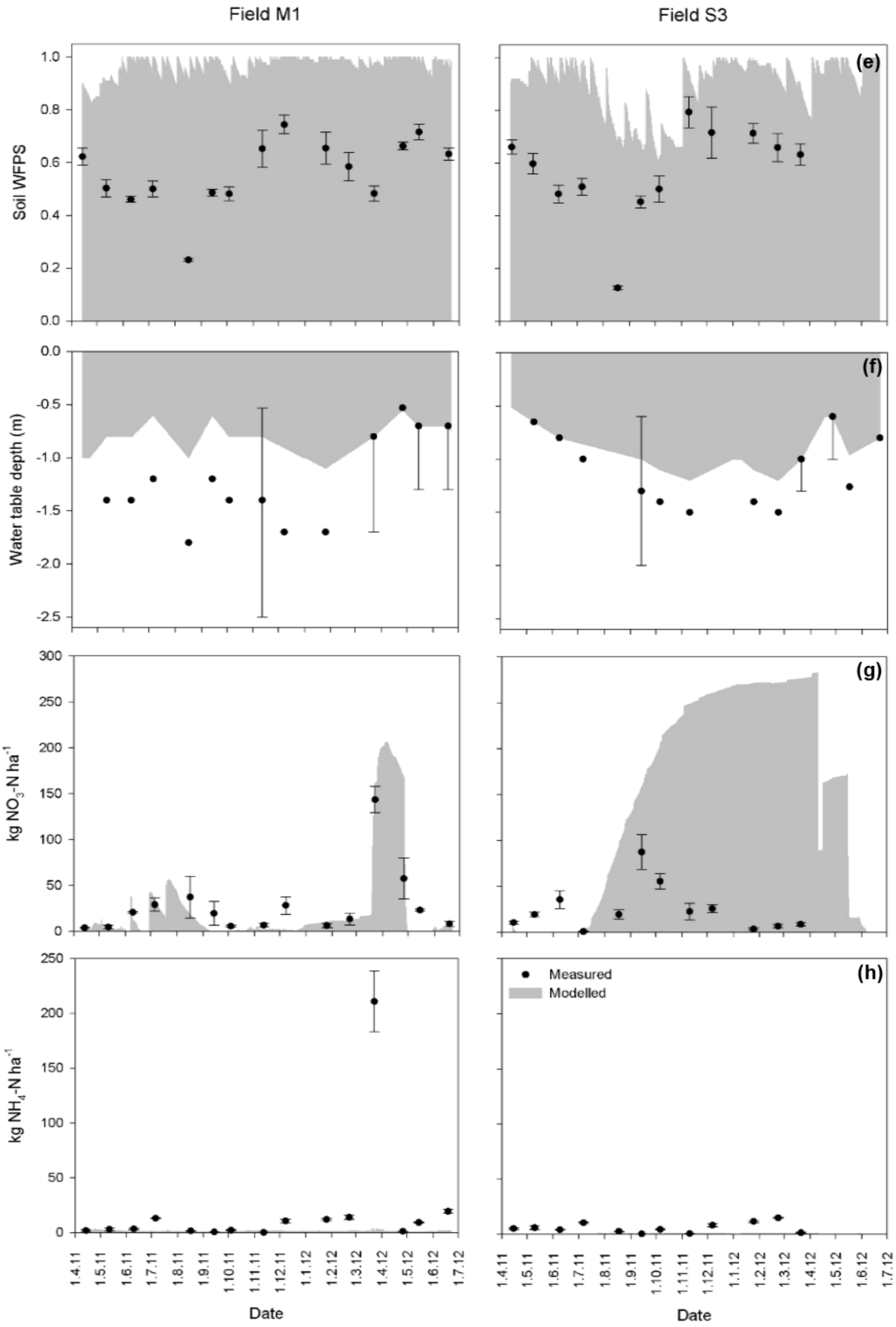

Figure. 3.3. ***(continued)*** Performance of the calibrated DNDC model at predicting episodic soil WFPS (e); water table depth (f); and soil NO_3^- (g) and NH_4^+ (h); 14th April 2011 to 22nd June 2012: cropped soils – on fields M1 (c. 35% SOM content) and S3 (c. 70% SOM content). Error bars represent ± 1 S.E. of the mean value.

3.3.5. Model validation

Only partial model validation was possible by applying M1 calibration principles to fields M2 and M3. Some further adjustment of seasonal water table dynamics was required in an attempt to optimise the magnitude and distribution of emissions over time (Figure 3.4.; Appendix D). Additionally, field M3 soil surface SOC was adjusted, owing to its higher soil clay content when compared to M1 and M2 (Table 3.1.). Its shallower impermeable sub-surface clay layer (c. 0.9 m depth), was also incorporated into the M3 input data.

After model validation was attempted, none of the simulated annual emissions for either field fell within 1 S.E. of mean observed estimates, although modelled M2 GHGs were very close to being within these bounds (Table 3.4.). Simulated M3 annual CO_2 and N_2O emissions were large over-estimates, and CH_4 was a negligible under-estimate. For both M2 and M3, harvested crop biomass was overestimated, by 102% and 113% respectively.

The smallest visual differences between observed and simulated data were for CH_4 emission, and soil temperature, pH, and exchangeable NH_4^+ content (Appendix D). Soil respiration also appeared to be reasonably well modelled for field M2, and N_2O emission for M3 (Figure 3.4.). Significant overall modelling error, compared to measurement error, was only observed for M3 CO_2 prediction ($LOFIT\ F = 2.65$). In contrast, most variables had some poorly predicted individual episodic values ($RMSE > RMSE_{95\%}$). Only simulated N_2O and soil pH data sets yielded a non-significant *RMSE*. Significant prediction bias was evident in soil temperature and WFPS for both fields ($E > E_{0.05}$), and M2 soil NO_3^- content. Significant correlations between observed and simulated data were only present for soil temperature (M2 $r = 0.89$; M3 $r = 0.93$), and water table depth at M3 ($r = 0.80$).

Some success was evident when validating the simulated data for site S70. Again, S1 and S2 water table dynamics were adjusted within plausible bounds, but this made negligible difference to emissions estimates. Simulated S1 annual emissions were closer than S2 data to measured annual emissions estimates, with both S1 N_2O and CH_4 simulated emissions falling within 1 S.E. of the mean estimated observed values (Table 3.4.). Field S1 CO_2 was over-estimated, while all S2 GHGs were under-estimated by DNDC. Estimates of total harvested crop biomass were very good for both fields, but in both cases were the product of over-estimates in 2011 (of 88% and 14% respectively) and under-estimates in 2012 (of 31% and 8% respectively).

For both S1 and S2, only CH_4 emission, and soil temperature and pH, appeared to show relatively small differences between observed and modelled data points (Appendix D). Soil respiration and N_2O appeared to be overestimated for S1 throughout most of the annual cycle, and underestimated for S2 (Figure 3.4.). Overall model error was non-significant for all variables, for both fields ($LOFIT\ F < F_{95\%}$). In contrast, a significant degree of error was seen in modelling individual data points, for all variables except CH_4 emission, and S2 N_2O emission ($RMSE = 2.58$, 2.06 and 1.39 respectively). Prediction bias was evident in simulated CO_2 emission, soil temperature, and soil NO_3^- and exchangeable NH_4^+ content ($E > E_{0.05\%}$). Bias tended towards over-prediction of soil temperature and NO_3^- content, and S1 CO_2 emission, and under-prediction of soil exchangeable NH_4^+ content, and S2 CO_2 emission. Significant linear correlations were found between observed and simulated variables for both fields, in relation to CO_2 emission ($r = 0.63$ and 0.65), soil temperature ($r = 0.86$ and 0.88), and water table depth ($r = 0.86$ and 0.87).

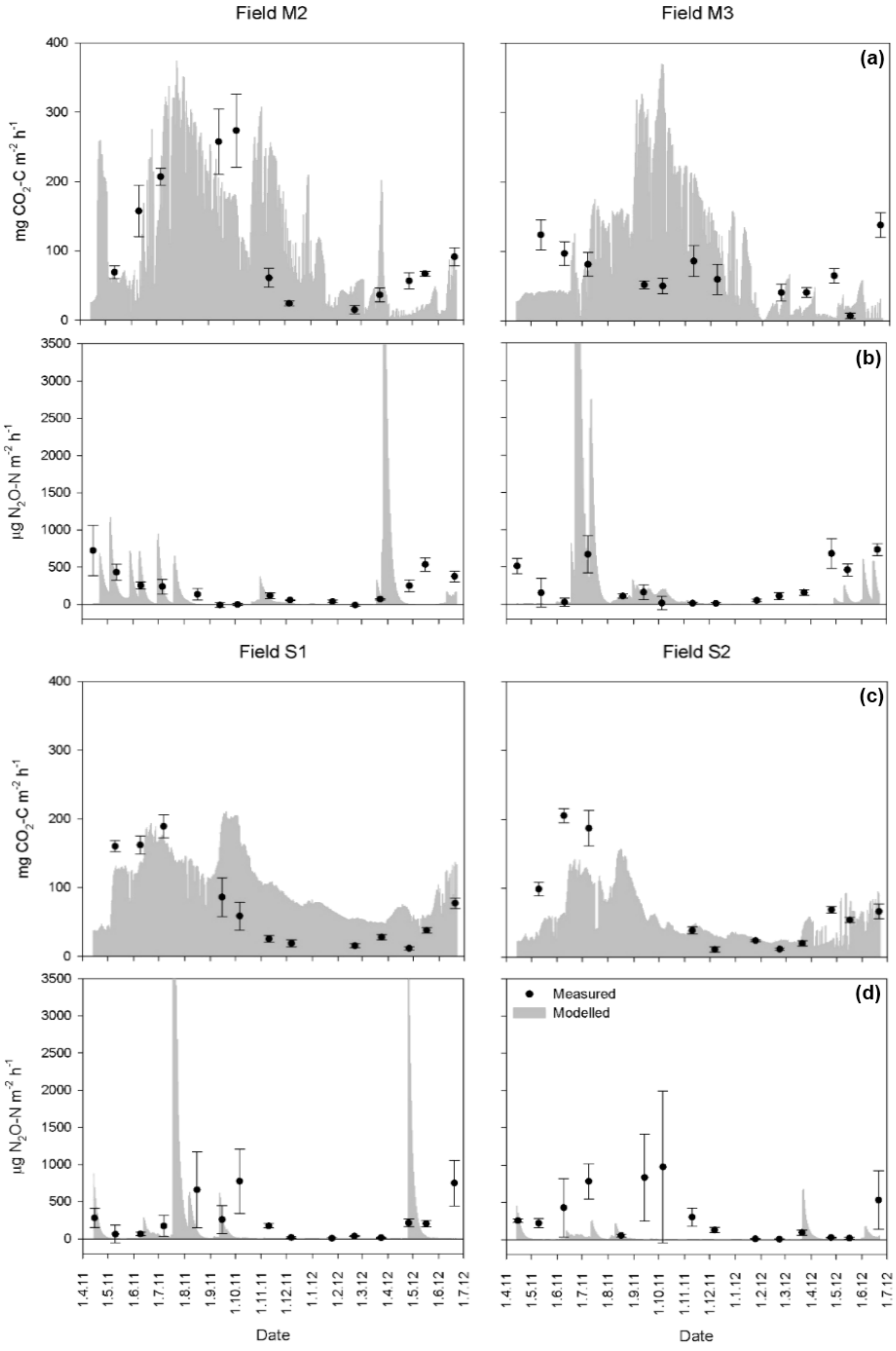

Figure. 3.4. Performance of the DNDC model at predicting episodic soil respiration (a) and (c); and fluxes of N_2O (b) and (d); 14th April 2011 to 22nd June 2012: cropped soils – fields M2 and M3 (c. 35% SOM content) and S1 and S2 (c. 70% SOM content). Error bars represent ± 1 S.E. of the mean value.

3.4. Discussion

The overall aim of this study was to assess the ability of the DNDC model to simulate emissions of CO_2, N_2O and CH_4 from two horticultural peat soils with differing SOM contents. Model performance was evaluated in terms of both annual and episodic emission predictions, and prediction of episodic values of soil physical variables. A sensitivity analysis successfully identified a number of variables strongly influencing predicted emissions, and was used to inform the DNDC calibration and validation exercises.

3.4.1. Use of the DNDC model with default parameterisation

Using the default DNDC parameters relating to soil physical properties, SOC partitioning, and crop physiological characteristics, generated simulated results with a poor fit to both annual emissions data and episodic emissions and soil physical variables. Other authors have noted that initial model conditions may be unstable, and consequently DNDC may require a pre-simulation period to avoid yielding unreliable results (e.g. Peltoniemi *et al*., 2007; Tonitto *et al*., 2007). Standardised guidance from the literature is lacking, regarding optimisation of the period and conditions for pre-simulation under different circumstances. Reported conditions vary from one year of pre-simulation, assuming land use and management identical to reported results (David *et al*., 2009; Tonitto *et al*., 2007), to 1400 years of pre-simulation, assuming native vegetation cover prior to agricultural use (Qin *et al*., 2013). Studies of arable agro-ecosystems often report the use of 10 to 20 years of pre-simulated data (e.g. Fumoto *et al*., 2008; Levy *et al*., 2007; Smith *et al*., 2012; 2013). None of these studies, however, presented statistical justification for their chosen pre-simulation period or conditions. A period of c. 25 years of pre-simulation was sufficient in this study for eliminating initial stochastic data variation, but it should be recognised that this choice was made subjectively. It became clear during the calibration and validation processes that although pre-simulation had eliminated initial chaotic data patterns, a considerable model calibration effort was required in addition to pre-simulation. As such, using DNDC with default parameterisation is inadvisable when simulating GHG emissions from horticultural peat soils, as it can result in significant under-estimates of fluxes.

3.4.2. Sensitivity of simulated GHG emissions to changes in model input values

Sensitivity to different climatic, soil, crop and management factors differed between the two chosen baseline fields, M1 and S3 (Section 3.3.3.). Nevertheless, adjusting certain types of input variables influenced emissions at both sites. Under baseline conditions, simulated soil respiration and N_2O emission were particularly responsive to changes in mean daily air temperature and mean annual precipitation. Other important variables related predominantly to SOC dynamics, vegetation characteristics, tillage and fertilisation applications, and water table depth. Soil water table depth was the only variable that influenced CH_4 emission, within the range of values tested. A number of other studies have observed sensitivity to the factors detailed here, including those conducted on peatland sites (Li *et al*., 1992a, b, 1994, 2010; Maljanen *et al*., 2012; Webster *et al*., 2013).

The degree of response of emissions to adjustments in model input variables is context dependent. This became particularly evident during the calibration and validation exercises, where enhanced or suppressed responses to one variable were seen when other variables had been changed from their baseline values. For example, under conditions of increased SOC decomposition rates, raising water table depth to within 50 cm from the soil surface for even a short period of time resulted in a much greater surge of CO_2 emissions, when compared to making the same change in depth under baseline conditions. Conversely, changing soil pH had a negligible effect on emissions, when compared to altering soil pH under baseline conditions.

Given that the purpose of the sensitivity analysis in this study was to inform a realistic calibration effort, the range of adjustment of each variable was limited to within plausible bounds for the system of concern, when this information was available. Consequently, the results of the sensitivity analysis should be used as a guide only when extrapolating to other sites; the characteristics of other sites may encompass a wider or narrower range of potential values. At the study sites, simulated GHG emissions were least responsive to variables relating to flooding and initial soil N conditions, which may have been an effect of the relative low water table depth and high fertiliser rates found on these sites, respectively. In a system frequently subject to flooding or with low rates of additional N application, sensitivity to these variables may be much greater.

Using a sensitivity analysis to inform model calibration does not appear to be a widely used practice in the literature. It adds a targeted aspect to the ‘trial-and-error’ approach implicit in most other studies, and may consequently aid model calibration efficiency. Despite this, a degree of trial-and-error was required during model calibration. It is therefore possible that more satisfactory results may have been achieved using a different combination of calibration parameters. Regardless of the approach used, optimal calibration and validation of simulated data sets were also likely to have been limited to an extent by other sources of error and uncertainty.

3.4.3. Sources of error and uncertainty in DNDC modelling

The DNDC calibration and validation exercises highlighted a number of issues which may merit further investigation, or provide useful lessons in model operation. These issues relate to the quality of field data and other input data, and internal model functioning, and will be discussed here in relation to the behaviour of DNDC simulation of soil nutrient (C and N) and water dynamics.

Disentangling the primary sources of output error in relation to a complex, process-based model such as DNDC is difficult, given that a large number of interacting inputs are required for model parameterisation. While the results of this study indicated that there were three probable primary sources of modelling error (soil C, soil N, and soil moisture behaviour), these factors interact within the model, so pinpointing a single priority for further improvement without access to the model source code is challenging.

3.4.3.1. Simulated soil moisture

Throughout the baseline, calibration, and validation exercises, soil WFPS at 5 cm depth was frequently greatly over-estimated by DNDC. Examination of the simulated soil moisture profile to a depth of 50 cm revealed a similarly high modelled water content at most depths (typically 98% to 100%), and corresponding low soil O_2 content. This appeared to be the case regardless of the simulated water table depth dynamics over the annual cycle. The simulated output could signify an idiosyncrasy of DNDC, which does not significantly affect model functioning, or may be a reflection of the fact that while soil moisture is modelled internally on an hourly time-step (Li *et al.*, 1992a), it is only reported daily, so the reported value may only provide a snapshot of soil moisture dynamics. Clarification of the nature of output values is required in the DNDC manual (ISEOS, 2012), and potentially, model code adjustment is needed to optimise the simulation of soil WFPS. It is very difficult to discern from the literature to what extent this is necessary, as the majority of studies only report emissions estimates, rather than a detailed assessment of the performance of sub-models. While the results of this study show that it is possible to simulate emissions at least partially successfully under simulated high soil moisture conditions, the validation exercise illustrated relatively poor performance of the model using input data from other fields, particularly with respect to predicted N_2O emissions. As N_2O emissions can contribute significantly to the overall annual GHG budget of horticultural peat soils, correcting discrepancies in soil moisture simulation is crucial if DNDC is to be used for national inventorying purposes.

A key component in DNDC is the response of GHG emissions to water table dynamics (Cui *et al*., 2005a; Webster *et al*., 2013). This study found that output variables were relatively insensitive to variations in water table dynamics within the bounds measured in the field, but became highly responsive to variation when simulated water table depth was raised above 50 cm below mean soil surface level, at which point, emissions became substantially over-estimated. This highly sensitive boundary was problematic, and could be a function of the interaction between calibrated model variables, in particular, water table and SOC partitioning and their effect on microbial response. Water table depth was measured approximately monthly at these sites. Greater temporal resolution of water table dynamics may be crucial in the correct calibration and validation of DNDC, and future studies incorporating model calibration should prioritise capture of water table dynamics on a daily time-step. Within this study, it may be possible to parameterise the empirical water table model function, with some further research into peatland hydrology on cultivated sites.

3.4.3.2. Simulated soil carbon

Using available input data from the field sites in this study, it was not possible to accurately model the magnitude and vertical distribution of soil carbon determined by soil core C and N analysis. By year 25 to 26 of simulation, SOC in the top 30 cm of soil was heavily depleted, typically to only half of its measured (initial input) concentration. Of the DNDC studies conducted on peat soils, none provide details of the effect of pre-simulation on SOC depletion, or the factors influencing modelled SOC depletion. In this study, the degree of depletion appeared to be contingent upon a number of factors, relating to the allocation of SOM between litter, humad and humus pools, SOC pool C/N ratios and decomposition rates, and crop characteristics, tillage operations, and the way in which DNDC simulates SOC loss.

The most significant issue limiting realistic CO_2 flux simulation is likely to be the way in which DNDC models soil carbon loss to the atmosphere. In a cultivated peatland, carbon loss is usually accompanied by a resulting loss in soil depth over time (Kasimir-Klemedtsson *et al*., 1997). Consequently, each subsequent soil tillage operation lifts and incorporates a little more peat from lower soil layers, replenishing the surface carbon stock (Fig. 3.5.). Since diminishing soil depth over time does not appear to be modelled by DNDC, carbon is continuously taken from within the same upper layer of soil and is assumed to be replaced by mineral material (Fig. 3.5.). While this problem may be negated by adjusting other model parameters, it tends to result in a decline in surface SOC concentration and consequently, annual CO_2 emissions over time.

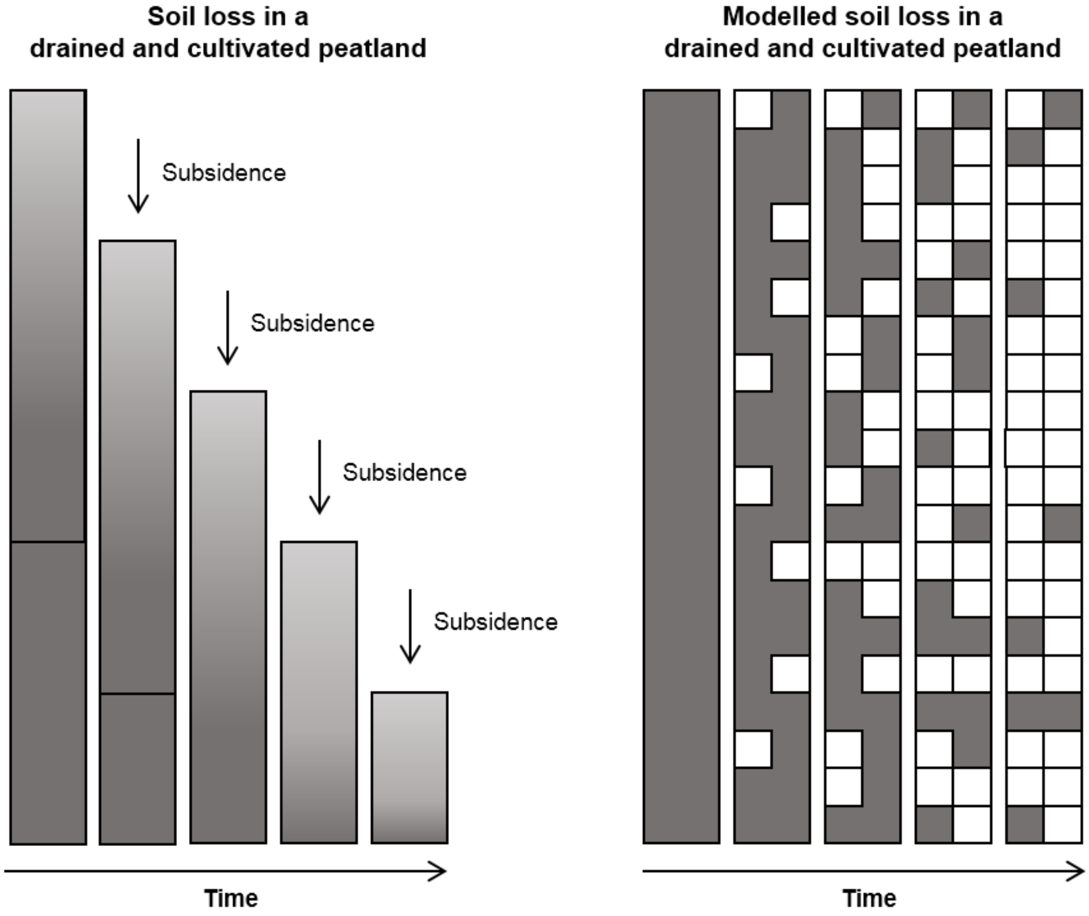

Figure 3.5. Soil loss in the upper 50 cm layer of soil under frequent cultivation. in a drained and cultivated arable peat soil (a), ploughing is accompanied by soil oxidation and consolidation, constantly replenishing soil C into the cultivated layer of soil (in this case, the upper 30 cm) over time. Conversely, in a DNDC-modelled drained and cultivated peat soil (b), oxidation is simulated by random removal of SOC from within the entire 50 cm soil layer, without subsidence or replenishment from deeper soil layers. Adapted from Couwenberg and Hooijer (2013).

Counteracting this problem associated with SOC simulation over time, may be to some extent achieved by adjustment of model parameters concerning SOC pools and crop characteristics (Cui *et al*., 2005 a; b; Frolking *et al*., 2004; Smith *et al*., 1997). In this study, the rate of SOC decline was slowed by providing additional simulated supplies of carbon to the soil from crop inputs – maximum crop biomass production, and crop root fraction, were both increased. This was necessary to counteract the imposed large increases in SOC pool decomposition rates, which were required to increase CO_2 emission to observed magnitudes. While the method of increasing crop carbon inputs has been used successfully on a mineral soil in this way by Hastings *et al*. (2010), it did not provide sufficient simulated SOC replenishment here. Possible limitations include the DNDC decomposition turnover period limiting litter incorporation into labile SOC pools, and the simulated yields and C/N ratios of the crops (Li *et al*., 1997). Notably, this study excluded cropping season and fallow period weed growth, since reliable estimates of vegetation composition, plant fractions, and fraction C/N ratios were unavailable. Including this additional vegetation input would be likely to improve DNDC modelling of GHG emissions. Preferably, any future research effort would also incorporate historical climatic and crop management data into the pre-simulation period, with the aim of recreating present-day soil characteristics.

One final point to add on the matter of SOC simulation concerns the substantial knowledge gap surrounding SOC pool characterisation in cultivated peat soils. Within DNDC, SOC is allocated between litter, humad (labile humus) and humus (recalcitrant humus) pools (Li *et al*., 1992a). The model automatically assigns a proportion of soil carbon to each pool (in this study, c. 17% humads and 82% humus, Table 3.1.). It is not known whether these proportions are realistic, since at present, analytical peat soil fractionation methods provide inconclusive evidence, especially when the anaerobic constraints to biodegradation are removed (Strosser, 2010). It is possible that this knowledge gap presents a substantial obstacle to accurate modelling of peat soil CO_2 emission.

3.4.3.3. Simulated soil nitrogen

Soil carbon and nitrogen cycles are closely coupled (Gruber and Galloway, 2008), so many of the factors affecting SOC behaviour outlined above, also apply to soil nitrogen simulation. Thus, accurate DNDC N_2O emission is dependent upon accurate modelling of DOC and soluble N (NH_4^+, NO_3^-) release from SOM (Li *et al*., 1992a). Additional sources of potential error include the reliability of available climatic and farm management data, and the method of estimating N_2O emissions from measured field data.

Originally, DNDC was constructed to model N_2O emissions in response to rainfall events, so a key feature of the model is the sensitivity of simulated N_2O to rainfall events (Li *et al*., 1992a). Weather data used for model input in this study was taken from the nearest UK Met Office weather stations to the field sites, since on-site daily rainfall measurement was not possible. Analysis of average rainfall data from the East Anglia region revealed that on a single day, rainfall could vary by up to 10 mm across the region (BADC, 2014). Clearly, any discrepancies caused by localised rainfall could strongly affect daily N_2O emission patterns and magnitude.

While efforts were made to obtain as many records of farm operations as possible, there were nevertheless some data gaps where records were not available (Table 3.4.). Gaps were filled by adapting management data from other fields on the same site, to known events observed during field measurement periods on the field of concern. It is possible that this resulted in some error in management input data, including the quantity and timing of mineral N fertiliser applications, propagating error into the model as a consequence. Given the high level of responsiveness to fertiliser application rates, and timing in relation to rainfall or irrigation events (e.g. Brown *et al*., 2002; Li *et al*., 1994), this may represent a substantial source of error in fields where farm records were incomplete. An additional potential source of error might be created through soil heterogeneity across the sampling area within each field, and its degree of representativeness of the heterogeneity across the whole site. While the results of studies in the same geographical region suggest that soil physical characteristics are well represented in this study (Dawson *et al*., 2010; Morrison *et al*., 2013), variation in soil N distribution may be more difficult to capture.

Annual emissions of measured N_2O were calculated using trapezoidal integration of episodic data points (Chapter 2). Field studies of high temporal resolution N_2O emissions demonstrate that fluxes tend to be highly episodic, and some authors suggest that calculation of annual emissions should take this into account (e.g. de Klein and Harvey, 2013). While the IPCC recommend using the trapezoidal integration method for the calculation of annual emissions from episodic data (IPCC, 2006), it is possible that it may over-estimate emissions, or if peaks are missed between field measurement events, under-estimate emissions. With this in mind, assessment of whether errors in N_2O magnitude lie with DNDC or with field measurements is difficult. Evaluation of the ability of DNDC to simulate N_2O emission should therefore primarily focus on the temporal distribution of peaks through the season. Since the temporal resolution of DNDC in predicting episodic peaks can sometimes vary by several days, this could be facilitated by the use of fit statistics that incorporate a temporal variability term such as a lag function, or by comparing observed and simulated emissions averaged over a period of 5-7 days. A similar approach incorporating lag functions has been applied successfully to DNDC-modelled data by Tonitto *et al*. (2007), and could form the basis of further investigation here.

3.5. Conclusions

This study presented a number of novel additions to the process of DNDC greenhouse gas emissions modelling: (1) Emissions of soil respiration CO_2, N_2O, and CH_4 were simultaneously modelled for two horticultural peat soils of differing SOM contents, over a range of commercially important crops; (2) A preliminary sensitivity analysis was used to successfully identify a number of key model variables influencing the magnitude of simulated GHG emissions; (3) Modelled and measured episodic values of soil physical variables were compared using fit statistics, to identify sources of error within DNDC sub-models which contribute to sub-optimal GHG emissions simulation.

The performance of DNDC at simulating emissions of GHGs from eutrophic Histosols under intensive horticultural production was variable, for different sites and GHGs. When calibrated, DNDC performed relatively well at reproducing the overall magnitude of emissions, but some refinement is required at the finer scale, particularly in terms of reproducing seasonal patterns of CO_2, N_2O and CH_4. Important sources of error within the model include limitations in the functioning of soil moisture and SOC pool dynamics. These could be addressed through further development of the model code (informed by further research on the nature of SOC partitioning), and by using historic climatic and farm management data sets to stabilise these sub-models during the pre-simulation period. Key components of model parameterisation include access to accurate current climatic and farm management data, and the ability to continuously monitor in-field water table depth. In conclusion, DNDC is unsuitable for ‘off-the-shelf’ modelling of horticultural peatlands, but may have the potential to provide reasonable estimates of GHG emissions with further refinement.

Chapter 4

Efficacy and practicality of mitigation measures for reducing greenhouse gases emissions from horticultural peat soils

Abstract

The UK government is committed to reducing greenhouse gas emissions from all national sectors by the year 2020. Emissions of CO_2, N_2O and CH_4 from the agricultural sector represented the second-largest national emissions source in 2012, at 10.1% of UK emissions. Peat soils under intensive horticultural production are a potentially substantial agricultural emissions source, and consequently represent a priority target for mitigation efforts. While a large number of candidate agricultural emissions mitigation measures are suggested in the literature, the evidence base for effective GHG mitigation when applied to horticultural peat soils is poorly developed. Ensuring adoption of measures on-farm requires the co-operation of farmers, including an understanding of causes of variability in potential levels of adoption within the horticultural sector. One accepted approach to prioritising mitigation measures in the absence of a consistent systematic evidence base is to seek the opinion of experts and farmers concerning the efficacy and practicality of measures, respectively. This study elicits the views of experts and farmers using Best-Worst Scaling, a discrete choice survey approach. The method allowed ranking of individual measures on a ratio scale of effectiveness (expert opinion) and practicality (farmer opinion). Three measures were considered to be both effective and practical, and may be considered as priority for implementation. The measure with the greatest combined approval from experts and farmers was *reduce nitrogen fertiliser dose*. Many of the top-ranked effective measures relate to implementing substantial land use changes, and were consequently perceived by farmers as impractical. Considerable policy changes may be required to incentivise significantly levels of GHG emissions mitigation from horticultural peat soils.

4.1. Introduction

Cultivated lowland peat soils (Histosols) are amongst the most productive agricultural soils in Europe (Parish *et al*., 2008). They are also susceptible to high rates of soil loss, primarily via microbial mineralisation of the peat substrate accompanied by greenhouse gas (GHG) release, as well as via other routes such as water and wind erosion (Dawson and Smith, 2007). Estimates of soil loss rates from European arable Histosols are scarce, but vary between 0.50 and 3.00 cm yr^{-1} in Sweden (Kasimir-Klemedtsson *et al*., 1997), and in the principal area of arable production in the UK (East Anglia) loss rates varied from 0.27 to 3.09 cm yr^{-1} between 1942 and 1971 (Richardson aand Smith, 1977; Hutchinson, 1980) and from 1.10 to 1.48 cm yr^{-1} between 1982 and 2004 (Dawson *et al*., 2010). Annual GHG budgets from cropped peats are commonly dominated by CO_2 release (e.g. Elder and Lal, 2008; Maljanen *et al*., 2004), indicating potentially high accompanying soil organic carbon (SOC) losses. With a typical peat depth of c. 1-2 m remaining across much of the UK cropped peat area, the soil resource may become exhausted entirely within the next 100-250 years, with soil quality tending to decline as lower depths are exploited (Cannell *et al*., 1999; Dawson *et al*., 2010). Long-term sustainable crop production on peat soils is therefore contingent upon minimising soil loss rates per unit crop yield; mitigating soil loss by reducing GHG emissions is one crucial means of achieving this. Reducing net emissions of CO_2, N_2O and CH_4 is also important in meeting national targets aimed at mitigating climate change (UK Parliament, 2008).

A large number of mitigation measures (MMs) have been identified to potentially reduce GHG emissions from horticultural peat soils (e.g. Akiyama *et al*., 2004; Asgedom and Kebreab, 2011; Baldock *et al*., 2012; Cole *et al*., 1997; Dalal *et al*., 2003, 2008; de Klein *et al*., 2001; Fitton *et al*., 2011; Garnett, 2011; Gregorich *et al*., 2005; Henault *et al*., 2012; Lal, 2008; Lal *et al*., 2011; Le Mer and Roger, 2001; Rounsevell and Reay, 2009; Smith *et al*., 2008; Synder *et al*., 2009). Despite many options being available, the evidence base upon which to make strategic changes in management practices remains poor. Specifically, robust evidence is lacking in relation to various MMs' actual ability to reduce net GHG emissions under different climatic, soil, and management practice scenarios. Further, any unexpected negative consequences associated with the adoption of new management operations or systems remains largely unknown (e.g. yield loss, pathogen persistence, or enhanced nitrate leaching).

The European Union has a relatively advanced agricultural GHG mitigation strategy compared to many world regions (Norse, 2012), and several studies have provided a basis for estimating potential effectiveness, practicality and cost of a range of MMs on arable soils in the UK (Barnes *et al*., 2010; Jones *et al*., 2010; Moran *et al*., 2008; MacLeod *et al*., 2010). There remains a substantial knowledge gap surrounding these mitigation options when implementing interventions targeted specifically towards peat-based soils. Implementation of MMs has also proven problematic as patterns of GHG emission from Histosols tend to differ from those from mineral soils (Smith *et al*., 2010b), with associated differences in efficacy, practicality and cost of mitigation implementation. Consequently, the evidence base for the systematic application of interventions, particularly to horticultural systems, remains incomplete. Farm management and policy decisions should be based on the current best available data. If this information is not available, an alternative approach is required to facilitate implementation of the potentially most appropriate MMs.

One accepted approach to prioritising MMs in the absence of a consistent systematic evidence base is to seek expert opinion on the efficacy of proposed options and to provide an estimate of the level of confidence in the resulting recommendations by including a measure of the uncertainty (level of consensus) associated with elicited expert opinion (Cross *et al*., 2012). In addition to reducing GHG emissions, MMs must be perceived by land owners and managers as being practical to implement on-farm (Barnes *et al*., 2010; Smith *et al*., 2007b). Farmer preferences for different MMs may vary according to the particular situation of individual farmers, for example the size of the farm holding (Jones *et al*., 2013), typical crop rotations, constraining environmental factors associated with the farm’s location or cost. By combining estimates of effectiveness and the practicality of MMs for different sectors of the horticultural community, sustainable development stakeholders (e.g. policy-makers) may better anticipate potential rates of adoption and reduction in environmental damage. Further, this can be used in the targeting of incentives or policy adjustments which may encourage uptake (Jones *et al*., 2010). Previous work eliciting expert or land managers’ opinions on mitigating GHG emissions for temperate cropped agro-ecosystems is scarce (e.g. Fairweather *et al*., 2009; Noble and Christmas, 2008). Expert and farmer consultation within the UK has been limited to a small number of, albeit, relatively comprehensive studies (Barnes *et al*., 2010; Jones *et al*., 2010; Moran *et al*., 2008; MacLeod *et al*., 2010), none of which specifically address the application of measures to cropped Histosols.

This paper aims to evaluate which farm-scale MMs may be most effective and practical for mitigating GHG emissions on horticultural peat soils, by novel application of a survey method known as Best-Worst Scaling (BWS). Specifically, it aimed to: (1) Determine which MMs are perceived by academic and industry experts as most effective at reducing overall GHG emissions; (2) Determine which MMs are perceived by farmers as most practical to apply on the farm; (3) Identify which MMs are considered to be both effective and practical; and (4) Examine the potential for reduction in GHG emissions and associated soil loss when identified effective and practical MMs are applied to horticultural peat soils.

4.2. Methods and materials

4.2.1. Best-Worst Scaling

Best-Worst Scaling (BWS) is an extension of the “paired comparisons” method (Thurstone, 1927) and requires respondents to choose at the extremes of their preferences (“best” and “worst”), from repeated sets of three or more different combinations of items drawn from a longer master list of items (Auger *et al*., 2007). The measures presented for comparison possessed the potential to mitigate GHG emissions from peat soils under horticultural production. Respondents were asked to indicate their preferences by choosing GHG MMs within each set which they perceived as “most effective” or “most practical” and “least effective” or “least practical”.

A mean preference score for each individual mitigation measure can be calculated using BWS by aggregating the “best” and “worst” choices from repeated sets made by a population of respondents. For a single set of five items A to E, it is possible to deduce the order of preference for seven out of ten pairs of combinations within that set. If the respondent chooses A as best and E as worst, then it is known that: A > B, A > C, A > D, A > E, B > E, C > E, and D > E, where “>” signifies “is preferable to” (Sawtooth Software, 2013). Although no information is provided on how B compares with C or C with D, analysing the best and worst choices from repeated sets of different combinations of measures gives enough information to compute a score for the relative preference of respondents for each mitigation measure along an arbitrary, interval scale (Jones *et al*., 2013).

Surveys were designed using Sawtooth SSI Web software (Sawtooth Software Inc., Orem, UT, USA), following the guidelines for optimal survey design. The recommended five items per set of MMs were used to minimise respondent fatigue, and twelve sets of MMs were used within each survey, to enable presentation of each measure to each respondent at least twice in different sets. Using the BWS survey design module, one thousand possible design iterations were prepared for each survey, and the optimal design selected by the software to fulfil the following criteria: (1) each measure appeared an equal number of times; (2) each measure was paired with each other measure an equal number of times; (3) each measure appeared on the left and the right of the design an equal number of times (Sawtooth Software, 2013). Multiple versions of the optimal survey design were used in the online survey to vary the position of

MMs within each set and combination of measures across respondents, thereby minimising context bias (Sawtooth Software, 2013). An example of a BWS set is provided in Table 4.1.

Table 4.1. Example of a “practicality” Best-Worst Scaling choice set.

Most Practical		Least Practical
○	**Eliminate tillage** (i.e. zero-till)	○
○	**Grow crops with a lower nitrogen requirement**	○
○	**Avoid additional draining and cultivation of uncultivated wetlands**	○
○	**Convert cropland to grassland** (e.g. pasture)	○
○	**Minimise avoidable use of farm machinery** to reduce fossil fuel consumption	○

Respondents were invited to check one, mutually exclusive, radio button in each column, corresponding to their opinion on the ‘most practical’ and ‘least practical’ option respectively within each set.

4.2.2. Round 1: Short-listing mitigation measures

Measures, with the potential to reduce overall GHG emissions from horticultural peat soils, were identified from relevant peer-reviewed papers and grey literature, resulting in an initial list of 58 candidate MMs. Initial selections were made on the basis of measures having the potential to be applied at the farm scale, to peat soils under horticulture in temperate zones; as very few specific references were made to horticultural peat soils in the literature, measures necessarily included options drawn from other contexts (e.g. mineral soils, arable systems). The MMs were arranged into themed sub-lists to facilitate comparison: nutrient management (fifteen measures); soil moisture management (five measures); crop management (seven measures); fallow and residue management (five measures); tillage and machinery operations (three measures); general soil management (seven measures); fossil fuel consumption (five measures); carbon sequestration (six measures); and miscellaneous items (five measures).

The list of MMs was shortened to a more cognitively bearable 30 by inviting an expert panel to classify each item according to its potential to reduce overall GHG emissions on horticultural peat soils if implemented. Experts were identified on the basis of authorship of relevant papers or employment at relevant organisations, or through peer networks, and were chosen from academia, government and industry. They were selected from a number of different disciplines, broadly relating to GHG emissions (agronomy, peatland/Histosol management, and soil sciences). Respondents were asked to classify each measure as “very effective”, “quite effective”, “slightly effective”, “not effective” or “don’t know”. Measures were assigned scores of “3”, “2”, “1”, “-1”, and “0” respectively (adapted from Jones *et al*., 2013), then summed to give a total score. Panel members were offered the opportunity to comment on the results prior to selecting the top ranking 30 measures which were subsequently presented in the second survey round (BWS survey; Table 4.2.).

4.2.3. Round 2: Best-Worst Scaling

In the second round, expert respondents were asked to re-evaluate MMs in terms of their *potential effectiveness at reducing or offsetting GHG emissions per hectare of horticultural peatland (relative to current practice)*, by viewing 12 sets containing different combinations of five measures and selecting the “best” and “worst” measure from each set, representing the most and least effective MMs respectively. A parallel survey of the same design was sent out to horticultural farmers and growers to capture their perceptions on the *practicality of implementation on the horticultural peat soils on their farm*. Each respondent was asked to disregard any concerns other than effectiveness (experts) or practicality (farmers) when comparing MMs, and to ignore the application of measures to other soils, enterprises or farms. *Horticultural peat soils* were defined as *land under horticultural production (vegetables, fruit or ornamentals) which is on peat or peaty soils*. Online BWS surveys were disseminated during February-March 2014.

4.2.4. Respondent recruitment

The expert panellists were selected using the same criteria as for the shortlisting round; additional respondents were invited after an initial low response rate.

Horticultural growers were selected directly from existing industry contacts and indirectly through newsletters and blogs of grower trade organisations. Growers qualified for survey invitation if they had peat soil on any part of their farm, and had grown any horticultural crops on all or part of their peat land in any years during the preceding five year period. Farmers were also asked for details of their gender, age, geographical region, main enterprise (horticulture vegetables, horticulture fruit, horticulture ornamentals, horticulture other, arable, other), proportion of their land under horticultural peat soils, prior or current implementation of mitigation measures, and prior knowledge of greenhouse gas emissions from cropping.

Table 4.2. Mitigation measures shortlisted by experts for inclusion in the Best-Worst Scaling survey.

No.	Measure
1	**Avoid additional draining and cultivation of uncultivated wetlands**
2	**Restore cropland to native wetland vegetation** (e.g. wet woodland; lowland meadow; lowland heathland; reed beds), using minimal cultivation
3	**Reduce nitrogen fertiliser dose** to avoid loss of excess nitrogen (e.g. avoid applying fertiliser to legumes)
4	**Improve timing of nitrogen fertiliser application** to avoid loss of excess nitrogen (e.g. apply just before crop growth; avoid application for 5 days after heavy rain; use split timing application)
5	**Minimise avoidable use of farm machinery** to reduce fossil fuel consumption
6	**Avoid growing artificially heated protected crops** (heated greenhouse or heated poly-tunnel crops)
7	**Use more fuel-efficient machinery and equipment with better power ratings**
8	**Increase on-farm planting of trees and shrubs** (e.g. in inter-row cropping; orchards; shelterbelts; buffer strips; on marginal land)
9	**Use frequent soil and crop nutrient status testing and nutrient management planning** to optimise farm nitrogen budgeting
10	**Eliminate tillage** (i.e. zero-till)
11	**Convert cropland to grassland** (e.g. pasture)
12	**Improve spatial nitrogen fertiliser placement** to avoid loss of excess nitrogen (e.g. use shallow injection or foliar feeds)
13	**Use alternative fuels to fossil fuels** (e.g. in machinery, and for on-farm processing and power and heat supply)
14	**Use precision farming** (e.g. GPS; annual machinery calibration) to apply chemicals and irrigation according to spatial variation in crop requirements, and to minimise soil compaction
15	**Increase the use of perennial food crops**, to promote soil organic carbon retention
16	**Optimise irrigation system efficiency** to keep soil moist but not saturated, control soil nitrogen release, and reduce pump fuel use
17	**Reduce tillage** (e.g. less frequent or shallower tillage; use direct drilling instead of transplanting)
18	**Diversify crop rotations** to optimise soil nutrient status and organic carbon return to soil (e.g. include perennial crops or temporary pasture in rotations)
19	**Avoid or shorten bare fallow periods** (e.g. grow fast-growing cover crops such as green manures)
20	**Increase reliance on nitrogen-fixing crops (legumes)** to replace mineral nitrogen fertiliser
21	**Use anaerobic digestion with energy recovery, or bio-fuel production, to process organic wastes** (e.g. crop residues)
22	**Use integrated pest management** to reduce synthetic pesticide use and resource wastage through crop losses
23	**Grow crops with a lower nitrogen requirement**
24	**Optimise ground water levels** at each stage of the annual crop cycle (e.g. maintain shallow water table for longer; maintain water table above depth of drainage pipes)
25	**Install buffer zones** between fields and watercourses, and within fields, to catch leached nutrients and soil organic carbon
26	**Change sowing and harvesting dates to optimise nutrient use efficiency**
27	**Minimise soil compaction from wheeled operations**, to minimise nitrous oxide and methane emissions
28	**Apply nitrification inhibitors to soil when applying urea or ammonium-based nitrogen fertiliser** (e.g. mineral nitrogen, manure)
29	**Maintain optimal soil pH**, to optimise plant growth and reduce overall greenhouse gas emission
30	**Grow crops that maximise both crop yield efficiency and residue return to the soil**

4.2.5. Analysis

Data acquired using BWS may be analysed by applying a choice model derived from random-utility (RU) theory (Thurstone, 1927). In a survey such as this one where respondents make both a “best” and a “worst” choice within each of the five MM sets, the model assumes that their choices will represent the maximum difference in performance (effectiveness or practicality) within each set. Within a set, the “best” selection is treated as a utility maximising decision and the “worst” selection treated as a utility minimising decision. Using the example of effectiveness, as score for respondent *r* choosing measure *A* as the best (most effective) option is computed:

$$E_{rA} = \delta_A + \varepsilon_{rA} \tag{4}$$

where δ_A is the position of measure *A* on the underlying effectiveness scale, and ε_{rA} is an error term. The error term is generated using a Gumbel (extreme value type I) distribution, which results in scores based on relative probabilities rather than absolute scores (Auger *et al.*, 2007).

Within a single set of *K* measures, the probability of respondent *r* choosing measure *A* as best (most effective, *b*) and *E* as worst (least effective, *w*) respectively can be calculated using the multinomial logit (MNL) model (after Marley and Louviere, 2005):

$$P(r_{Ab}, r_{Ew}) = \frac{e^{\delta_A - \delta_E}}{\sum_{b=1}^{K} \sum_{w=1}^{K} exp^{\delta_b - \delta_w} - K} \tag{5}$$

The model posits that the probability of choosing the pair *A*, *E* as best and worst respectively is greater than the maximum probability of any other pair of MMs within that set being chosen as best-worst (Finn and Louviere, 1992). Incorporation of the Gumbel error term allows the model to estimate scores by maximising the fit between the distribution of best-worst probabilities and the actual observed choice behaviour of the respondent.

Using an extension of the MNL model, it is possible to aggregate the estimated scores for each set from each respondent, and calculate individual-level scores. In the Sawtooth BWS analysis module (Sawtooth Software, 2013), this is achieved using hierarchical Bayes (HB) analysis, and borrows information on the pattern of responses across the whole sample of respondents to stabilise each respondent’s scores. An anchoring point is provided by setting the utility of the final item of *K*-1 items in the study to zero, then computing all other scores relative to the zero-anchor. Bayesian analysis is useful because it allows estimation of a mean score for each individual respondent and for each MM, as well as an estimate of variability around MM mean scores (Cross *et al*., 2012). Measures may then be compared to each other along the common continuum of interest (effectiveness or practicality), while the estimate of variability provides important information on the degree of consensus or disagreement between experts or growers.

Model performance (scores stability) was assessed by examining the overall model Fit Statistic (FS) after modifying the number of spin-up iterations (range 10000 to 50000), model iterations (range 10000 to 50000), and prior degrees of freedom (range 2 to 5) within the HB module. Changing the default parameters resulted in < 3% difference to both the overall model and individual-level fit statistics. Examination of the probability generation graphic revealed the most stable signal using the default parameters (20000 spin-up iterations, 20000 model iterations, prior degrees of freedom = 5, prior variance = 1, and starting seed = 1) compared to modified parameters; these results were subsequently used for further analysis.

Following scores generation, unreliable respondents (those who appeared to answer randomly) were identified by examining the individual-level FS, and excluding any respondent with a FS of < 25% (Sawtooth Software, 2013). Since all expert and farmer respondents demonstrated an individual-level FS of ≥ 40%, all data were retained for further analysis.

After computation of raw estimated scores, the BWS module transforms these scores so that they sum to 100 across all MMs for each respondent (“rescaled scores”), giving scores that are more easily interpretable and on a ratio scale. All values presented in the results section are in terms of mean rescaled score ± 95% confidence interval; all analyses have been conducted on rescaled scores. Figures 1 to 4 use zero-centred rescaled scores, where the origin (zero) represents the mean of all effectiveness or practicality scores; any positive score represents an above-average preference for an MM, and any negative score represents a below-average preference for an MM.

Non-parametric analyses (1-sample Kolomogorov-Smirnov test, with Bonferroni-corrected Mann-Whitney U tests on significant results) were conducted to compare mean scores of both individual MMs and MMs grouped by category. For the effectiveness surveys, the effect of confidence scores, and area of expertise, on mean scores were tested. The number of completed practicality surveys was small, so statistical analysis of the effect of farmer characteristics on mean scores was limited, and no comparison was made between categorical demographic variables and mean scores. It was possible however to conduct correlation analyses using Kendall’s Tau statistic (τ), between mean scores and farm size characteristics (total farm area, total peat area, and total horticultural peat area on each farm, all in ha; and the % of each farm area cropped to horticulture on peat soils). These correlations should be treated as a preliminary exploration of the data, since $n = 6$ or 7 in each case.

4.3. Results

4.3.1. Expert and grower characteristics

Forty-three experts (49% of those contacted) completed the first (short-listing) round of the survey. In the second (BWS) round, thirty-eight complete expert surveys (38% of those contacted) and seven completed farmer surveys were collected.

All experts in both survey rounds gave details of their expertise, with most experts specifying multiple areas of expertise (Table 4.3.). The range of expertise of respondents was broadly similar in both rounds, but with slightly fewer peatland experts and a slightly greater number of agricultural experts in the BWS round, as a proportion of the range of expertise. The confidence of experts in their answers during the shortlisting round was relatively low on a per-category basis, with only 7-30% of respondents stating they were “very sure” or “fairly sure”, and 7-30% of respondents stating they were “very unsure” of their selection of measures for retention or disposal. In the BWS round, most experts showed a relatively high level of confidence in their answers, with 71% saying they were “very sure” or “fairly sure” of their best-worst responses; no respondents said they were “very unsure” of their answers.

All farmers volunteered some demographic information, summarised in Table 4.4. All growers farmed in England apart from one, who farmed in Wales. Five farmers stated they already implemented one or more measure on the list provided, with three of them stating they practice four MMs.

Table 4.3. Expertise of respondents completing the short-listing and BWS effectiveness rounds

Area of expertise	Number of respondents	
	Short-listing round	BWS round
Agriculture general	9	9
Agricultural greenhouse gas emissions	15	21
Agricultural carbon dioxide emissions	8	12
Agricultural nitrous oxide emissions	13	19
Agricultural methane emissions	6	15
Nutrient management	10	13
Crop breeding and productivity	2	1
Soil science general	22	25
Peatlands general	18	15
Peatland greenhouse gas emissions	20	19
Renewable energy	8	7
Carbon sequestration	15	18
Other	3	8

Experts were allowed to specify multiple areas of expertise. Number of experts in shortlisting round 1 = 43, and in BWS round 2 = 38.

Table 4.4. Farmer demographic information provided by BWS respondents

Grouping variable	Category	Number of respondents
Gender	Male	7
	Female	0
Age group	18-34	2
	35-44	0
	45-54	3
	55-64	2
	65+	0
Location	England	6
	Wales	1
	Scotland	0
	Northern Ireland	0
0	Horticulture (vegetables)	1
	Horticulture (fruit)	0
	Horticulture (ornamentals)	1
	Other horticulture	0
	Arable	3
	Other [2]	2
Farm size (ha)	<100 ha	0
	100-199	2
	200-499 ha	2
	500+ ha	3
Peat area (ha) [1]	<100 ha	1
	100-199	2
	200-499 ha	1
	500+ ha	2
Horticultural peat area (ha) [1]	<100 ha	3
	100-199	0
	200-499 ha	1
	500+ ha	2
Tenancy	Owner	4
	Tenant	0
	Other	2
Number of measures implemented	0	2
	1	1
	2	1
	3	0
	4	3

[1] Not all respondents provided a response to these questions. [2] Both respondents stated “salads”.

4.3.2. Short-listed measures

After shortlisting, the categories of MMs with the greatest number of retained measures were nutrient management (seven out of fifteen measures), fossil fuel consumption (all five measures), and carbon sequestration (five out of six measures); these measures dominated the top-ranked half of the shortlist, occupying twelve of the top fifteen places. Only two or three measures were shortlisted from each of the remaining categories.

4.3.3. Expert effectiveness scores

Five MMs had mean scores significantly greater than the overall mean score (i.e. their confidence intervals did not overlap the zero-centre; Fig. 4.1.). *Avoid additional drainage and cultivation* (a carbon sequestration MM; MM number 1 in Table 4.2.) was preferred as the most effective measure for reducing overall GHG emissions, with a score of 5.27 ± 1.70. The other four measures related to carbon sequestration (*Restore cropland to native wetland vegetation* (MM2), 4.75 ± 2.45; *Convert cropland to grassland* (MM11), 3.23 ± 2.36), nitrogen management (*Reduce nitrogen fertiliser dose* (MM3), 3.34 ± 1.91), and fossil fuel consumption (*Avoid growing artificially heated protected crops* (MM6), 3.09 ± 2.21).

Thirteen measures had significantly lower scores than the overall mean score (Fig. 4.1.), with the lowest scoring MM (-3.24 ± 0.15) being *Use Integrated Pest Management* (MM22, relating to fossil fuel consumption). The other twelve measures perceived as relatively ineffective were drawn from across the range of MM categories, with the exception of fallow and residue management, and tillage and machinery operations categories, which contained no MMs considered to be of less than average effectiveness.

A general increase in the width of the confidence interval was observed as the MM mean score increased, indicating a lower degree of consensus between experts regarding effective measures compared to ineffective measures. No trend was observed in the categories of MMs receiving high or low mean scores, although both miscellaneous measures, and four out of five fossil fuel consumption measures had scores below the overall mean, and both tillage measures were perceived as having above-average effectiveness.

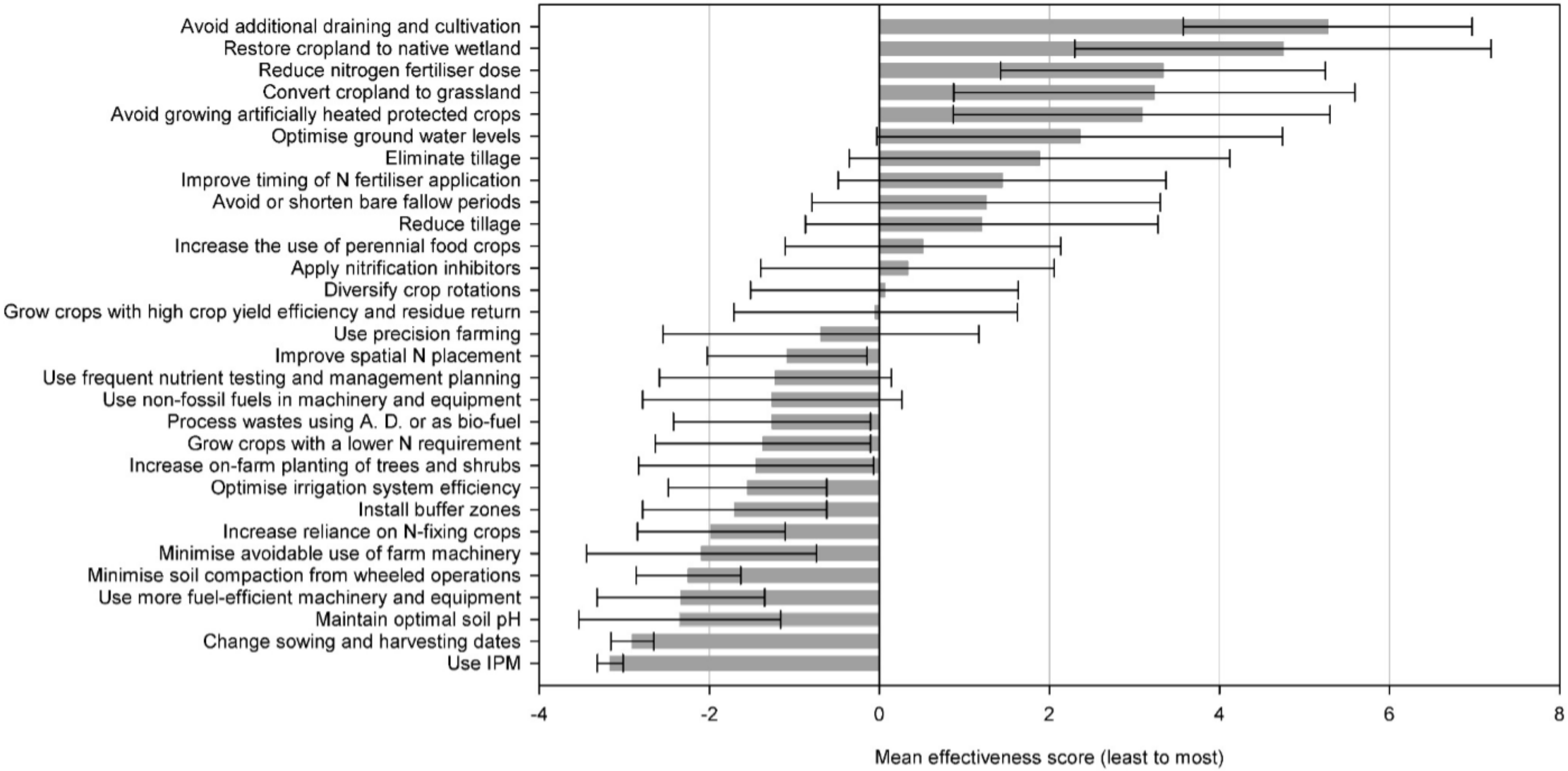

Figure 4.1. Mean estimates of relative effectiveness scores across all experts for the 30 shortlisted MMs. Error bars represent 95% confidence intervals of the mean scores.

4.3.4. Farmer practicality scores

Five MMs scored significantly higher than the overall mean (zero) score for practicality (Fig. 4.2.). The measure perceived as most practical to implement on the farm (i.e. the highest-scoring measure, at 5.21 ± 1.60) was *Improve timing of N fertiliser application* (MM4; a nutrient management measure). The remaining four MMs related to miscellaneous measures (*Use precision farming* (MM14), 5.18 ± 1.63), nutrient management (*Use frequent soil nutrient testing and management planning* (MM9), 5.15 ± 0.79), soil moisture management (*Optimise irrigation system efficiency* (MM16), 5.00 ± 1.39), and fossil fuel consumption (*Use more fuel-efficient machinery and equipment* (MM7), 3.37 ± 2.36).

Ten measures were perceived as significantly less practical than the overall mean practicality score (Fig. 4.2.). The lowest-scoring measure was *Restore cropland to native wetland* (MM2, a carbon sequestration measure, with a mean score of -3.32 ± 0.05). The other nine measures were drawn from just over half of the full range of MM categories (tillage management, fallow and residue management, carbon sequestration, crop management, and fossil fuel consumption categories).

The greatest degree of consensus between growers on the practicality of MMs was observed at the extremes of preference, with greater disagreement over the measures with mean scores closer to the overall mean (Fig. 4.2.). The highest agreement between farmers was seen for the measures perceived as least practical to implement. Similarly to the effectiveness results, categories of measures were distributed across the whole spectrum of scores, although tillage and crop management measures received negative scores relative to the overall mean, and nutrient and general soil management MMs tended to score highly.

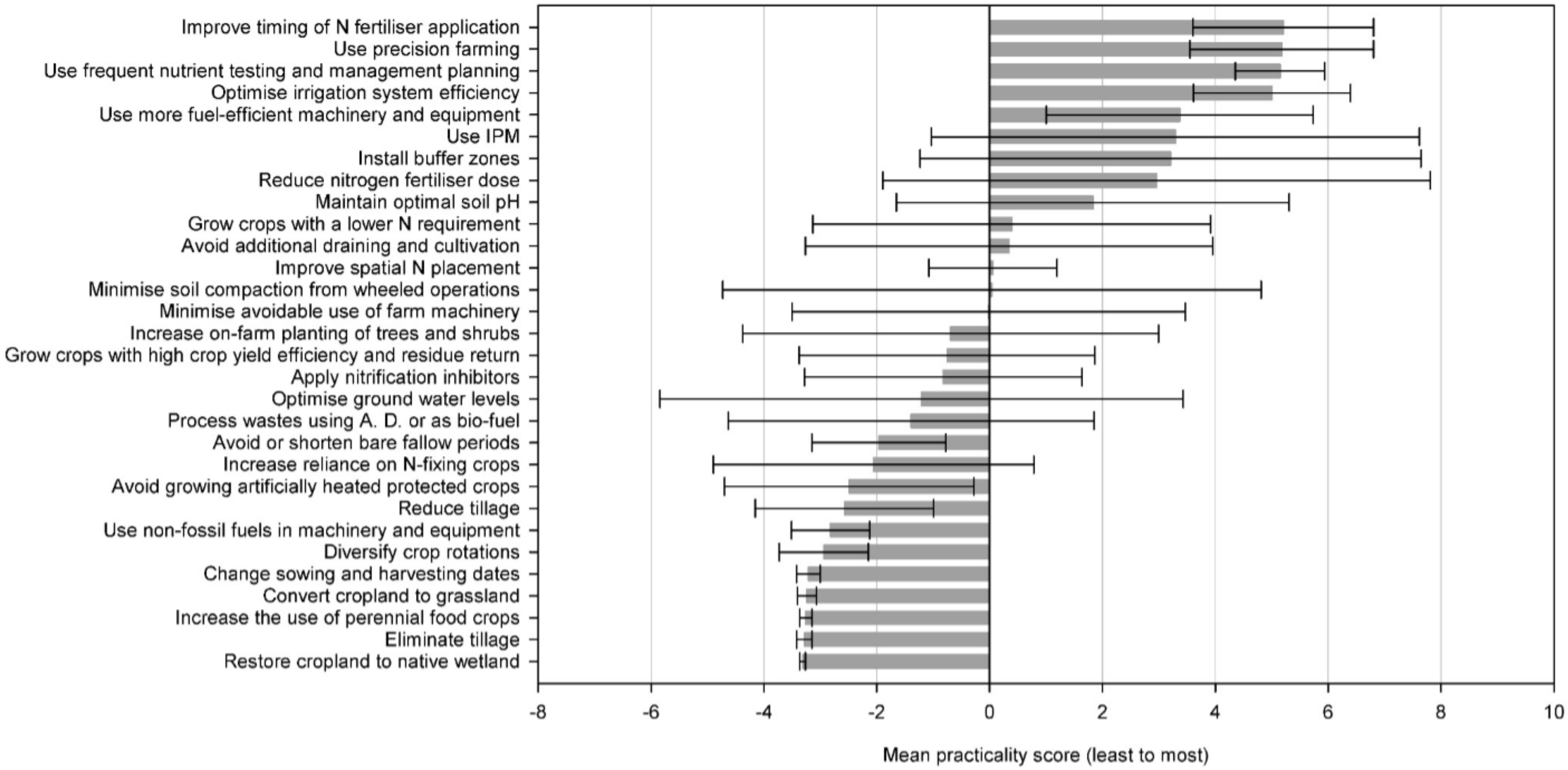

Figure 4.2. Mean estimates of relative practicality scores across all farmers for the 30 shortlisted MMs. Error bars represent 95% confidence intervals of the mean scores.

4.3.5. Effectiveness and practicality combined

Mean effectiveness and practicality scores are plotted for cross-comparison in Fig. 4.3. Only three MMs scored above-average for both effectiveness and practicality (Fig. 4.3., top right-hand quadrant). Two related to soil nutrient management (*Reduce nitrogen fertiliser dose* (MM3) and *Improve timing of N fertiliser application* (MM4)), and the other related to carbon sequestration (*Avoid additional draining and cultivation* (MM1)).

Measures considered effective at reducing net GHG emissions, but impractical to apply on the farm, occupy the bottom right-hand quadrant of Figure 4.3. These include MMs from most categories of measures, including both of the tillage measures (*Eliminate tillage* (MM10) and *Reduce tillage* (MM17)). Ten measures were considered practical but relatively ineffective at reducing GHG emissions (Fig. 4.3, top left-hand quadrant). Practical but ineffective MMs were also drawn from the majority of MM categories, with no particular category dominant.

In the bottom left-hand quadrant of the effectiveness-practicality matrix, seven measures were perceived as both ineffective and impractical (Fig. 4.3.). These comprised MMs within only half of measure categories (fossil fuel consumption, crop management, carbon sequestration, nutrient management, and miscellaneous measures).

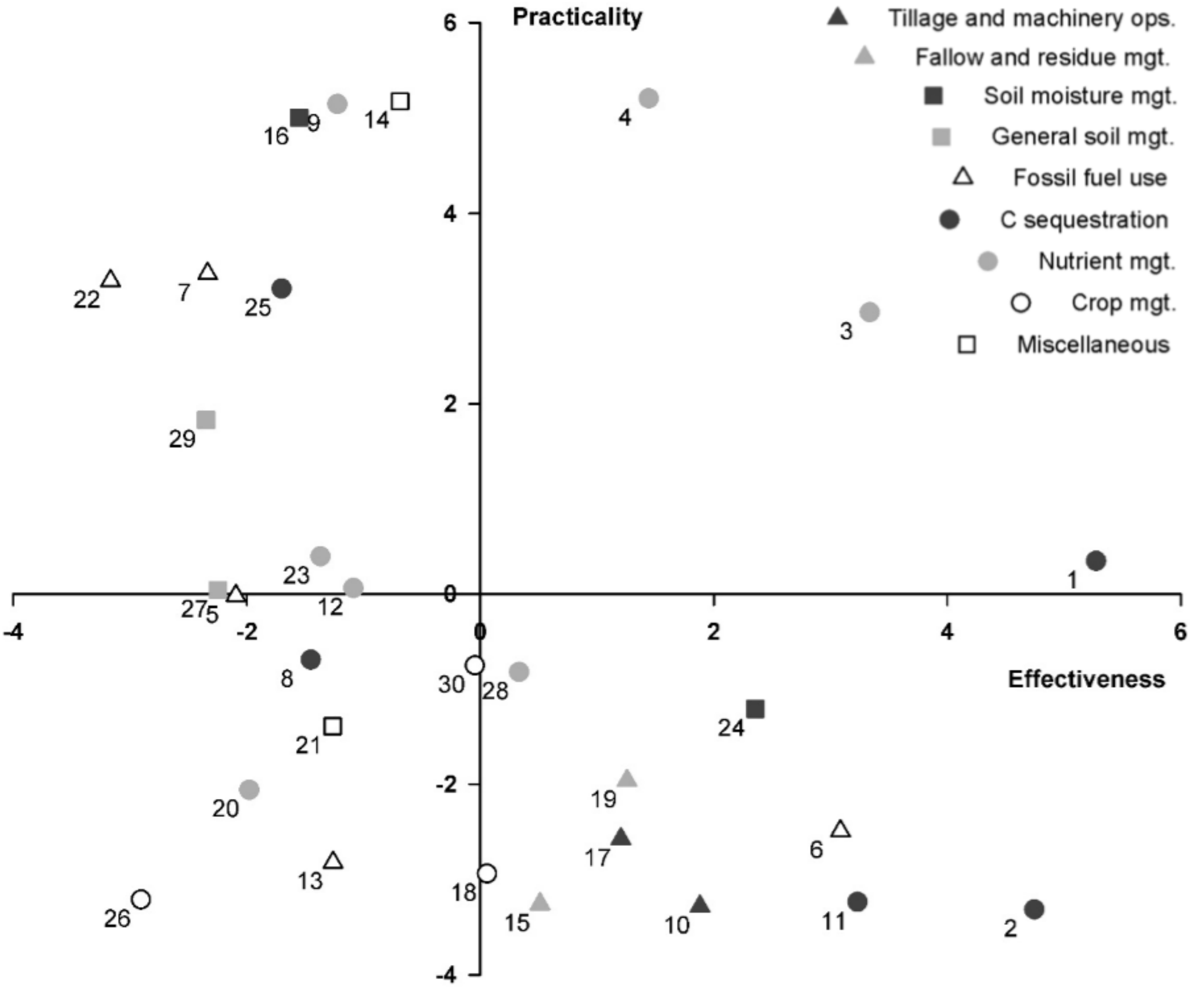

Figure 4.3. Zero-centred scatter plot of relative mean effectiveness and practicality scores for the 30 measures, classified by MM type.

4.3.6. Heterogeneity of responses

4.3.6.1. Frequency distributions of respondent scores

The degree of consensus between individual respondents can be inferred from the distributions of individuals' mean scores. The distribution of expert and grower mean scores for the measures which received mean scores greater than the overall mean (MM1, MM3 and MM4, Fig. 4.3.) were plotted separately and examined for degree of consensus (Fig. 4.4.).

There was a particularly high level of consensus amongst experts on measure 1, as indicated by the highly positively skewed distribution of scores greater than the overall mean (Fig. 4.4.a.). Similarly, although individual scores for MM3 were more widely distributed, a positive skew was also observed (Fig. 4.4.b.). The degree of agreement between experts for MM4 was lower, with the scores exhibiting a slightly negatively skewed distribution (Fig. 4.4.c.).

Few farmers completed the survey, but some observations can still be made in relation to score distributions. For MM1, despite the above-average mean score, individual-level scores show a negatively skewed distribution (with the modal score lower than the mean score), suggesting some disagreement between growers about the practicality of this measure (Fig. 4.4.d.). The pattern of scores for both MM3 and MM4 exhibit a positive skew, more strongly observed for MM4 which has no scores allocated below the overall mean score (Figs. 4.4.e. and 4.4.f. respectively). This indicates that the small number of growers taking part in the survey exhibited a high level of consensus on the practicality of applying MM4 to their horticultural peat soils.

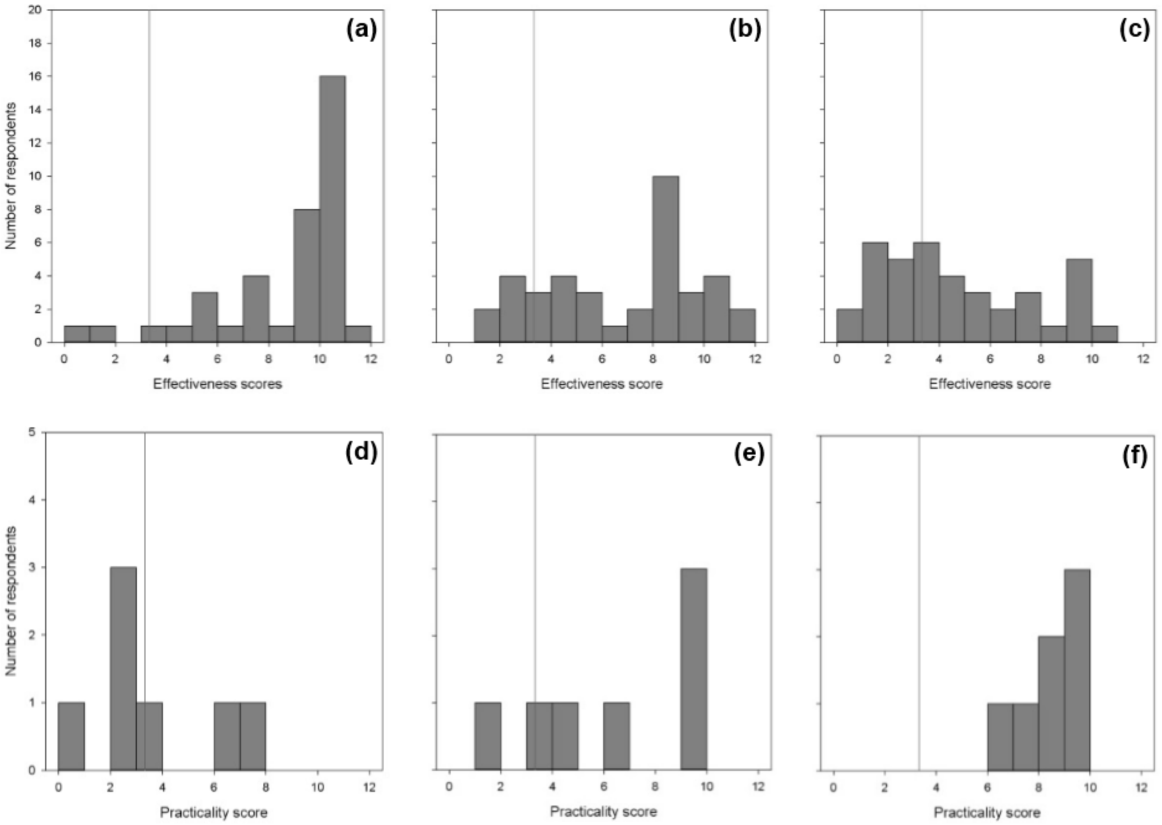

Figure 4.4. Distribution of individual, expert effectiveness scores for the top ranking MMs: (a) Avoid additional drainage and cultivation (MM1), (b) Reduce nitrogen fertiliser dose (MM3), and (c) Improve timing of fertiliser N application (MM4); and farmer practicality scores for (d) Avoid additional drainage and cultivation (MM1), (e) Reduce nitrogen fertiliser dose (MM3), and (f) Improve timing of fertiliser N application (MM4).

4.3.6.2. Comparison of scores between expert and farmer sub-groups

Non-parametric analysis revealed an insignificant effect of area of expertise on the mean scores of different MM categories ($p > 0.05$). Area of expertise also had no significant effect on perceived efficacy of individual measures, except for *Optimise ground water levels* (measure 24, $p < 0.05$), where *crop breeding and productivity specialists* gave a significantly lower score than *peatlands (general) experts* ($p < 0.05$), and *other experts* gave a significantly lower score than *agricultural CO_2 specialists* ($p < 0.05$).

No significant relationship was found between experts' confidence scores and mean scores of the nine MM categories ($p > 0.05$), or between confidence scores and individual measures' mean scores, with the exception of MM3 (*Reduce nitrogen fertiliser N dose*; mean score was significantly higher within the "fairly sure" group than the "unsure" group; $p < 0.05$) and MM24 (*Optimise ground water levels*; a significant declining trend in mean score was observed with increasing confidence; $p < 0.05$).

Correlations between farm size characteristics (total farm area, total peat area, and total horticultural peat area on each farm (in hectares) and the proportion of each farm area cropped to horticulture on peat soils) revealed very few significant relationships with mean scores of individual or grouped measures. Only MM8 (*Increase on-farm planting of trees and shrubs*) showed a significant relationship with any of these variables, negatively correlating with farm area ($p < 0.01$, $R^2 = 0.82$) and total peat area ($p < 0.05$, $R^2 = 0.54$); an almost-significant correlation was also observed with horticultural peat area ($p = 0.09$, $R^2 = 0.36$). A number of other measures exhibited almost-significant negative correlations with farm size variables: MM1 *Avoid additional draining and cultivation* (farm area, $p = 0.05$, $R^2 = 0.38$), MM2 *Restore cropland to native wetland* (farm area, $p = 0.10$, $R^2 = 0.27$; horticultural peat area, $p = 0.09$, $R^2 = 0.36$), and MM6 *Avoid growing artificially heated protected crops* (percentage of farm which is horticultural peat, $p = 0.09$, $R^2 = 0.36$). Measure 28 (*Apply nitrification inhibitors*) showed an almost-significant positive correlation with percentage of farm which is horticultural peat ($p = 0.09$, $R^2 = 0.36$).

4.4. Discussion

4.4.1. Utility of the Best-Worst Scaling approach

This study aimed to evaluate the potential effectiveness and practicality of implementation of a number of farm-scale MMs, with the potential to reduce GHG emissions and related SOC loss from Histosols under horticultural production. Best-Worst Scaling is a powerful method for eliciting respondents' opinions by gauging their extremes of preference, and was used here to rank GHG mitigation options on the basis of their perceived effectiveness and practicality. This is the first known application of this method to eliciting preferences for measures with the potential to mitigate GHG emissions explicitly from horticultural peat soils. Online completion of BWS surveys enabled a variety of respondents from a wide geographical area to complete the exercise in real time.

Prioritising MMs to reduce emissions from horticultural peatlands under UK conditions is currently challenging, given the scarcity of experimental evidence relating to both baseline total emissions under typical crop rotations (e.g. Evans *et al*., 2011; Worrall *et al*., 2011), and to potential abatement rates of specific measures compared to baseline emissions (e.g. Moran *et al*., 2008). While the scientific basis for mitigation may be more extensive in other countries, expert opinion in this study relied on combining fragmented experimental evidence with knowledge of UK conditions and their influence on measures' potential abatement rates. A wide range of experts were invited and participated in the survey ensuring that specialism bias was minimised. Consequently, only one MM (MM24, *Optimise ground water levels*) showed any significant deviation from the mean scores for different expert groups (section 4.3.6.2.). Likewise, the level of confidence in responses had no effect on most rankings, and the general level of confidence in the BWS survey round was relatively high overall. The higher level of confidence in ranking measures in the BWS round compared to the first round (7-30%) may indicate that BWS offers a more intuitive and cognitively manageable method than scaling exercises, as has been suggested in a direct comparison of BWS and monadic scaling methods by Jaeger *et al*. (2008). Overall, the relationship between expert demographic information and BWS mean scores suggest that the collective expert rankings provided in this study are sufficiently robust.

During the shortlisting survey round, 30 MMs were identified from an original list of 58 items as priority measures for inclusion in the BWS survey (section 4.3.2.). The greatest numbers of measures were retained from the nutrient management, fossil fuel consumption, and carbon sequestration categories, with only two or three measures retained from each of the other MM categories. Nutrient management, and to a lesser extent carbon sequestration, categories also featured prominently in recent expert assessments of GHG mitigation options for UK cropped agro-ecosystems (Jones *et al*., 2010; MacLeod *et al*., 2010; Moran *et al*., 2008; 2011), although only nutrient management MMs were shortlisted by expert panels as both effective and feasible for implementation. This indicates that some categories of measures may be effective when applied to a range of soils and agro-ecosystems, but also reinforces the value of researching mitigation of Histosol emissions separately, where direct soil CO_2 emissions are much greater than from mineral soils. Extrapolating observations from other studies in this case would omit a category of interventions which are highly relevant for peat soils.

4.4.2. Potential economic costs of mitigation

Three measures were perceived to be effective at reducing overall GHG emissions and practical to implement (section 4.3.5., Fig. 4.3.), and could be considered a reasonable starting point for future mitigation efforts. The cost-effectiveness of implementing measures similar to MM1 (*Avoid additional draining and cultivation*), MM3 (*Reduce nitrogen fertiliser dose*) and MM4 (*Improve timing of nitrogen fertiliser application*) on arable land has been estimated in a small number of UK-based studies based on costs for the year 2006 (Barnes *et al*., 2010; Jones *et al*., 2010; Moran *et al*., 2008; MacLeod *et al*., 2010). Avoiding additional drainage in principle carries zero cost, but the immediate opportunity cost of forfeited crops balanced with the cost of land drainage labour and infrastructure instalment could potentially yield a range of net costs depending on circumstances. If undertaken as active peatland restoration (categorised as a separate MM to *Avoid additional draining and cultivation* in this study), other studies have estimated costs at between c. £5 per t CO_2-e abated (Smith *et al*., 2008) and £27 to £68 per t CO_2-e abated (MacLeod *et al*., 2010). Depending on the exact mode of implementation, reducing N fertiliser dose may result in a small net saving of between -£50 to -£18 per t CO_2-e abated and a larger net cost of £104 to £154 per t CO_2-e abated (Jones *et al*., 2010; MacLeod *et al*., 2010; Moran *et al*., 2008, 2011). The cost-effectiveness of improving the timing of N application is estimated to vary between -£103 and £44 per t CO_2-e abated (Jones *et al*., 2010; Moran *et al*., 2008, 2011), or between £83 and £1068 per t CO_2-e abated if implemented using controlled-release fertilisers (MacLeod *et al*., 2010; Moran *et al*., 2008, 2011). The current cost-effectiveness of implementation on Histosols is likely to differ from these estimates according to a number of factors including baseline rates of fertiliser applied, and the predicted GHG abatement rate when compared with mineral soils. The considerable variation in costs of implementing the top-ranked three measures, suggest that growers may consider multiple potential options within a certain margin of cost, providing they have other incentives for changes in management practices. For example, farmers may be driven by additional factors such as social drivers (peer pressure, feelings of social responsibility) or compliance with other agri-environment or accreditation schemes (Barnes *et al.*, 2010). The presence of other incentives within our farmer cohort is supported by the fact that 23% of the measures that farmers stated they had already implemented, fell into this category. To enhance uptake rates, changes in policy or further financial or informational support for farmers, consultants and contractors may be required, especially where a net cost is incurred (Jones *et al.*, 2013).

4.4.3. Heterogeneity of responses within and between expert and farmer groups, and policy implications

Examination of the score distributions within each of the three most-preferred measures (Fig. 4.4.), demonstrates that opinion can be divided amongst both experts and farmers, even for high-ranking MMs. In this study, none of the top three mitigation measures attracted a high level of consensus from both groups of respondents, although a moderately high level of agreement was achieved for MM3 (*Reduce nitrogen fertiliser dose*). This MM might be considered a particular priority for inclusion in future GHG mitigation policy. With regards to other MMs, reasonably robust models for incentivising a range of MMs within heterogeneous agricultural sectors already exist in the UK, in the form of agri-environment schemes. These usually incorporate a range of compliance options for farmers, allowing a degree of customisation to most appropriately match farmers' particular circumstances (Rose, 2011). An emphasis on paying farmers for ecosystem services rather than purely for food production, is emerging in European policy, and may be a promising starting point in mitigating emissions from peatlands (Reed *et al.*, In Press).

This study has identified a strong disparity between MMs considered effective and those considered practical. Such measures either related to major changes of land use (MM2, *Restore cropland to native wetland vegetation*; MM11, *Convert cropland to grassland*), or required changes that would directly conflict with the current horticultural management practices (MM6, *Avoid growing artificially heated protected crops*; Fig. 4.1.). Lower-ranking but above-average scoring measures (Fig. 4.1.) may be characterised by their requirement for considerable changes to management or cropping practices (MM10, MM15, MM17, MM18, and MM19), use of little-used or unknown technologies (MM28), or MMs which may be technically impossible or require prohibitively large capital costs to enable implementation on some farms (MM24). It is possible there would be moderate to high costs associated with all of these measures. The relatively low rate of uptake of such measures in this study (15% of currently or previously implemented MMs, Table 4.4.) suggests that barriers to uptake may be substantial but do not appear to exist across the whole horticultural sector. Obstacles to uptake may vary between measures and cover a range of issues. For example, focus group studies by Barnes *et al*. (2010) and Jones *et al*. (2010) revealed that UK farmers resist implementing GHG MMs on a combination of structural, financial, educational, management and administrative grounds. Understanding which barriers are restricting horticultural growers on peat soils, and providing appropriate levels and types of additional support are key to the successful implementation of mitigation efforts. For example, psychological or attitudinal barriers to implementation may be as substantial, if not more so, than financial barriers in some cases. Encouraging uptake of MMs in these circumstances may require a more involved approach when compared to overcoming financial barriers, for example.

Measures considered the most practical to implement but having lower than average effectiveness were *Use more fuel-efficient machinery and equipment* (MM7), *Use frequent nutrient testing and management planning* (MM9), *Use precision farming* (MM14), and *Optimise irrigation system efficiency* (MM16; Fig. 4.2.). These are likely to be implementable immediately; with minimal labour costs; and without substantial capital investments, perceived changes in existing technology or land-use, or perceived negative effects on crop yields. Lower-ranking but above-average scoring measures also follow these criteria of relatively low-cost, familiar technologies (MM12, MM27, and MM29), while others imply access to financial and informational support already in place, perhaps from participation in other agri-environment schemes (MM22, MM23, and MM25). In this study, 62% of farmers stated they had already implemented measures within this latter category, indicating that a reasonable level of existing infrastructure or support is available for at least some of the measures. Future GHG abatement policies could incorporate these MMs using existing levels of support, subject to understanding the factors contributing to the current attractiveness of these options.

In this study, the highest level of consensus amongst both experts and farmers was for MMs considered both ineffective and impractical (Fig. 4.1. and 4.2.; section 4.3.5.). These measures should be considered low priority for inclusion in mitigation policy and support schemes. However, measures perceived as ineffective and impractical by the majority may still be considered practical by some farmers (one has already been implemented by a grower: MM5, *Minimise avoidable machinery use*), and may have been implemented for reasons other than GHG mitigation (in this case, possibly to save fuel costs). Examination of current agri-environment and other support schemes incorporating measures that also mitigate GHG emissions (even minimally) could aid initial identification of the easiest means of encouraging uptake.

4.4.4. Wider evaluation of mitigation potential

Any complete assessment of a suitable GHG mitigation action plan would include a number of factors not covered in this study. Firstly, a complete life cycle assessment or carbon footprint of each MM for a range of scenarios (mode of implementation, farm characteristics, variation in inputs) would be advised. Following this, the efficacy and cost of single and combined measures should be computed, taking account of any compatibilities and conflicts between individual MMs. Guidance is already available for implementing this type of analysis (e.g. Moran *et al.*, 2008; 2011), although further specific information would be required for horticultural peat systems, which would ideally be provided through experimental evidence. Finally, further consultation with farmers would aid knowledge of the current structure of the horticultural industry, and of potential uptake incentives and obstacles to mitigation implementation. Active engagement with farmers could form the basis of an initial framework for adaptive policy and support schemes, with future changes made to enable the most effective MMs to be implemented as further experimental evidence came to light (e.g. new technologies). Building incentivisation schemes around existing agri-environment scheme models, comprising a number of compliance options and tiers of payment for priority measures, would allow farmers to choose from a suite of measures prioritised by effectiveness, suitable for that particular farm. Policy could be re-evaluated periodically according to changes in external factors which constrain farmer decisions (e.g. the market for horticultural products; the value of peatland).

A further important wider consideration is the trade-off between private costs to the farmer, and societal cost-effectiveness. While re-wetting peatlands might result in a loss of livelihood for individual farmers, its potential low cost of implementation and potential high GHG mitigation potential may provide a more cost-effective alternative than other publicly-funded

GHG mitigation measures (e.g. renewable energy subsidies). Assessment of the relative private and public cost-effectiveness of a broader category of tax-funded MMs would aid prioritisation of policy objectives contributing to reducing overall UK GHG emissions, whilst considering the potential positive and negative impacts on other ecosystem services. For example, this might include considering additional wildlife or recreational benefits from re-wetting peatland, balanced against the issue of displacing crop production abroad or to UK mineral soils with lower productivity, with its potential consequences for food security and emissions.

4.5. Conclusions

This study set out to evaluate the perceived efficacy and practicality of interventions with the potential to reduce net GHG emissions from horticultural peat soils, using the Best-Worst Scaling method. The results illustrate that the approach to mitigation on UK peat soils will need to be multi-faceted, since no single category of mitigation measures wholly fulfils the criteria of effectiveness at reducing net GHG emissions, or practicality of application. Tensions currently exist between interventions perceived as effective and those perceived as practical. Considerable efforts are required to refine the experimental knowledge base around mitigating emissions from horticultural peat soils, as well as understanding the motivations of growers and obstacles to mitigation specific to the horticultural industry. Despite the relatively small area of UK agricultural land under horticultural production, emissions from horticultural peat soils are significant at the national scale. Given the additional importance of peatlands as a soil carbon stock and highly productive substrate for crop production, GHG emission abatement is important. Implementation of flexible, adaptive policies would allow optimisation of mitigation efforts as the UK evidence base strengthens and with changing circumstances over time.

Chapter 5

Efficacy of mitigation measures for reducing greenhouse gases emissions from horticultural peat soils

Abstract

Modifying water table management, tillage, and crop residue and horticultural fleece application practices may offer effective approaches for mitigating net GHG emissions from horticultural peat soils. Evidence in the literature on the efficacy of these measures when applied to cultivated peatlands is incomplete, and requires clarification. This study evaluated the GHG mitigation potential of measures that varied water table depth, and tillage, crop residue application and horticultural fleece application practices. Emissions of CO_2, N_2O and CH_4 from intact soil columns were intensively monitored from May to November 2013, using a closed chamber method. Concurrent measurements of soil physical and chemical characteristics allowed identification of the key controls on variation in emissions. Raising the water table to soil surface level provided the strongest reduction in GWP, at 2.06 ± 0.46 t CO_2-e ha^{-1} $80d^{-1}$, compared to a free-draining control (6.45 ± 0.11 t CO_2-e ha^{-1} $80d^{-1}$), but this was partially negated by the observed pulse of emissions which followed draining of the raised water table. The highest emissions were from cores with a water table level at 15 cm below the soil surface, proving GWP to be highly sensitive to water table variation near the soil surface. Current practice in horticultural fleece application during spring appears to have no significant effect on GWP, but prolonged fleece application could significantly elevate emissions. It is recommended that mitigation measures be applied during the growing season, when they are likely to produce the strongest mitigative effect.

5.1. Introduction

Drained and cultivated lowland peats (Histosols) represent some of Europe's most productive agricultural soils. Their management is highly problematic due to the potential for soil loss either from wind erosion or from microbial mineralisation of the peat substrate (Dawson and Smith, 2007). Whilst this microbial activity results in the release of nutrients previously locked up in soil organic matter (SOM), enhancing crop productivity, it also progressively diminishes the resource base (Cannell *et al*., 1999). There is an ecosystem services trade-off between (1) preserving (and enhancing) peat C storage for climate change mitigation, maintaining high biodiversity habitats, and improving water quality, and (2) using this resource to promote food security.

In the UK, peat soils comprise only 18.3% of the total soil area (Montanarella *et al*., 2006), but contain a disproportionately large amount of the UK's soil C stock (c. 53%; Dawson and Smith, 2007). Evidence suggests that potential greenhouse gas (GHG) emissions and C losses from cultivated peats may be large, with an estimated 39% of English deep fen peats currently under cultivation and classed as being at risk from severe soil loss (Natural England, 2010). The depth of cultivated lowland peats has been declining by an estimated 0.27-3.09 cm soil yr^{-1} since c. 1850 (Richardson and Smith, 1977; Hutchinson, 1980), with a recent estimate from East Anglian arable fens of 1.10-1.48 cm soil yr^{-1} between 1982 and 2004 (Dawson *et al*., 2010). Although estimates of GHG emissions from arable peat soils are scarce (Evans *et al*., 2011; Worrall *et al*., 2011), it is possible that 35% to 100% of this soil loss may be attributable to microbially mediated CO_2 production (Leifeld *et al*., 2011). The small net consumption of CH_4 in these soils does little to offset CO_2 loss, whilst N_2O emissions can be substantial, forming approximately one third to one half of the total GHG budget from these soils (Taft *et al*., 2013). Mitigating GHG emissions could contribute substantially towards a reduction in the C footprint of agricultural production (UK Parliament, 2008).

GHG emissions from agricultural soils are influenced by a large number of interacting drivers, including soil (e.g. moisture, temperature, porosity, substrate availability), climate (rainfall, temperature), and vegetation (yield, water uptake) factors, which in turn are driven by human activities such as farm operations (Li, 2007). Often, a change in a single factor may simultaneously increase emissions of one GHG and result in the reduction of another (Smith *et al*., 2008). Therefore, mitigation studies should consider the overall effect of a measure on the total emissions of CO_2, CH_4 and N_2O, rather than on a single GHG, as in some previous studies (e.g. Dalal *et al*., 2008; Henault *et al*., 2012; Lal, 2008). This is particularly important where measures reducing CO_2 emission increase efflux of the more radiatively powerful CH_4 and N_2O, causing a disproportionately large increase in the overall global warming potential (GWP) of the system. Given the relationship between GHG efflux and soil organic C (SOC) loss (Dawson & Smith, 2007), and the importance of SOC to long-term soil sustainability, it is also useful for mitigation studies to include an estimate of the effects of treatments on SOC retention.

Significant gaps remain in the literature regarding the effects of potentially mitigative management practices on arable peat soils. Many review papers apply to arable systems and contain few interventions specific to Histosols, while many experimental studies, investigating specific GHG mitigation measures, were carried out in geographical areas with very different climatic, cropping characteristics or risks (e.g. peat loss from fire) in comparison to the UK (e.g. Couwenberg *et al*., 2010). Much of the evidence remains inconclusive, largely because only a very small number of studies have been undertaken on arable or horticultural peat soils, for example regarding the effects of water table variation (e.g. Kasimir-Klemedtsson *et al*., 1997; Kechavarzi *et al*., 2007; Nykänen *et al*., 1998), cultivation and residue management (e.g. Elder and Lal, 2008), soil pH (Nykänen *et al*., 1998), and specific crops or land uses (e.g. Flessa *et al*., 1998; Weslien *et al*., 2012). This study investigates the effect of changes in common management operations with the potential to be incorporated into current crop rotations, and for which little information was available in the literature.

The overall aim of this study was to test the potential of a small number of shortlisted mitigations to reduce emissions of CO_2, CH_4 and N_2O and promote SOC retention, on soils of comparatively high organic matter content (c. 70% SOM), from a site managed under intensive commercial horticulture. The variables tested were: (1) Water table depth, (2) Horticultural fleece application, (3) Cultivation, and (4) Crop residue management (surface application, incorporation).

5.2. Methods and materials

5.2.1. Study sites

Soils (Sapric Histosols; FAO, 2006) utilised in this study originate from East Anglia, UK. East Anglia has a mean annual rainfall of < 700 mm, a mean annual temperature of 10.2°C (ranging from mean 4.2°C in winter to 17.2°C in summer), and mean annual sunshine hours of 1550 (UK MetOffice, 2014). The study area comprises drained lowland fen typified by flat topography, which is under intensive commercial scale horticultural and arable production, growing primarily vegetables (including lettuces, potatoes, leeks, onions, red beet, and celery), sometimes in rotation with cereals (primarily wheat). Soil was collected from a field which had been under a typical rotation for the previous growing season, in order to be representative of the soils from these farming systems. Table 5.2. shows the physical and chemical characteristics of the soils used in the experiments.

5.2.2. Field sampling

Intact cores were taken on 25th – 26th October 2012 from S70 field site (Chapter 2), from a visually representative area of c. 10 m^2 of a field to minimise any microsite variability caused by soil heterogeneity, after Dinsmore *et al.* (2010). A PVC soil pipe ($d_{internal}$ = 103 mm; h = 400 mm) with a chamfered base was driven into the soil to give a final core depth of 300 mm with c. 100 mm remaining at the top of the core to act as chamber headspace when gas sampling. The PVC tubes were inserted into bare soil free from roots and above-ground biomass using a similar method to Freeman *et al*. (1993), to minimise disturbance to the soil profile. Briefly, the core was aligned perpendicular to the ground, pushing in the core as far as possible (generally, 3-5 cm); subsequently, small trenches were excavated around the next level down and the tube hammered onto the lower soil section, avoiding gaps between the soil column and the inside of the tube. Upon completion, a small spade was slid under the base of the column and soil levelled. Each core was covered at the base and top with a plastic bag and taped for storage to minimise moisture and soil loss. Cores were stored upright in boxes at ambient low temperature (c. 10°C) in the dark until transportation back to the experimental site at Bangor University on 28th October.

5.2.3. Preliminary soils analysis

Five additional cores were taken from the field and a number of chemical and physical analyses performed before commencement of the experiment; the same analyses were conducted at the end of the experiment on all experimental cores (Table 5.2.).

Cores were split into three layers of 0-10 cm, 10-20 cm and 20-30 cm depth, and each layer analysed separately. A Rhizon® suction sampler was inserted to 10 cm depth for 30 minutes and a soil water sample obtained then stored at c. -20°C. Next, a soil sample was taken using a bulk density ring (h_{total} = 10 cm, V_{total} = 200 cm^3) for calculation of soil gravimetric moisture content and bulk density by placing in an oven at 105°C for ≥ 24 h. The remaining soil from that layer was homogenised and subsampled for soil chemical analysis within 48 h of sampling after storing at c. 4°C. Soil samples were weighed out in triplicate for each layer and extracts obtained for the separate analysis of available NO_3^- and NH_4^+ (5 g soil in 25 mL 0.5 M KCl), available P (5 g soil in 25 mL 0.5 M acetic acid), and available K (5 g soil in 25 mL 1 M NH_4Cl). Extracts were obtained by shaking the soil suspensions at 200 rev min^{-1} for 30 min, centrifuging at 4000 rpm, then filtering using a Whatman 42 filter paper. Filtrates were stored at c. -20°C until further analysis. Available soil NO_3^-, NH_4^+ and P were determined by microplate analysis (PowerWave XS Microplate Spectrophotometer, Biotek Intruments, Inc., USA) using methods similar to those outlined in Mulvaney (1996), Miranda *et al*. (2001), and Murphy and Riley (1962) respectively. Flame photometry (Model 410 flame photometer, Sherwood Scientific Ltd., UK) was used to determine available soil K, after Sherwood (2013).

5.2.4. Core storage and acclimation

Cores were stored in an unheated glasshouse for c. 6 months after extraction. Glasshouse air temperature was monitored using automatic loggers (iButton DS1921G-F5 thermochrons, HomeChip Ltd., Milton Keynes, UK) and compared to average temperatures for the study area (UK Met Office, 2014); there were no significant differences between either average day- or night time temperatures between the storage and field sites over the storage period. Water loss from the base and top of the core was minimised by retaining a perforated plastic covering, which also allowed gaseous exchange between the soil surface and atmosphere (O_2 and CO_2 concentrations measured on 12th – 13th February 2013 using a GFM400 Series Gas Analyser (Gas Data Ltd., Coventry, UK) were within the expected range for aerobic activity within the soil). A consistent soil moisture content was maintained by periodically weighing cores and when necessary, pipetting distilled water evenly across the soil surface.

On 25th April, cores were moved outdoors and plastic covers removed to allow acclimation to ambient conditions. Cores were randomly laid out in sets of four in free-draining buckets, and greenhouse gas measurements were made using the method outlined in section 5.2.5.2.1. to determine a baseline emission so that cores could be randomised for the final experimental design. No significant differences were found between mean background emissions of CO_2, CH_4 or N_2O from any of the cores. Ibutton automatic temperature loggers were set up near the cores in a sheltered position to monitor air temperature; other environmental variables (rainfall, wind speed, sunlight hours, etc.) were assumed to be similar to those at the nearest Met Office monitoring stations to the Bangor experimental site.

5.2.5. Experiment Phase I

5.2.5.1. Treatment application

After background emissions had been established, cores were assigned treatments (Table 5.1.) and packed singly into buckets (Appendix E). Each core base was secured with a fine permeable mesh to allow for water penetration whilst minimising soil loss from the intact core base. All cores not subject to a raised water table treatment were also placed into a shallow covered plastic tray connected to a tube and bung, from which water runoff could be collected over the course of the experiment. All buckets had effectively free draining bases except for the raised water table treatments, which were drilled with holes around the bucket sides at the appropriate water table level to allowing drainage of excess water and maintenance of the correct water table height. The tops of all cores were left open to ambient conditions throughout the experiment. After core assembly in the centre of each bucket, sterile fine sand was packed around each core casing until level with the soil surface, to simulate field conditions (in particular, to act as a temperature buffer). Mesocosms were laid out in a randomised block design with five replicates of each of the seven treatments, with blocks aligned to the prevailing wind direction (SW-NE) to account for differences in sheltering and evapotranspiration across replicates.

Treatments were all applied on 27th May. Soil water table treatments were established by filling selected buckets with artificial rainwater until the excess ran out of the lateral drainage holes. Subsequently, water table height was maintained with natural or artificial rainfall. Plastic covers were applied to the sand surrounding WH and WL treatment core casings on 1st July to minimise evaporation during hot weather conditions.

White horticultural fleece was secured over the top of the core headspace using plastic-coated wire.

Cultivation treatments were based on the typical ploughing depth at the field site (c. 30-35 cm), and were implemented by removing the whole volume of soil from the core, mixing in residues where appropriate, and packing loosely back into the core.

Soil residue treatments were assigned according to the typical mean percentage of Iceberg lettuce residue left on the cropped area of the field post-harvest (52% including plants gone to waste, G’s Fresh, unpubl. data), assuming an even distribution over the cropped area. Lettuces for residue simulation were purchased from a supermarket and stored at c. 4°C until further use. Residues were torn into approximately equal sized pieces (c. 5 x 5 cm) to fit into the core headspace, placed into the core headspace, and pressed into the soil surface to simulate post-harvest tractor traffic. Fresh sub-samples were analysed for moisture content by drying at 80°C to reach constant dry weight, and for C and N content using a CHN2000 analyser (Leco Corp., St Joseph, MI, USA).

Table 5.1. Summary of the water table, fleece, cultivation, and residue treatment characteristics used in the experiment.

Treatment name (code)	**Water table depth (cm)**	**Lettuce biomass ($g\ cm^{-2}$ / $t\ ha^{-1}$)**	**Cultivation (cm)**	**Fleece**
Control (C1)	> 30 cm (free-draining)	None	None	None
High water table (WH)	0 cm (at soil surface)	None	None	None
Low water table (WL)	15 cm below soil surface	None	None	None
Fleece (F)	> 30 cm (free-draining)	None	None	Fleece applied
Cultivated (C2)	> 30 cm (free-draining)	None	To 30 cm depth	None
Residue, surface applied (R)	> 30 cm (free-draining)	35.5 $g\ cm^{-2}$ / 29.7 $t\ ha^{-1}$	None	None
Residue, incorporated (RC)	> 30 cm (free-draining)	35.5 $g\ cm^{-2}$ / 29.7 $t\ ha^{-1}$	To 30 cm depth	None

5.2.5.2. *Mesocosm measurements*

Field measurements were performed in a sequence of least invasive of the peat substrate to most invasive, to minimise the effect of other experimental procedures on GHG emissions. Measurements were made for seven consecutive days following treatment application, then twice per week for two weeks, then approximately once per week thereafter.

5.2.5.2.1. *Monitoring fluxes of GHGs using closed static chambers*

Closed, non-vented static chambers were used to measure emissions of CH_4 and N_2O, between the hours of 09:00 and 17:30 each day. On each sampling date during the first phase of the experiment (28th May to 16th August), GHG sampling was conducted in three consecutive stages on sets of randomly selected replicate blocks, to minimise any time effect of sampling on emissions estimates. Thus, GHGs were measured from chambers in two blocks, followed by a further two blocks, followed by the remaining single block.

The optimal chamber size for GHG detection was identified prior to the experiment (data not shown). Identical, white opaque polypropylene cylindrical chambers (internal dimensions d_1 = 10.0 cm, d_2 = 2.2 cm, h = 12.0 cm), with a rubber septum sampling port in the lid of the chamber, were used for gas sampling. Chambers were inserted into the top of the core casing with the sampling needle inserted into the rubber septum and the side of the chamber slightly pressed inwards for pressure equalisation, then released so that the flexible seal around the chamber rim formed a seal with the tube, giving a final enclosed headspace volume of 1.25 to 2.19 dm^3 (depending on core headspace depth). Electrical tape was then secured around the join to ensure air-tightness.

Prior to sampling, an ambient gas sample was taken, and temperature loggers were randomly placed face-up inside eight of the cores to monitor within-chamber temperatures compared to ambient temperature. No significant heating effect was found within chambers compared to ambient temperature ($p < 0.05$; data not shown). Each chamber was attached immediately before taking the first gas sample ($t = t_0$). Gas sampling and storage procedures and materials followed those described in Chapter 2.

5.2.5.2.2. *Monitoring soil CO_2 emission using the IRGA*

After taking chamber measurements, soil respiration was estimated using an infra-red gas analyser (IRGA, PP Systems, UK) equipped with an automatic SRC-1 soil respiration chamber (internal dimensions d = 10.35 cm, h = 6.50 cm; total volume = 1171 mL) attached. Measurements were taken between the hours of 09:00 and 23:00.

5.2.5.2.3. *Soil and weather measurements*

On each sampling date, a number of additional non-destructive measurements were made on the cores.

Immediately after GHG sampling from each block of chambers, soil temperature (0-10 cm) was measured using a Checktemp1® temperature probe (accurate to ±0.3°C; Hanna Instruments Ltd, Leighton Buzzard, UK), inserted into the centre of each core. After measuring, the hole was gently re-filled to prevent GHG channelling from lower soil layers during subsequent GHG measurements.

A Rhizon® with a 5 cm porous section (Rhizosphere Research Products B. V., The Netherlands) was inserted into the soil surface of each mesocosm immediately after the application of treatments, to a depth of 10 cm and c. 2 cm from the core casing. Thereafter, soil water samples were extracted on each sampling date following soil temperature measurements, and the Rhizon cap replaced to avoid subsequent interference with GHG estimation. When available, runoff samples were taken from the lysimeter tube at the base of each core. Soil water and runoff samples were stored at -20°C until further analysis for available nitrate and ammonium.

Meteorological data (daily rainfall, daily air temperature) for North Wales and East Anglia stations were obtained from the UK Met Office via the BADC (BADC, 2014), to allow comparison of mean weather variables during the field collection, acclimation and experimental periods.

Samples of rainwater were collected from a collection bucket in the same area as the experimental buckets, periodically through the course of the experiment. Additionally, a subsample of each batch of artificial rainwater was taken. Both were stored at ~ -20°C until analysis for nitrate and ammonium content.

5.2.6. Experiment Phase I

5.2.6.1. Treatment application: water table drainage

By 16th August, differences in GHG emission between control and single-event treatments (C2, R, RC) were negligible, so the next phase of the experiment was implemented. The second stage of experiment aimed to monitor the effect of lowering the water table on emissions, and involved only the C1, WH and WL treatments. The buckets containing WH and WL cores were drained on 20th August one randomly selected block at a time. Holes were drilled in the sides of the bucket base to allow all of the water to drain away (c. 30-45 minutes). Within that block, each core was then deconstructed and re-packed in sequence. First, the protective cover was removed from the sand surrounding the core and the sand was unpacked and core removed. Soil surface redox potential was measured using an Eijkelkamp BNC glass Platinum electrode with an Ag/AgCl reference electrode and 3 M KCl electrolyte (Eijkelkamp Agrisearch Equipment, The Netherlands), and corrected for soil temperature at the same depth (1-2 cm), using the method outlined in Eijkelkamp (2009). Two 15 mm holes were drilled in the back of the core casing (at -10 cm and -20 cm below the soil surface respectively), and the soil redox potential immediately measured by inserting the probe into the hole; soil temperature was then measured at approximately the same position in the soil. Finally the hole was sealed to exclude air using strong waterproof adhesive tape. After measuring the redox potential at 0 cm, -10 cm, -20 cm and -30 cm for each core, the bucket base was drilled with holes and the core replaced over a fine mesh base to minimise soil loss. Sand was repacked into a fully permeable cloth sack surrounding the core, until level with the soil surface. This process was repeated for all three cores in the block, before repeating for the remaining blocks.

5.2.6.2. Mesocosm monitoring

Monitoring processes during the second phase of the experiment (21st August to 13th November) were identical to that of the first phase, with a small number of modifications. The C2, R, RC and F treatments were not monitored during this period, but left in position until the end of the experiment. Chamber sampling was conducted for all fifteen C1, WL and WH cores simultaneously, from a randomised corner of the experiment. Soil redox potential and soil temperature measurements were made at each soil depth, before soil temperature (0-10 cm) measurements and Rhizon soil water sampling.

5.2.7. Post-experiment measurements

By 13th November, observable differences in GHG emissions between C1, WH and WL treatments were negligible. One final set of GHG emission, soil temperature, redox potential, and Rhizon soil water measurements were taken on 12th-13th November 2013, for all 35 cores. Cores were dismantled on 14th-16th November by block in a randomised order, by removing each core from the surrounding sand, covering the base to minimise soil loss, and transporting to the laboratory. Any remaining runoff from lysimeters was collected and stored at -20°C until further analysis for nitrate and ammonium content. In the laboratory, each peat core was split into 10 cm depth fractions and analysed as outlined in section 5.2.3. and Table 5.2.

5.2.8. Data cleaning and statistical analysis

5.2.8.1. Data cleaning procedure

Flux calculation and data cleaning procedures for chamber data (N_2O and CH_4) and IRGA data (CO_2) were identical to those described in Chapter 2.

5.2.8.2. Statistical analysis

For each sampling date, the flux mean and standard error were calculated separately for each treatment, to give values in mg CO_2-C m^{-2} h^{-1}, µg N_2O-N m^{-2} h^{-1} and µg CH_4-C m^{-2} h^{-1} respectively.

Mean cumulative fluxes over the whole measurement period were calculated separately for each core by multiplying hourly values by 24 to give a daily flux, followed by stepwise interpolation of the flux values on known dates, then summing the resulting values over the required period. Cumulative standard errors for each treatment were taken as the standard error of the cumulative means of individual cores within that treatment. Owing to some missing data points, cumulative fluxes were calculated for the periods 29th May to 16th August inclusive for all treatments, and 21st August to 28th October after drainage for the water table treatments. All cumulative flux estimates were converted to 100-year global warming potential (GWP_{100}) CO_2 equivalents (CO_2-e) according to the methodology outlined in IPCC (2000), allowing comparison between treatments for total GWP and individual GHG GWPs, and comparison with UK GHG Inventory emissions EFs (Webb *et al*., 2014).

Statistical analyses were performed using SPSS v. 20 (IBM, Inc.), with significance being accepted at $p \leq 0.05$ except where stated otherwise. All data values are quoted as mean ± standard error unless stated otherwise. All statistical analyses were performed separately on the water table group of treatments (C1 vs WL vs WH), the fleece treatment (C1 vs F), and the cultivation and residue group of treatments (C1 vs C2 vs C vs RC). Normality was tested using the Shapiro-Wilk test (Field, 2005), and non-normal data were log-transformed or square-root transformed; where transformation was ineffective, or where heterogeneity of variances was observed (Levene's or Welch's test statistic), appropriate non-parametric tests were used to compare medians of those data groups. Soil physical and chemical characteristics for each soil depth layer were compared using ANOVA or the independent t-test, or Kruskall-Wallis or Kologorov-Smirnoz Z tests for data deviating greatly from normality or homogeneity of variances. Significant effects of treatment and time (each treatment including the control C1, compared to the baseline) were tested.

Cumulative fluxes of CO_2, N_2O, CH_4 and total GWP_{100} for each treatment were compared using ANOVA, independent t-test, Kruskal-Wallis or Kologorov-Smirnoz Z tests as appropriate to the data. Post-Hoc tests were conducted to determine significantly different treatments using Tukey's HSD, Gambrell-Howell, or Kologorov-Smirnoz Z statistics (with the Bonferroni correction for multiple comparisons).

Relationships between individual GHGs, temperature, rainfall, and soil N concentrations were explored using Kendall's tau statistic (τ), a powerful correlation statistic for the non-normal distribution of these variables.

5.3. Results

5.3.1. Soil physical and chemical characteristics

5.3.1.1. Soil characteristics before and after treatment application

Some significant differences between treatments and the baseline treatment were observed within all soil layers for all soil characteristics, with the exception of available potassium (Table 5.2.). By the end of the experiment, soil moisture content (GMC%) was significantly higher in the upper soil layers of all treatments except for the cultivation (C2) treatment ($p < 0.05$ to $p < 0.001$), with differences becoming smaller with soil depth. Bulk density in the upper part of the cores had significantly increased in the control (C1, $p < 0.05$), water table (WL, WH; $p < 0.01$), and surface residue treatments (R, $p < 0.05$), and had significantly declined in the cultivated treatments (C2, RC; $p < 0.001$); again, the effect declined with depth. Fleece application had no effect on bulk density. The effect on soil pH was more variable with depth, although generally, soil pH increased slightly at all depths in all treatments between May and November. Electrical conductivity (EC) was lower at all depths in all treatments by November, with significant differences being observed at all depths for C2, F and R treatments, and at the 20-30 cm depth only for all other treatments ($p < 0.05$ to $p < 0.001$). Few significant effects of treatment on available phosphorus were found, but available P was significantly lower in the upper peat layers in C1, C2, WL and WH treatments ($p < 0.05$ to $p < 0.01$). Available nitrate and ammonium were influenced by most treatment applications in most soil layers. Soil nitrate was significantly lower in the top 20 cm soil for all treatments except the fleece and incorporated residue treatments ($p < 0.05$), while only the WH treatment showed a significantly lower nitrate level in the 20-30 cm layer ($p < 0.05$). Available ammonium was also significantly lower in most soil layers and in all treatments than at the start of the experiment ($p < 0.05$).

5.3.1.2. Treatment effects on soil characteristics

Within each soil layer at the end of the experiment, no significant differences were found between water table treatments (C1, WL, WH) for any of the soil characteristics ($p > 0.05$), except that soil moisture content was significantly higher in both of the water table treatments WL and WH in the 0-10 cm soil layer when compared to the control (Tukey's HSD $p < 0.01$ and $p < 0.001$ respectively), and nitrate content was significantly lower in the WL and WH treatments in the 20-30 cm soil layer than the control (Tukey's HSD; both $p < 0.01$).

No significant differences were found between fleece and control treatments for most of the soil characteristics ($p > 0.05$). However, pH was significantly lower under the F treatment than the control in the 10-20 cm and 20-30 cm layers ($t(8) = -3.020$, $p < 0.05$, and $t(8) = 3.892$, $p < 0.01$ respectively), fleece EC was significantly higher in the 20-30 cm layer ($t(8) = -7.773$, $p < 0.001$), fleece phosphate was significantly higher in the 0-10 cm and 20-30 cm layers ($t(8) = -2.761$, $p < 0.05$; and $t(8) = -2.635$, $p < 0.05$ respectively), and fleece nitrate was significantly higher in the 10-20 cm and 20-30 cm soil layers ($t(8) = -2.546$, $p < 0.05$; and $t(5.235) = -3.805$, $p < 0.05$ respectively).

Within each soil layer at the end of the experiment, no significant differences were found between C1 and C2, R or RC treatments for most of the soil characteristics, with the exception of bulk density, EC, and available nitrate. Bulk density was significantly lower in the 0-10 cm and 10-20 cm layer in the cultivated treatment (Tukey's HSD; $p < 0.001$) and in the incorporated residue treatment (Tukey's HSD; $p < 0.001$) than the control, and in the 20-30 cm layer in the cultivated treatment (Tukey's HSD; $p < 0.05$). Additionally, bulk density was significantly higher in the top two soil layers in the surface-applied residue than both the ploughed and incorporated residue treatments (Tukey's HSD; all $p < 0.001$), and in the base layer between R and C2 treatments (Tukey's HSD; $p = 0.025$). Soil EC was significantly lower in the 10-20 cm layer in the ploughed treatment than the control (Tukey's HSD; $p < 0.05$) and in the 20-30 cm layer for the surface-applied residue treatment (Tukey's HSD; $p < 0.05$). Significant differences in nitrate levels were only found in the 10-20 cm layer between the incorporated residue and control treatments (lower; Tukey's HSD; $p < 0.05$), and in the 20-30cm layer between surface-applied residue and control treatments (lower; Tukey's HSD; $p < 0.05$).

5.3.2. Weather conditions

The mean air temperature for the wetted period 28th May to 16th August was 15.4°C, with the highest temperatures observed in July (Fig. 5.1.a., 5.3.a., 5.4.a., 5.5.a.). Total cumulative rainfall for this period was 191 mm, with c. 50% of rainfall observed between 24th July and 6th August (Fig. 5.1.a., 5.3.a., 5.4.a., 5.5.a.). During the drained period 21st August to 13th November, mean MIDAS air temperature was 13.2°C (Fig. 5.1.a.), with peak air temperature observed at the start of this period and declining to a mean of 8.8°C during November. Total cumulative rainfall during the drained period was 229 mm, with c. 50% of total rainfall observed during October (Fig. 5.1.a.).

Table 5.2. Soil characteristics of cores sampled at the S70 farm site at the start and end of the experimental period.

Treatment, depth	**Mean core depth (cm)**	**GMC (% DW)**	**Bulk density (g cm^{-3})**	**pH (H$_2$O)^1**	**EC (µS cm^{-1})^1**	**Available K (g K kg^{-1})**	**Available P (g P kg^{-1})**	**Available NO$_3^-$ (g N kg^{-1})**	**Available NH$_4^+$ (g N kg^{-1})**
Initial									
0-10cm		151.6 ± 0.6	0.68 ± 0.01	6.2 ± 0.08	598.2 ± 49.9	0.96 ± 0.21	0.39 ± 0.01	0.15 ± 0.016	0.05 ± 0.0237
10-20 cm		156.1 ± 1.6	0.76 ± 0.02	6.2 ± 0.06	551.8 ± 48.8	0.63 ± 0.11	0.38 ± 0.01	0.15 ± 0.033	0.04 ± 0.0080
20-30cm		163.1 ± 4.6	0.75 ± 0.02	6.3 ± 0.06	401.2 ± 23.7	0.56 ± 0.11	0.35 ± 0.02	0.13 ± 0.033	0.03 ± 0.0014
Post-expt									
C1	30.5 ± 0.1								
0-10cm		164.0 ± 0.7†	0.73 ± 0.01*	6.7 ± 0.04†	160.6 ± 12.6	0.54 ± 0.08	0.27 ± 0.02†	0.01 ± 0.001†	0.01 ± 0.0008†
10-20 cm		168.4 ± 1.8**	0.77 ± 0.01	6.7 ± 0.06***	166.0 ± 8.1	0.51 ± 0.19	0.27 ± 0.01**	0.03 ± 0.004†	0.01 ± 0.0012
20-30cm		180.0 ± 2.1	0.75 ± 0.01	6.7 ± 0.04*	220.4 ± 8.5***	0.58 ± 0.15	0.21 ± 0.04	0.06 ± 0.008	0.01 ± 0.0004†
WL	29.9 ± 0.1								
0-10cm		169.8 ± 0.9†	0.74 ± 0.01**	6.7 ± 0.04†	136.2 ± 3.2	0.63 ± 0.08	0.29 ± 0.02†	0.01 ± 0.001†	0.01 ± 0.0005†
10-20 cm		170.7 ± 1.5***	0.78 ± 0.01	6.7 ± 0.03***	160.1 ± 5.6	0.50 ± 0.13	0.31 ± 0.02	0.02 ± 0.001†	0.005 ± 0.0007†
20-30cm		175.3 ± 6.3	0.75 ± 0.01	6.7 ± 0.03*	223.0 ± 11.0***	0.44 ± 0.10	0.26 ± 0.04	0.03 ± 0.006	0.01 ± 0.0005†
WH	30.5 ± 0.3								
0-10cm		171.9 ± 1.2†	0.74 ± 0.01**	6.7 ± 0.03†	159.0 ± 8.3	0.61 ± 0.16	0.27 ± 0.01†	0.01 ± 0.001†	0.01 ± 0.0006†
10-20 cm		169.3 ± 2.6**	0.78 ± 0.02	6.8 ± 0.07***	176.0 ± 17.1	0.62 ± 0.16	0.27 ± 0.01**	0.02 ± 0.001†	0.01 ± 0.0004†
20-30cm		174.4 ± 4.8	0.77 ± 0.01	6.7 ± 0.06**	196.0 ± 16.4***	0.49 ± 0.17	0.33 ± 0.04	0.02 ± 0.003†	0.01 ± 0.0045
F	30.3 ± 0.2								
0-10cm		161.2 ± 2.4†	0.73 ± 0.01	6.6 ± 0.05†	154.1 ± 8.8†	0.42 ± 0.07	0.35 ± 0.03	0.01 ± 0.001	0.01 ± 0.0018†
10-20 cm		166.2 ± 2.5*	0.76 ± 0.01	6.4 ± 0.05*	204.7 ± 20.2†	0.45 ± 0.12	0.31 ± 0.01	0.04 ± 0.006	0.01 ± 0.0008†
20-30cm		174.9 ± 5.2	0.76 ± 0.01	6.4 ± 0.05	321.4 ± 9.9**	0.42 ± 0.11	0.31 ± 0.02	0.10 ± 0.003	0.01 ± 0.0004†
C2	34.9 ± 0.3								
0-10cm		158.4 ± 1.6	0.62 ± 0.01***	6.7 ± 0.08	133.4 ± 13.1†	0.49 ± 0.08	0.31 ± 0.01†	0.01 ± 0.001†	0.01 ± 0.0041
10-20 cm		166.0 ± 1.8	0.65 ± 0.02***	6.6 ± 0.07***	140.2 ± 6.5†	0.55 ± 0.09	0.30 ± 0.03	0.02 ± 0.002†	0.01 ± 0.0003†
20-30cm		175.2 ± 2.3	0.69 ± 0.02	6.5 ± 0.08	183.5 13.2***	0.61 ± 0.14	0.33 ± 0.02	0.04 ± 0.006	0.01 ± 0.0016†
R	29.7 ± 0.3								
0-10cm		163.8 ± 2.4†	0.76 ± 0.02***	6.7 ± 0.03†	138.5 ± 2.4†	0.59 ± 0.03	0.30 ± 0.02	0.01 ± 0.001†	0.005 ± 0.0004†
10-20 cm		163.8 ± 1.1	0.76 ± 0.01	6.7 ± 0.04***	149.0 ± 6.3†	0.49 ± 0.10	0.32 ± 0.01	0.02 ± 0.001†	0.01 ± 0.0006†
20-30cm		165.3 ± 5.0	0.76 ± 0.01	6.5 ± 0.08	178.0 ± 3.7***	0.42 ± 0.13	0.29 ± 0.04	0.03 ± 0.003	0.01 ± 0.0006†
RC	34.2 ± 0.2								
0-10cm		159.6 ± 2.1	0.59 ± 0.01***	6.6 ± 0.12	142.3 ± 11.8	0.48 ± 0.11	0.30 ± 0.02	0.01 ± 0.002	0.01 ± 0.0005†
10-20 cm		170.1 ± 2.1***	0.65 ± 0.01***	6.7 ± 0.08***	159.1 ± 3.2	0.62 ± 0.16	0.35 ± 0.02	0.02 ± 0.001	0.01 ± 0.0006†
20-30cm		177.6 ± 1.9	0.71 ± 0.01	6.6 ± 0.13	184.1 ± 9.7***	0.49 ± 0.17	0.34 ± 0.03	0.04 ± 0.008	0.01 ± 0.0006†

*[1] Using a ratio of 1:2.5 field moist soil: H_2O. Values are presented as mean ± S.E. Significant differences between initial core values and post-experiment values for each treatment (within each soil layer) are marked with * for $p<0.05$, ** for $p<0.01$, ***$p<0.001$, and † for non-parametric (Kolmogorov-Smirnov Z statistic, Bonferroni corrected).*

5.3.3. Patterns of GHG emissions and related soil variables over time

5.3.3.1. Effect of water table raising and draining on GHG emissions

Mean carbon dioxide flux varied from 0.01 ± 0.75 to 137.86 ± 11.72 mg CO_2-C m^{-2} h^{-1} across all treatments during the wetted period of the experiment, with markedly lower fluxes observed in the WH treatment (0.01 ± 0.75 to 68.66 ± 15.89 mg CO_2-C m^{-2} h^{-1}) compared with the C1 and WL treatments (26.50 ± 4.74 to 137.86 ± 11.72 mg CO_2-C m^{-2} h^{-1}) (Fig. 5.1.b.). Soil respiration responded rapidly to soil wetting, with the mean WH flux falling to close to zero within five days of water table raising, and remaining lower (11.40 ± 1.39 mg CO_2-C m^{-2} h^{-1}) than mean fluxes from C1 and WL treatments (75.54 ± 3.62 mg CO_2-C m^{-2} h^{-1} and 77.86 ± 3.92 mg CO_2-C m^{-2} h^{-1} respectively) for the remainder of the 81 d wetted period.

After draining the water table cores on 20th August (85 days after the start of the experiment), both the WH treatment and WL treatments peaked almost immediately, on day one at 176.28 ± 39.09 mg CO_2-C m^{-2} h^{-1} and on day ten at 121.98 ± 4.56 mg CO_2-C m^{-2} h^{-1} respectively). By day 44 after draining, all treatments showed similar fluxes.

During the wetted period, mean nitrous oxide emission varied between 4.97 ± 6.03 and 4452.76 ± 576.60 µg N_2O-N m^{-2} h^{-1} across all treatments (Fig. 5.1.c.). A substantial peak of 4452.76 ± 576.60 µg N_2O-N m^{-2} h^{-1} was observed from the WL treatment 14 days after treatment application; WL emissions remained consistently higher than the WH and C1 treatments on most days until after day 44. Although smaller peaks were observed from the WH treatment on days 4 (859.78 ± 304.41 µg N_2O-N m^{-2} h^{-1}) and 20 (733.19 ± 376.83 µg N_2O-N m^{-2} h^{-1}), the average N_2O emission from the WH treatment was similar to that of the control throughout the wetted period, with no differences apparent by day 24. Core draining resulted in a short-lived rise (c. 14 d) in N_2O flux in all three treatments, compared to the average for the 23 days preceding draining (71.06 ± 17.77 µg N_2O-N m^{-2} h^{-1}) and the remainder of the drained period (36.36 ± 5.58 µg N_2O-N m^{-2} h^{-1}). The most pronounced response was observed from the WL treatment immediately following draining, which peaked at 1506.20 ± 498.88 µg N_2O-N m^{-2} h^{-1} one day after draining the cores; WH emission exhibited a smaller, delayed response of 698.71 ± 276.99 µg N_2O-N m^{-2} h^{-1} three days after draining.

Fluxes of CH_4 were lower and more variable than those of the other GHGs throughout both the wetted and drained period (Fig. 5.1.d.). Mean emissions ranged from -205.89 ± 203.50 to

242.47 ± 227.43 µg CH_4-C m^{-2} h^{-1}, with the largest variation observed on day 69 of the drained period.

5.3.3.2. Effect of water table raising and draining on dissolved N

Mean soil water NO_3-N content (0-10 cm soil layer) was generally much higher (0.08 ± 0.54 to 217.14 ± 5.80 mg NO_3-N L^{-1}) than mean NH_4-N content (< 0.05 to 0.73 ± 0.73 mg NH_4-N L^{-1}), with the greatest differences observed during the wetted period (Fig. 1.e. and f.). Throughout most of the experimental period, mean WH NO_3-N was substantially lower than mean C1 or WL NO_3-N, with dissolved NO_3-N dropping from day one of the wetted period and declining to negligible levels (7.50 ± 0.69 mg NO_3-N L^{-1}) for the duration of the drained period. Nitrate levels in the WL and C1 treatments fluctuated around 166.47 ± 3.76 mg NO_3-N L^{-1} over the wetted period with the exception of days 44 and 58 inclusive, during which time WL nitrate fell to a lower level both than C1 and WH nitrate. Dissolved nitrate was also very similar during the drained period for WL and C1 treatments, and declined steadily over the measurement period to reach comparable levels to the WH treatment by day 62. Dissolved ammonium levels remained consistently low from May to November, with the exception of a few small peaks across all three treatments.

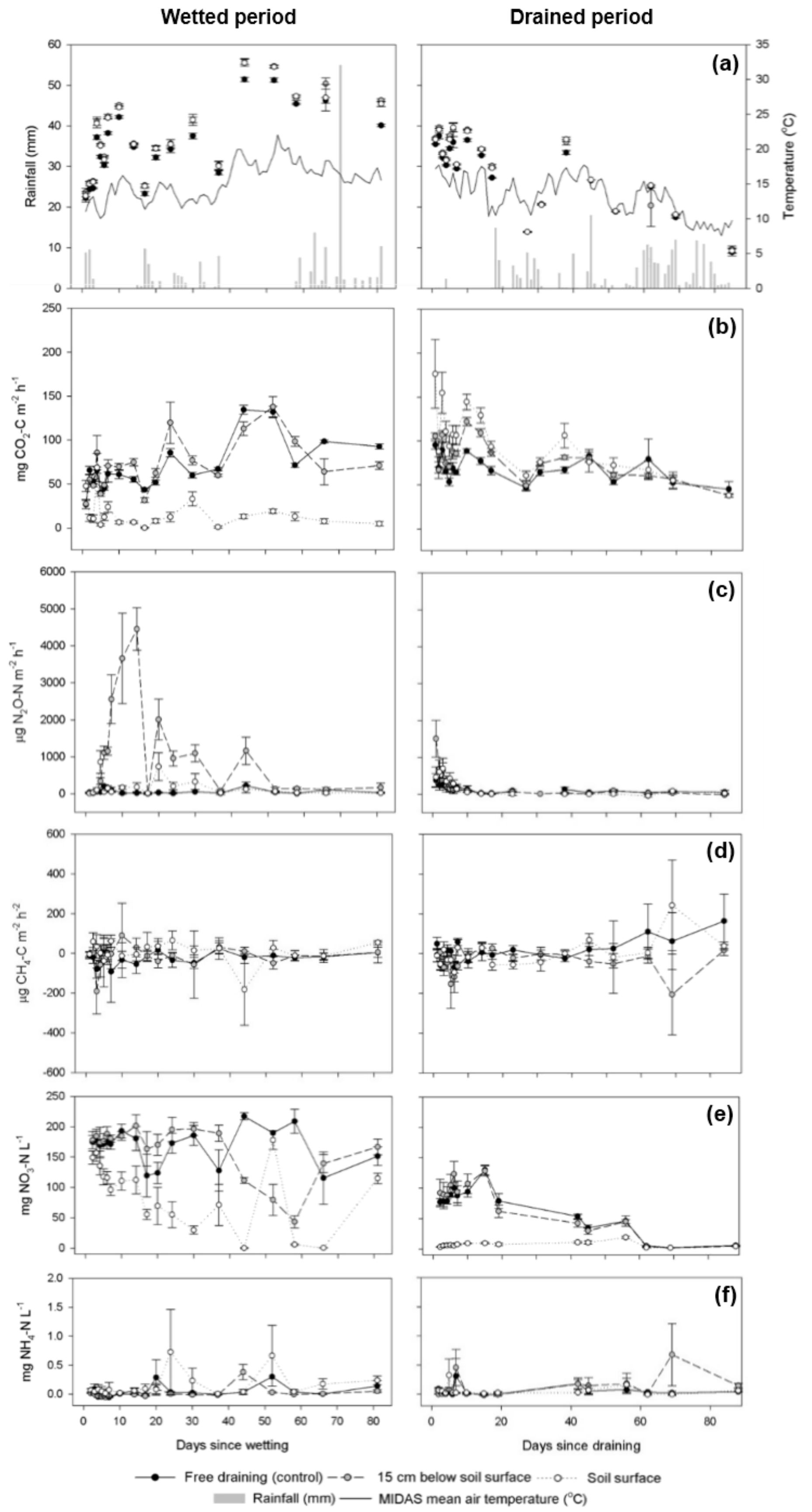

Figure 5.1. Daily rainfall, air temperature and soil temperature (a); fluxes of CO_2 (b), N_2O (c) and CH_4 (d), and soil water NO_3^- (e) and NH_4^+ (f); 28th May to 16th August (wetted) and 21st August to 13th November 2013 (drained): free-draining control (C1), water table at 15 cm below soil surface (WL) and water table at soil surface (WH). Error bars represent ± 1 S.E. of the mean value.

5.3.3.3. Effect of water table draining on Eh

Redox potential (Eh) was measured at depths of 0 cm, 10 cm, 20 cm and 30 cm below the soil surface on each sampling date during the drained period (Fig. 5.2.). Eh values from all treatments at all depths fell within the range 202.2 ± 10.4 to 490.3 ± 8.0 mV.

Very similar Eh values were observed across all treatments at the soil surface (Fig. 5.2.a.). Redox values persisted at Eh > 400 mV for most of the monitoring period, except for drops in mean Eh values observed on days 4 (379.5 ± 27.1 to 393.1 ± 18.9 mV), 33 (386.8 ± 9.5 to 400.2 ± 16.4 mV), and 60 (329.4 ± 7.9 to 343.7 ± 1.3 mV). On the day of core draining, C1 Eh values were lower than in the deeper soil layers (438.7 ± 13.5 mV compared to an average of 486.8 ± 5.6 mV).

Similarly to the soil surface measurements, Eh at the 10 cm soil layer remained above 400 mV over the 85 d period, showing almost identical declines to those at the 0 cm layer on days 4, 33 and 60 (Fig. 5.2.b.). Additionally, on the day on which the cores were drained, Eh was notably lower in the WH treatment (368.7 ± 36.2 mV) than in the WL and C1 treatments (480.0 ± 11.4 to 487.3 ± 10.4 mV).

Upon draining, an immediate and marked drop in Eh was observed in the 20 cm soil layer in both the WH (315.4 ± 45.5 mV) and WL (422.0 ± 41.7 mV) cores, compared with the control (490.3 ± 8.0 mV, Fig. 2.c.). From day four onwards, there were no observable differences between treatments, although mean WH Eh did remain consistently slightly lower compared to the WL and control Eh.

Redox potentials in the 30 cm soil layer were the most responsive to water table treatments when compared to the other soil layers (Fig. 5.2.d.). Both WH and WL treatments exhibited similar substantially lower mean Eh values (217.7 ± 17.4 mV and 226.9 ± 18.9 mV respectively) compared with the control cores (341.1 ± 23.6 mV), for 38 days after water table lowering. The greatest difference between the control and other treatments was observed during the first ten days after draining, after which, mean Eh of both WL and WH treatments increased, and mean WH redox values tended to remain lower than mean WL redox values. By day 62, WL redox values had returned to that of the control values, whereas the WH Eh took until day 85 to recover to levels seen in the control.

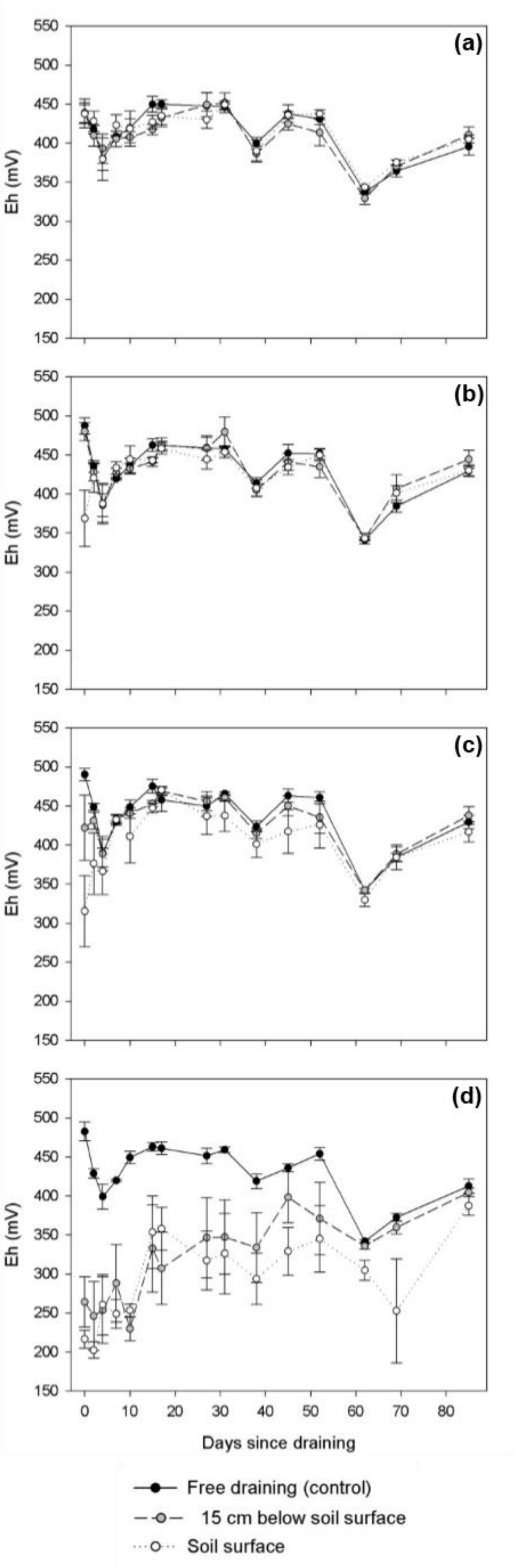

Figure 5.2. Redox potentials at soil depths of 0 cm (a), 10 cm (b), 20 cm (c) and 30 cm (d); drained period 21st August to 13th November 2013: free-draining control (C1), water table at 15 cm below soil surface (WL) and water table at soil surface (WH). Error bars represent ± 1 S.E. of the mean value.

5.3.3.4. Effect of fleece application on GHG emission

Mean CO_2 emission from C1 and F cores varied between 44.67 ± 14.54 and 232.26 ± 61.04 mg CO_2-C m^{-2} h^{-1} during the 81 d experimental period. Soil respiration from the F and C1 cores followed a similar pattern (Fig. 5.3.b.), with mean CO_2 emission being observably higher from the F treatment on the majority of measurement dates. Fleece treatment peak flux (232.26 ± 61.04 mg CO_2-C m^{-2} h^{-1}) occurred on day 52, and was almost double that of the control emission 132.07 ± 6.64 mg CO_2-C m^{-2} h^{-1}).

Mean N_2O emissions were similar from the F and C1 treatments throughout most of the experimental period, ranging from -32.13 ± 11.32 to 542.07 ± 180.75 µg N_2O-N m^{-2} h^{-1} (Fig. 5.3.c.). Maximum N_2O emission from the F treatment (542.07 ± 180.75 µg N_2O-N m^{-2} h^{-1}) occurred seven days after fleece application, returning to C1 levels by day fourteen. Further notable emission peaks were observed on day 44 (524.61 ± 224.13 µg N_2O-N m^{-2} h^{-1}) and day 81 (216.06 ± 57.61 µg N_2O-N m^{-2} h^{-1}).

Fluxes of CH_4 of -62.10 ± 23.22 to 127.91 ± 29.20 µg CH_4-C m^{-2} h^{-1} were observed between May and August (Fig. 5.3.d.). Substantial differences in fluxes between C1 and F treatments were observed on very few sampling dates. A maximum efflux of 127.91 ± 29.20 µg CH_4-C m^{-2} h^{-1} was observed ten days after fleece application.

5.3.3.5. Effect of fleece application on dissolved N

Similarly to the water table treatments, mean soil water NO_3-N content (115.52 ± 43.00 to 217.14 ± 5.80 mg NO_3-N L^{-1}) was much greater than mean NH_4-N content (< 0.05 ± 0.01 to 0.29 ± 0.30 mg NH_4-N L^{-1}) (Fig. 5.3.e. and f.). Other than a short period 10-20 days after fleece application where mean fleece dissolved nitrate concentration (197.09 ± 9.42 mg NO_3-N L^{-1}) was markedly higher than mean control concentration (156.08 ± 12.29 mg NO_3-N L^{-1}), mean NO_3-N and NH_4-N concentrations were very similar to each other on all sampling dates.

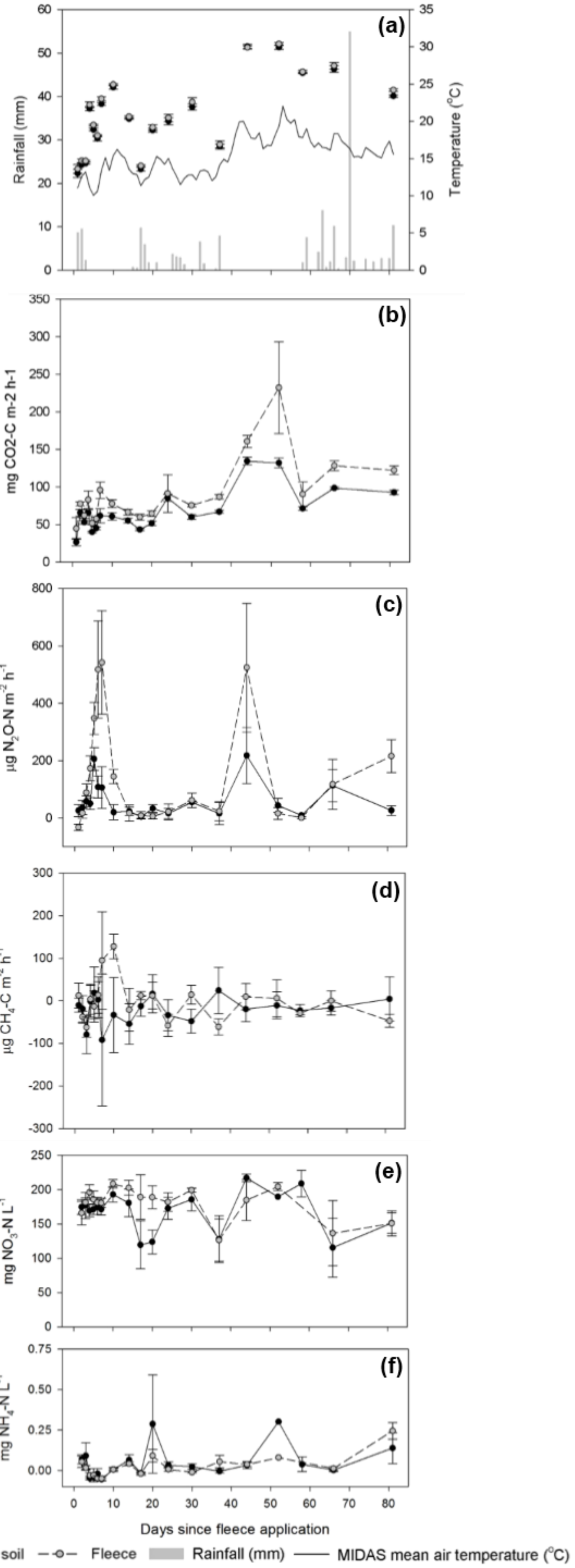

Figure 5.3. Daily rainfall, air temperature and soil temperature (a); fluxes of CO_2 (b), N_2O (c) and CH_4 (d), and soil water NO_3^- (e) and NH_4^+ (f); 28th May to 16th August 2013: uncovered control (C1), and fleece (F) treatments. Error bars represent ± 1 S.E. of the mean value.

5.3.3.6. Effect of cultivation and residue application on GHG emissions

Greenhouse gas fluxes from cultivation, surface-applied residue, and incorporated residue treatments are presented in Fig. 5.4.

5.3.3.6.1. Cultivation

Mean soil respiration was very similar on most sampling dates from the C1 and C2 treatments, ranging from 26.50 ± 4.74 to 134.51 ± 5.18 mg CO_2-C m^{-2} h^{-1} (Fig. 5.4.b.). A marked peak in the C2 flux was observed almost immediately following ploughing (day 2, 128.25 ± 13.43 mg CO_2-C m^{-2} h^{-1}). Carbon dioxide emission was observably lower from the C2 than from the control cores on days 10, 37, 44 and 52. Mean N_2O fluxes (-8.47 ± 6.31 to 218.23 ± 97.75 µg N_2O-N m^{-2} h^{-1}) were also comparable between the ploughed and control cores (Fig. 5.4.c.). Peaks in emission occurred on days 5, 44 and 66 from both C1 and C2 cores. Mean CH_4 emission (-91.67 ± 154.91 to 208.72 ± 87.67 µg CH_4-C m^{-2} h^{-1}) differed very little between C1 and C2 cores, except for on day 10 when C2 showed a large positive flux of 208.72 ± 87.67 µg CH_4-C m^{-2} h^{-1} (Fig 5.4.d.).

5.3.3.6.2. Surface residue application

Surface residue application had a pronounced effect on CO_2 fluxes: R treatment fluxes varied from 65.58 ± 4.06 to 196.47 ± 13.59 mg CO_2-C m^{-2} h^{-1} compared with control fluxes of 26.50 ± 4.74 to 134.51 ± 5.18 mg CO_2-C m^{-2} h^{-1} (Fig. 5.4.b.). Soil respiration from the R treatment markedly increased immediately following residue application, with elevated levels persisting until day 20. Nitrous oxide emissions also responded to surface residue application, but with a slower response (6 d), and a longer period (37 d) of markedly elevated, fluctuating emissions compared to the control treatment (Fig. 5.4.c.). Mean N_2O emissions for the R treatment were within the range 3.07 ± 5.88 to 496.21 ± 171.52 µg N_2O-N m^{-2} h^{-1}, with the highest positive flux observed on day 17. Methane fluxes from the control and R treatments fell within a comparable range (-450.25 ± 366.15 to 178.72 ± 108.85 µg CH_4-C m^{-2} h^{-1}) during the 81 d monitoring period (Fig 5.4.d.). A small peak in residue CH_4 emission on day three, whilst distinctive negative fluxes were observed on days 17, 44 and 81.

5.3.3.6.3. Incorporated residue

Patterns of fluxes from cores into which the lettuce residues were incorporated were similar to those observed when residue was surface applied, but with a smaller effect on emissions; all flux values fell within the range 26.50 ± 4.74 to 193.23 ± 3.54 mg CO_2-C m^{-2} h^{-1} (Fig. 5.4.). Soil respiration responded to residue incorporation within two days, rising to its maximum for the monitoring period (193.23 ± 3.54 mg CO_2-C m^{-2} h^{-1}), which was markedly higher than peak emissions on that day from both the cultivated treatment (128.25 ± 20.03 mg CO_2-C m^{-2} h^{-1}) or the application of residue alone (173.85 ± 30.63 mg CO_2-C m^{-2} h^{-1}, Fig. 5.4.b.). An elevated CO_2 response in the RC treatment was observed until day 30, after which RC fluxes were on average lower than the C1 and C2 fluxes. Mean N_2O fluxes varied from -2.57 ± 13.20 to 413.33 ± 168.46 µg N_2O-N m^{-2} h^{-1} (Fig. 5.4.c.). Incorporating residue produced a notable flux peak by day five (413.33 ± 168.46 µg N_2O-N m^{-2} h^{-1}), after which, RC emission returned to close to control emissions (41.29 ± 17.00 and 15.14 ± 10.36 µg N_2O-N m^{-2} h^{-1} respectively); subsequently, peak RC fluxes were observed on days 20 and 30. Like soil respiration, RC N_2O emission was lower than from the C1 and C2 treatments towards the end of the experimental period (day 52 onwards). There were no observable differences between CH_4 emissions from incorporated residue cores compared to C1 and C2 treatments between May and August, although comparatively large effluxes were observed on days 17 and 20 (Fig. 5.4.d.). Methane fluxes fell within the range (-112.35 ± 121.12 to 496.64 ± 485.55 µg CH_4-C m^{-2} h^{-1}).

5.3.3.7. Effect of cultivation and residue application on dissolved N

Mean soil water NO_3-N content (115.52 ± 43.00 to 217.14 ± 5.80 mg NO_3-N L^{-1}) for all cultivation and residue treatments was considerably higher than mean soil NH_4-N content for these cores (< 0.05 to 0.50 ± 0.21 mg NH_4-N L^{-1}) (Fig 5.4.e. and f.). On the majority of sampling dates, there were no discernible differences in NO_3-N and NH_4-N concentrations between treatments, with a small number of exceptions. Dissolved nitrate concentration was markedly lower in the cultivated treatment on day ten and days 44-52, higher in the surface-applied residue treatment on days 17-20, and higher in the incorporated residue treatment on day 52 when compared to the control treatments; ammonium content was notably higher in the cultivation treatment on day 66 only, and higher in the incorporated residue treatment on days 44-52 than the control.

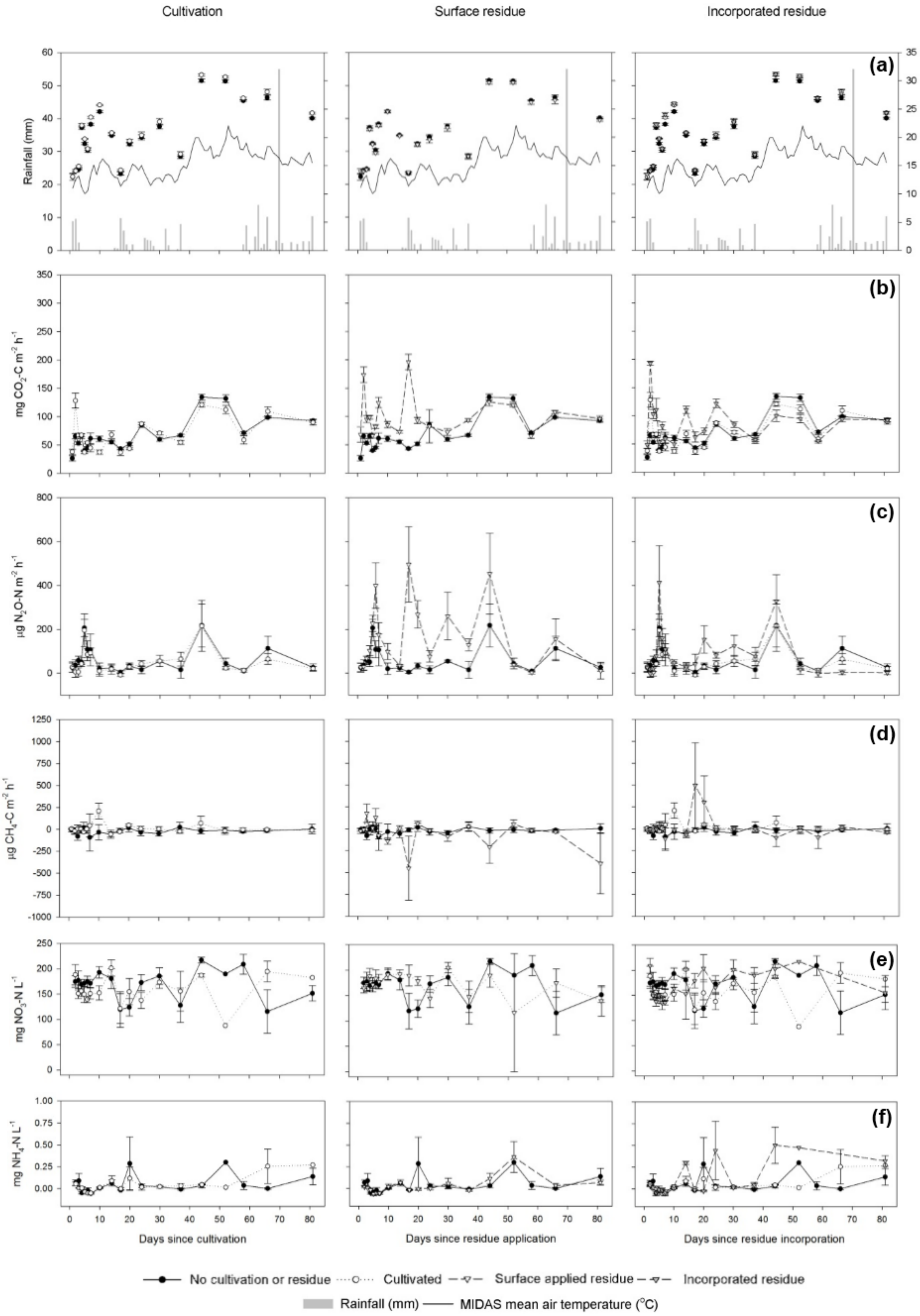

Figure 5.4. Daily rainfall, air temperature and soil temperature (a); fluxes of CO_2 (b), N_2O (c) and CH_4 (d), and soil water NO_3^- (e) and NH_4^+ (f); 28th May to 16th August 2013: control without cultivation or residue application (C1), cultivation (C2), surface applied residue (R) and incorporated residue (RC) treatments. Error bars represent ± 1 S.E. of the mean value.

5.3.4. Effect of soil and weather conditions on GHG emissions

Redox potential became significantly correlated with soil respiration at 20 cm depth and below ($p < 0.05$), and with N_2O at 30 cm depth ($p < 0.05$), and was not significantly correlated with CH_4 emissions ($p > 0.05$) (Tables 5.3. and 5.4.). At 20 cm below the soil surface, Eh was positively associated with CO_2 emission in the control and WL treatments, explaining 3% of the variability in soil respiration (τ = -0.176 to -0.179). At 30 cm depth, Eh was negatively associated with CO_2 emission in the WH treatment, and N_2O emission in the WH and WL treatments, explaining 3% of CO_2 emission variability and 3-6% of N_2O emission variability (τ = -0.174 to -0.254).

Soil temperature, MIDAS mean air temperature, and measured air temperature were positive, highly significant predictors of soil respiration within most treatments, accounting for between 12-31%, 3-38%, and 5-18% of fluxes respectively (τ = 0.341 to 0.559, p < 0.05 to < 0.01); Table 5.3.). Temperature variables were less suitable for predicting N_2O emissions, although some highly significant correlations were still found. Soil temperature, MIDAS mean air temperature, and measured air temperature predicted 2-10%, 3-7%, and 3-12% of N_2O emissions respectively (τ = 0.147 to 0.313, p < 0.05 to < 0.001). There were few significant linear predictors of CH_4 emissions, but temperature variables showed a significant negative relationship with CH_4 emission in the control and surface-applied residue treatments, accounting for 3-21% CH4 emissions (τ = -0.170 to -0.461, $p < 0.05$; Table 5.4.).

Daily and 5-day rainfall (cumulative rainfall from the day of measurement and the five preceding days) were negative highly significant predictors of CO_2 emissions for most of the treatments (τ = -0.112 to -0.460; p < 0.05 to < 0.001), while daily rainfall was positively significantly correlated with surface-applied residue CO_2 efflux (τ = 0.180, p < 0.05; Table 5.3.). Daily rainfall explained 1-8% and 5-day rainfall explained 2-21% of soil respiration. Nitrous oxide emissions and daily rainfall were highly significantly negatively correlated in all but the drained C1 treatment, accounting for 2-34% of emissions (τ = -0.136 to -0.579, p < 0.05 to < 0.001). Cumulative 5-day rainfall was a significant predictor of N_2O emission for only the -15 cm water table treatment, explaining 4-7% of N_2O flux (τ = -0.199 to -0.260; $p < 0.001$). Methane emissions were not significantly correlated with rainfall ($p > 0.05$; Table 5.4.).

Dissolved nitrate, ammonium, or total dissolved N were significant predictors of soil respiration in most treatments. Declining nitrate concentration was correlated with increasing

CO_2 emissions for the WL treatment during the wetted period and for the WL and WH treatments overall (2-9% of variability, $\tau = -0.111$ to -0.298, $p < 0.05$ to < 0.001), but was significantly positively correlated with WL CO_2 emissions during the drained period (20% of variability, $\tau = 0.445$, $p < 0.001$). Nitrate concentration was also positively correlated with C1 (drained), C2 and RC treatments (3-6% of variability, $\tau = 0.182$ to 0.243, $p < 0.05$). Nitrous oxide emissions and nitrate concentration were significantly positively correlated in the C1 (wetted) and WL (drained, whole period) treatments, with NO_3-N accounting for 3-13% of variability in N_2O emission ($\tau = 0.185$ to 0.358, $p < 0.05$ to < 0.001). Dissolved NO_3-N concentration was a positive predictor of CH_4 emissions within the F treatment (3% of variability, $\tau = 0.179$, $p < 0.05$; Table 5.4.). Soil solution ammonium concentration was positively associated with soil respiration in the C1 (wetted), WL (wetted, whole period), and C2 treatments (2-7% of variability, $\tau = 0.135$ to 0.255, $p < 0.05$ to < 0.01), but negatively associated with soil respiration in the C1 (drained) treatment (3% of variability, $\tau = -0.187$, $p < 0.05$). A significant correlation between dissolved ammonium concentration and N_2O emission was found in only the surface-applied residue treatment (9% of variability, $\tau = -0.292$, $p < 0.01$), and with CH_4 emissions in the fleece treatment (6% of variability, $\tau = -0.239$, $p < 0.01$; Table 5.4.). Total dissolved N was generally a poorer predictor of GHG emission than NO_3-N or NH_4-N alone, either showing fewer significant correlations (as with CO_2 or CH_4 emission), or explaining less of the variability (as with N_2O emission).

Table 5.3. Significant linear correlations between measured environmental variables and emissions of CO_2 and N_2O.

	Treatment	Eh (mV)				Soil t	Mean air t	Air t	D rain	5d rain	NO_3-N	NH_4-N	N
		0 cm	10 cm	20 cm	30 cm	(°C)	(°C)	(°C)	(mm)	(mm)	(mg L^{-1})	(mg L^{-1})	(mg L^{-1})
CO_2	**C1, wetted**	-	-	-	-	0.539***	0.617***	0.322***		-0.174*		0.254**	
	WL, wetted	-	-	-	-	0.559***	0.538***	0.420***	-0.238**	-0.360***	-0.152*	0.254**	-0.199*
	WH, wetted	-	-	-	-				-0.169*				
	C1, drained			0.176*		0.345***	0.384***	0.231**		-0.219**	0.182*	-0.187*	
	WL, drained			0.179*		0.443***	0.442***	0.357***	-0.279***	-0.460***	0.445***		
	WH, drained				-0.174*	0.474***	0.481***	0.395***	-0.289***	-0.404***			
	C1, whole period	-	-	-	-	0.381***	0.528***	0.279***		-0.212***			
	WL, whole period	-	-	-	-	0.353***	0.523***	0.359***	-0.236***	-0.407***	-0.111*	0.135*	
	WH, whole period	-	-	-	-		0.162**		-0.236***	-0.130***	-0.298***		-0.191**
	F	-	-	-	-	0.539***	0.595***	0.365***		-0.153*			
	C2	-	-	-	-	0.341***	0.392***	0.365***			0.243**	0.255**	
	R	-	-	-	-		0.230**		0.180*				0.216**
	RC	-	-	-	-		0.166*		-0.112*		0.219*		
N_2O	**C1, wetted**	-	-	-	-				-0.212**		0.185*		
	WL, wetted	-	-	-	-	0.180*			-0.579***	-0.260***			
	WH, wetted	-	-	-	-				-0.357***				0.207*
	C1, drained												
	WL, drained				-0.174*	0.283***	0.258**	0.345***	-0.271**		0.358***		0.254**
	WH, drained				-0.254*	0.285**	0.160*	0.302**	-0.216*				
	C1, whole period	-	-	-	-				-0.136*				
	WL, whole period	-	-	-	-	0.313***		0.204***	-0.440***	-0.199***	0.347***		0.241***
	WH, whole period	-	-	-	-	0.153**		0.168**	-0.291***				
	F	-	-	-	-	0.147*			-0.237**				
	C2	-	-	-	-				-0.240**				
	R	-	-	-	-		-0.185*	-0.171*	-0.186*			-0.292**	
	RC	-	-	-	-		-0.171*		-0.407***				

*Values are presented as Kendall's tau statistic (τ), with significance levels presented as * ($p < 0.05$), ** ($p < 0.01$), or *** ($p < 0.001$).*

Table 5.4. Significant linear correlations between measured environmental variables and emissions of CH_4.

	Treatment	Eh (mV)				Soil t	Mean air t	Air t	D rain	5d rain	NO_3-N	NH_4-N	N
		0 cm	10 cm	20 cm	30 cm	(°C)	(°C)	(°C)	(mm)	(mm)	(mg L^{-1})	(mg L^{-1})	(mg L^{-1})
CH_4	C1, wetted	-	-	-	-								
	WL, wetted	-	-	-	-								
	WH, wetted	-	-	-	-								
	C1, drained					-0.170*	-0.164*	-0.179*					
	WL, drained												
	WH, drained												
	C1, whole period	-	-	-	-								
	WL, whole period	-	-	-	-								
	WH, whole period	-	-	-	-								
	F	-	-	-	-						0.179*	-0.239**	
	C2	-	-	-	-								
	R	-	-	-	-	-0.461*		-0.199**					
	RC	-	-	-	-								

*Values are presented as Kendall's tau statistic (τ), with significance levels presented as * ($p < 0.05$), ** ($p < 0.01$), or *** ($p < 0.001$).*

5.3.5. Cumulative GHG emissions

5.3.5.1. Effect of water table raising and draining on cumulative GHG emissions

5.3.5.1.1. Wetted period

Cumulative CO_2 emission over the period 29th May to 16th August was significantly influenced by water table depth ($H(2) = 8.54$, $p < 0.01$; Table 5.5.). Jonckheere's test revealed a significant decline in the median CO_2 emission as the water table was raised closer to the soil surface ($J = 10$, $z = -2.91$, $r = -0.75$). Bonferroni-corrected Post-Hoc Kolmogorov-Smirnov Z tests ($p = 0.0167$), revealed a significant difference between the control and surface water table median CO_2 emission only ($KSZ = 1.581$, $p < 0.01$, $r = 0.49$), although the difference between WL and WH median CO_2 emission was almost significant ($KSZ = 1.265$, $p = 0.08$, $r = 0.40$). Cumulative N_2O emission was significantly influenced by water table depth ($F = 52.83$, $p < 0.001$), with mean WL cumulative flux being significantly higher than both the control and WH cumulative emission (Tukey HSD; both $p < 0.001$). No significant differences or trends in cumulative CH_4 emissions were found between treatment groups. Cumulative GWP for water table treatments was significantly different between groups ($F = 93.92$, $p < 0.001$); with a highly significant increase in cumulative GWP_{100} in the order WH < C1 < WL (Tukey HSD; all $p \leq 0.001$).

5.3.5.1.2. Drained period

Significant differences were found in median CO_2 emissions between water table groups ($H = 7.740$, $p < 0.05$), although when Bonferroni-corrected Post-Hoc Kolmogorov-Smirnov Z tests were applied, differences between all pairs of treatments were only almost significant (in all cases, $KSZ = 1.265$, $p = 0.08$; Table 5.5.). Despite this, a significant trend in cumulative CO_2 emission with a large effect size was found ($J = 66$, $z = 3.01$, $r = 0.78$): median cumulative soil respiration increased as previous water table depth became shallower (closer to the soil surface). No significant differences were found between water table treatment groups for median cumulative N_2O, mean cumulative methane, or mean cumulative GWP_{100}.

5.3.5.1.3. Entire wetted and drained period

For the whole measurement period 29th May to 28th October, soil respiration was highly significantly influenced by water table depth ($F = 62.99$; $p < 0.001$), with a highly significant decline in soil respiration between WL and WH treatments (Tukey HSD; $p < 0.001$) but no significant difference in cumulative CO_2 emission between the control and WL treatments. Water table also had a highly significant effect on log-normal cumulative N_2O emission over the five month period ($F = 35.29$, $p < 0.001$). Mean N_2O emissions were significantly higher from the WL treatment compared to the control (Tukey HSD; $p < 0.001$) and to the WH treatment (Tukey HSD; $p < 0.001$). The effect of water table depth on cumulative methane emission was non-significant. Water table treatment had a highly significantly effect on mean cumulative GWP_{100} ($F = 53.67$, $p < 0.001$), and all treatments were significantly different to each other: WH was lower than both C1 and WL (Tukey's HSD; $p < 0.05$ and $p < 0.001$ respectively), and C1 was lower than WL (Tukey's HSD; $p < 0.001$).

5.3.5.2. Effect of fleece application on cumulative GHG emissions

Cores with fleece applied to them gave a significantly higher mean cumulative emission of CO_2 than the control, with a large size effect ($t(8) = -3.357$, $p < 0.05$, $r = 0.81$; Table 5.5.). Fleece cumulative N_2O emission was higher than control cumulative emission, at an almost significant level ($t(5.313) = 2.399$, $p = 0.06$). Mean cumulative CH_4 emissions were slightly higher from the F than C1, although this difference was not significant. The fleece treatment had a significantly greater cumulative GWP_{100} emission than the control ($t(4.307) = -4.553$, $p < 0.01$).

5.3.5.3. Effect of cultivation and residue application on cumulative GHG emissions

Over the 80 d experimental period, ploughing had no significant effect when compared to undisturbed soil on cumulative individual GHG emissions or overall GWP_{100}, although C2 cumulative emissions of CO_2, N_2O and overall GWP_{100} were slightly lower than for the control, and CH_4 cumulative emission was slightly higher in the cultivated treatment compared to the control (Table 5.5.).

The surface-applied residue treatment yielded a significantly higher mean cumulative soil respiration (Tukey's HSD; $p < 0.01$), mean cumulative N_2O emission (Tukey's HSD; $p < 0.05$), and median cumulative GWP_{100} ($KSZ = 1.582$; $p < 0.01$) than the undisturbed bare soil control treatment. Cumulative CH_4 emission (net uptake) was lower in the R than the C1 treatment, but not at a significant level. Although the C2 treatment was not intended for comparison against the R treatment, it is worth noting that this comparison produced the same pattern and significance values as when C1 and R were compared.

No significant differences were observed in any of the individual cumulative GHG emissions or overall GWP_{100} between the undisturbed bare soil control and the incorporated residue treatment, or the cultivated control and the incorporated residue treatment, although in all emissions categories, cumulative emissions were higher from the RC treatment than the C1 or C2 treatments. Compared to the surface-residue application treatment, cumulative emissions from the incorporated residue treatment were lower for soil respiration (Tukey's HSD; $p < 0.05$), nitrous oxide (Tukey's HSD; *ns*), and GWP_{100} (Tukey's HSD; *ns*), and slightly higher for methane (KSZ; *ns*).

Table 5.5. Cumulative fluxes of CO_2, N_2O and CH_4, and total cumulative GHG emissions (GWP_{100}) in t CO_2-e ha^{-1} $period^{-1}$ (± S.E.), for control (C1), cultivated (C2), fleece (F), surface residue (R), incorporated residue (RC), water table at -15 cm below soil surface (WL), and water table at soil surface (WH) treatments. Totals are reported separately for the water table treatments for the wetted (28th May to 16th August), drained (21st August to 28th October), and whole measurement period (28th May to 28th October).

	28th May – 16th August, t CO_2-e ha^{-1} 80 d^{-1}				**21st August – 28th October, t CO_2-e ha^{-1} 69 d^{-1}**				**28th May – 28th October, t CO_2-e ha^{-1} 153 d^{-1}**			
	CO_2	**N_2O**	**CH_4**	**GWP_{100}**	**CO_2**	**N_2O**	**CH_4**	**GWP_{100}**	**CO_2**	**N_2O**	**CH_4**	**GWP_{100}**
C1	5.87 ± 0.06	0.58 ± 0.11	0.002 ± 0.01	6.45 ± 0.11	4.09 ± 0.29	0.74 ± 0.26	0.01 ± 0.01	4.84 ± 0.32	10.29 ± 0.35	1.41 ± 0.38	0.01 ± 0.01	11.72 ± 0.43
C2	5.63 ± 0.22	0.52 ± 0.10	0.01 ± 0.10	6.16 ± 0.27								
F	7.83 ± 0.58	1.25 ± 0.26	0.03 ± 0.03	9.11 ± 0.57								
R	7.07 ± 0.26	1.48 ± 0.30	-0.04 ± 0.02	8.50 ± 0.31								
RC	5.99 ± 0.18	0.82 ± 0.23	0.01 ± 0.01	6.82 ± 0.35								
WL	5.72 ± 0.22	8.01 ± 0.96	-0.00003 ± 0.01	13.72 ± 0.94	4.58 ± 0.11	0.77 ± 0.13	0.0003 ± 0.02	5.35 ± 0.20	10.61 ± 0.30	9.17 ± 1.15	0.0004 ± 0.01	19.78 ± 1.19
WH	0.85 ± 0.12	1.21 ± 0.38	-0.0002 ± 0.01	2.06 ± 0.46	5.30 ± 0.23	0.46 ± 0.22	0.01 ± 0.01	5.77 ± 0.37	6.47 ± 0.20	1.78 ± 0.44	0.01 ± 0.01	8.26 ± 0.60

5.4. Discussion

5.4.1. Effect of water table raising and draining on GHG emissions

5.4.1.1. Carbon dioxide

In this study, raising the water table to the soil surface significantly lowered soil respiration rate and reduced cumulative CO_2 emission, a relationship observed in other studies of both fen and blanket peat mesocosms (e.g. Dinsmore *et al*., 2009; Freeman *et al*., 1993; Kechavarzi *et al*., 2007) and field studies of grass moorland (e.g. Lloyd, 2006). Similarly to this study's results, Dinsmore *et al*. (2009), Freeman *et al*. (1993), and Kechavarzi *et al*. (2007) reported a high sensitivity of CO_2 emission to comparatively small variations in water table depth, although the magnitude of this response (1-59% change in emission within a 20-30 cm change in depth, compared to this study's 29-58% change in emission within a 15-30 cm change in depth), and the depth at which a change in water table significantly affected CO_2 emission (20-50 cm depth, compared to 15 cm depth here), differs between studies.

The magnitude of CO_2 emissions observed from water table cores here (0.01 ± 0.75 to 176.28 ± 39.09 mg CO_2-C m^{-2} h^{-1}) is comparable to the range observed in peat cores from a nutrient-poor grazed grassland by Dinsmore *et al*. (2009) (90.39 ± 6.98 to 164.07 ± 25.54 mg CO_2-C m^{-2} h^{-1}) and a lowland arable soil by Kechavarzi *et al*. (2007) (2.92 to 104.97 mg CO_2-C m^{-2} h^{-1}), but considerably higher than those observed by Freeman *et al*. (1993) in an upland flushed peat mire (7.35 to 18.08 mg CO_2-C m^{-2} h^{-1}), possibly due to predominantly anaerobic soil conditions restricting microbial metabolism in the latter. It is hard to tell if the rapid response of soil respiration to wetting and draining found in this study is typical, as few studies measure emissions daily after changing the water table depth, but response time appears to be relative to the rate of wetting, with slower wetting resulting in a slower decline in soil respiration rates (Estop-Aragonés and Blodau, 2012; Freeman *et al*., 1993).

Highly significant correlations were found between elevated soil respiration, and high air and soil temperatures and low rainfall (Table 5.3.). This result is supported by a number of studies investigating seasonal variations in peat soil respiration (e.g. Kechavarzi *et al*., 2007; Morrison *et al*., 2013). The effect of treatment interacted with temperature and moisture conditions, with the most pronounced peaks in emission in the control and WL treatments during the wetted period, and in the WH treatment during the drained period. WH soil respiration appeared to be unaffected by temperature during the wetted period (Table 5.3.), suggesting that soil moisture content was the dominant factor determining emissions in these cores. A similar dampening of CO_2 response to temperature under saturated or near-saturated conditions has been observed by Kechavarzi *et al*. (2007) and Lloyd (2006), as a result of the increasingly anaerobic soil environment inhibiting microbial respiration (Mitsch and Gosselink, 2000). Patterns of correlation between CO_2 emission and soil NO_3-N and NH_4-N were more complex and appeared to interact with water table treatment (Table 5.3.), although these correlations may be more of a reflection on N_2O emissions, which often co-vary with CO_2 emissions.

5.4.1.2. Nitrous oxide

Raising the water table to 15 cm below the soil surface resulted in significantly elevated N_2O emissions, but raising the water table to the soil surface had an overall insignificant effect on emissions compared to the control (section 5.3.3.1.; Fig. 5.1.c.). In accordance with this study's findings, Freeman *et al*. (1993) also found N_2O emission to be highly significantly inversely correlated with water table depth when changed from 0 cm to 20 cm depth, and Velthof and Oenema (1997) observed a general increase in N_2O emission with increasing soil water-filled pore space (WFPS%), falling to very low levels when soils were waterlogged.

The highest emission observed from the water table treatments in this study (4452.76 ± 576.60 $\mu g\ N_2O\text{-}N\ m^{-2}\ h^{-1}$) was two orders of magnitude higher than emissions from semi-natural peatland mesocosms observed by Freeman *et al*. (1993) and Dinsmore *et al*. (2009), but similar to the upper range found in field studies of arable peatlands (e.g. Flessa *et al*., 1998; Weslien *et al*., 2012). The low mean background emissions from the surface water table treatment (Fig. 5.2.c.) were comparable to those found from other flooded peats, both semi-natural (e.g. Dinsmore *et al*., 2009; Freeman *et al*., 1993) and agricultural (e.g. Flessa *et al*., 1998; Velthof and Oenema, 1997).

An extreme peak in N_2O emissions was observed from the WL treatment over the first 14 d after core wetting, and only a small pulse from the WH treatment. Conversely, WH cores provided the highest immediate emissions after draining, while the N_2O pulse from WL cores was smaller. These relatively rapid, short-lived, strong responses to wetting and draining events in peat soils are common and differ primarily according to the treatment effect on the balance between soil available N and moisture (Li *et al*., 1992a), while soil temperature interacts with these variables. Low emissions from semi-natural peats, even under optimal moisture conditions for N_2O release, are often a result of low soil nitrate limiting N_2O production (Dinsmore *et al*., 2009). Despite the cores having experienced a relatively long fallow period before wetting, and a large amount of N mineralisation in the WL treatment, the arable peat soil did not appear to be N-limited during the wetted period (available nitrate levels in the top 10 cm soil remained relatively high, Fig. 5.2.e.), possibly due to rapid NO_3-N replenishment or N_2O diffusion from below 10 cm soil depth. It is likely that the relative magnitude and pattern of N_2O emissions between water table treatments in this study were more strongly determined by soil moisture conditions (Fig. 5.2.; Table 5.3.). Nitrous oxide is usually produced when soil WFPS lies within the range 40-70%, above which, N_2 starts to dominate as the primary denitrification product (Dalal *et al*., 2003). During the wetted treatment period, WH WFPS would have approximated 100%, resulting in dominant N_2 rather than N_2O production; this is supported by the declining dissolved nitrate concentration observed in the WH soil water by the end of the wetted period (Fig. 5.1.e.), despite negligible N_2O emissions (Fig. 5.1.c.). The pulse of N_2O from the WH treatment immediately after drainage, coupled with the rapid drop in nitrate concentration, indicate a short-term optimisation of moisture conditions before the remaining nitrate was leached away or mineralised (Fig. 5.1.c., e.; Table 5.3.). All emissions, NO_3-N levels, and soil and air temperatures generally declined over the drained period (Fig. 5.1.c., e., a.), which along with the positive correlation between temperature and N_2O emission (Table 5.3.), suggest that conditions for N_2O production were suboptimal for all water table treatment cores by the end of the drained period.

5.4.1.3. Methane

Methane emissions observed throughout the experimental period were highly variable over time, both in terms of mean emission and between-core variability; it is likely that this high variability obscured any treatment effects or correlations with environmental variables, which tended to be more stable (Table 5.4.). High temporal and spatial within-treatment variability in CH_4 fluxes is commonly observed from drained and cultivated peat soils (e.g. IPCC, 2006).

There was no marked effect of water table treatment on CH_4 production over the wetted or drained experimental periods, contrary to the general trend of water table raising increasing emissions (Bussell *et al*., 2010). However, arable peats often exhibit lower emissions of CH_4 than semi-natural peats, even under temporarily waterlogged conditions (Kasimir-Klemedtsson *et al*., 1997). The cores were extracted from land subjected to cultivation for more than 50 years, resulting in well-aerated, relatively decomposed peats with a higher bulk density than semi-natural peats (Kechavarzi *et al*., 2010). The strictly anaerobic conditions required for substantial CH_4 emissions are sometimes slow to develop (more than one year; Oomes *et al*., 1997), particularly in well-aerated soils where there may be few anaerobic microsites, so the low fluxes recorded may be a function of the relatively short period of exposure to saturated conditions (three months). In addition, CH_4 emissions only tend to occur when there are no other terminal electron acceptors available (i.e. NO_3^-, Fe^{3+}, SO_4^{2-}).

5.4.2. Effect of water table draining on Eh

Measured redox values during the drained period fell within the range 202.2 ± 10.4 to 490.3 ± 8.0 mV. Estimates of the exact Eh thresholds for transitions in the dominant oxidative chemical species (and therefore the dominant GHG produced) vary, but as soil conditions become more reducing (anaerobic), GHG production tends to shift from CO_2 to N_2O to CH_4 (Li, 2007). The values fell within the range typically associated with dominant production of CO_2 (> +400 to +500 mV) and N_2O (+200 to +500 mV), but not the range associated with CH_4 production (< -200 to -100 mV, Li, 2007; Mitsch and Gosselink, 2000). The results here followed the Eh pattern commonly observed on predominantly aerobic drained peat soils (CO_2 production dominated overall GWP_{100}, N_2O was produced in smaller amounts, and CH_4 emission was negligible or a small net uptake).

After draining, a decline was observed in redox potential with increasing soil depth in the WL and WH treatments, which was more pronounced in magnitude and duration in the WH treatment than the WL treatment (Section 5.3.3.2., Fig. 5.2.), which is expected considering the probable soil moisture saturation within each treatment at each depth. Similar patterns of declining Eh with soil depth, particularly under saturated conditions, have been observed on peatland under arable and pasture land uses by Deuterelo *et al*. (2009), and on an unfertilised grassland peat soil by Oomes *et al*. (1997). It was not possible to measure soil moisture during the course of the experiment without causing core disturbance, but the observed Eh values serve as a proxy for soil moisture status over the drained period. The similarity between treatments in soil surface Eh from the day of draining onwards suggests that within the 0-10 cm layer, all treatment cores drained relatively rapidly. The low Eh values measured at 20 cm and 30 cm depths during the first ten days of draining, corresponding to the period of greatest N_2O and CO_2 production, suggest a slower drainage rate from the more saturated cores (especially at depth), and related to this, a gradient of soil moisture up through the soil profile which is proportionally influencing both CO_2 and N_2O emissions from the top soil (Table 5.3.).

In support of this theory, Deuterelo *et al*. (2009) observed that even when the water table was temporarily lowered in agricultural peats, Eh did not increase, suggesting persistent soil saturation after draining. Further, this predicted moisture gradient is supported by the pattern of CO_2 fluxes through most of the drained period (WH > WL > C1), probably reflecting a gradient in soil moisture levels conducive to microbial respiration activity.

Although it might be expected that the Eh of cores with previously raised water tables would be more sensitive to rainfall events owing to residual moisture at depth (Figs. 5.1.a., 5.2.), significant negative correlations were only found in a few soil layers and mainly in the C1 treatment (data not shown), suggesting that the opposite may be true, and that Eh was more stable (but lower) after draining in the WH and WL cores than in the control. This is supported by the observation that by the end of the drained period, soil moisture content was still significantly higher in the 0-10 cm soil layer in the WH and WL than the control cores (Table 5.2.). This could be due to the peat being more hydrophobic in the control treatment due to a greater number of soil wetting and drying events (Valat *et al.,* 1991).

5.4.3. Effect of fleece application on GHG emissions

This study found that fleece application significantly increased GWP_{100}, soil respiration and nitrous oxide emission, and slightly increased CH_4 emission, although not significantly (section 5.3.3.4.; Fig. 5.3.). There is consistent evidence that non-woven fleece appears to primarily affect the enclosed environment by increasing and stabilising variation in soil temperature (e.g. Hamouz *et al*., 2005; 2006; Siwek *et al*., 2012; 2013), and to a lesser extent by increasing enclosed air temperature (Hamouz *et al*., 2006), in keeping with the results of this study. In this study, temperature was the strongest predictor of soil respiration, showing a significant positive correlation from fleece-enclosed cores between CO_2 emissions and both air temperature (explaining 13-29% variability in soil respiration), and soil temperature (35% of variability; Table 5.3.). This is in accordance with other studies on the effect of temperature on peat soil respiration (e.g. Estop-Aragonés and Blodau, 2012; Maljanen *et al*., 2002). Soil temperature has also been shown to positively correlate with N_2O emissions (Maljanen *et al*., 2002), although in this study the relationship was not strong, only explaining 2% of the variability in emission (Table 5.3.). In this study, there was no significant correlation between methane flux and soil or air temperature (Table 5.4.). This is also in agreement with the observations of Maljanen *et al*. (2002) that the temperature dependence of CH_4 emissions tends to be weaker in agricultural peatlands than natural wetlands, and weaker in CH_4 uptake processes (the majority of fluxes here) than in methanogenesis. Considering the relationship between individual GHG emissions and temperature variables, the fluxes observed in this study are comparable to those which we might expect from fleece-induced temperatures (e.g. Maljanen *et al*. 2002), although the CO_2 and CH_4 emissions were somewhat higher, possibly related to the peat in this study being deeper with a higher SOC content than in other studies, providing a greater quantity of substrate for microbial breakdown.

5.4.4. Effect of cultivation and residue application on GHG emissions

5.4.4.1. Effect of cultivation

Applying a ploughing treatment in this study resulted in an immediate, small and short-lived peak in soil respiration and methane, and a negligible response of N_2O (Section 5.3.3.6., Fig. 5.4.). Ploughing-induced peaks in CO_2 emission from cultivated Histosols have been noted by Elder and Lal (2008), although the response found in their study (c. 625 mg CO_2-C m^{-2} h^{-1}) was several-fold greater than in this one. Mean emissions from a bare-tilled peat measured by Maljanen *et al*. (2002) (300 mg CO_2-C m^{-2} h^{-1}) were also higher than the peak emission of 134.51 ± 5.18 mg CO_2-C m^{-2} h^{-1} recorded in this study. In accordance with Elder and Lal (2008) and Maljanen *et al*. (2002), this study found that increased air and soil temperature were significantly correlated with elevated soil respiration, but did not observe the relationship Elder and Lal (2008) reported between increased rainfall and suppressed CO_2 emissions. The lack of a significant response to rainfall in this study's ploughed cores may be related to soil drainage. Here, the significantly lower bulk density after 153 d in ploughed cores compared to control cores (Table 5.2.) indicates comparatively faster drainage in ploughed cores, which may have counteracted any suppression of CO_2 emission resulting from soil moisture accumulation. No details of soil bulk density are provided by Elder and Lal (2008), but a lack of significant difference in drainage between control and ploughed plots could have allowed for a more pronounced effect of rainfall. There were significant positive correlations between dissolved NO_3-N and NH_4-N concentration in the soil and CO_2 emission, which may have been indicative of conditions inhibiting N_2O emission (with N therefore remaining in the soil), but more favourable for soil respiration.

Production of N_2O was not stimulated by a ploughing event. This is in contrast to the findings of Elder and Lal (2008), although Maljanen *et al*. (2002) and Weslien *et al*. (2012) report negligible effects of ploughing on N_2O emission. It is probable that the considerably lower peak N_2O emissions observed here compared with those of Elder and Lal (2008) are a result of suboptimal soil moisture conditions inhibiting N_2O production, again relating to the comparatively good drainage and low bulk density of this study's ploughed cores (Dalal *et al*., 2003). Correlation analysis in this study did not reveal temperature as a significant predictor of emissions, despite fluxes over time tending to track temperature patterns (Table 5.3.); rainfall was negatively associated with N_2O emission, despite observing that the soil surface was considerably drier in the C2 treatment than the control throughout most of the experimental period. It is possible that differing interactions between soil factors in the control and cultivated cores simply resulted in coincidental similar N_2O emissions from both treatments.

Both control and ploughed treatment CH_4 fluxes were highly variable and apparently unrelated to the cultivation event in this study (section 5.3.3.6., Fig. 5.4.), aside from the peak on day ten which may have represented a release of trapped CH_4 from lower soil layers in response to ploughing. Again, background CH_4 fluxes are similar to those observed in other arable peats with a history of cultivation (e.g. Nykanen *et al*., 1995).

Ploughing resulted in a non-significant, slight reduction in cumulative GWP_{100} over the 80 d measurement period. Similarly, Elder and Lal (2008) found a small, non-significant difference between CO_2 emissions from conventionally-tilled and no-till cropped peat soils, although in contrast to this study's findings, they observed a higher mean flux from ploughed than no-till soils. This study's results are in strong contrast to the widespread theory that cultivation results in a large efflux of both CO_2 and N_2O (e.g. Dawson and Smith, 2007; Kasimir-Klemedtsson *et al*., 1997). The cores in this experiment were taken from the 0-30 cm soil layer of previously cultivated and homogenised soil. Cultivation of strongly decomposed peats tends to produce a smaller GHG response than when virgin peat is cultivated (e.g. Cannell *et al*., 1999), which, combined with low soil moisture, is probably the reason for a negligible response in this study.

5.4.4.2. Effect of surface residue application

Elevated levels of both CO_2 and N_2O production were observed for 20-30 days after surface application of lettuce residue when compared to a no-residue control, with an initial peak one to six days after residue application. Other studies have similarly found an increase in CO_2 and N_2O within 1-10 days of crop residue application, both in mineral soil mesocosms (e.g. Giannopoulos *et al*. 2010; Loecke & Robertson, 2009; Velthof *et al*. 2002), and on peat soils (e.g. Elder and Lal, 2008).

The strength of observed CO_2 response after surface residue application (196.47 ± 13.59 mg CO_2-C m^{-2} h^{-1}) is an order of magnitude lower than that attributed to residue-induced CO_2 emission by Elder and Lal (2008) (785.83 mg CO_2-C m^{-2} h^{-1}), but higher than the 92.79 mg CO_2-C m^{-2} h^{-1} flux following harvest measured by Morrison *et al*. (2013). Similarly, the magnitude of residue-induced peak N_2O fluxes varies widely between studies (e.g. Kasimir-Klemedtsson *et al*., 2009; Rochette *et al*., 2010; Weslien *et al*., 2012), with this study's peak N_2O emission (496.21 ± 171.52 µg N_2O-N m^{-2} h^{-1}) towards the lower bound of observed emissions.

The pattern and magnitude of CO_2 and N_2O fluxes in this study may be attributed in part to the characteristics of the residues added. In a study comparing emissions from soils amended with crop residues with differing compositions, Velthof *et al*. (2002) observed a rapid response and pronounced peak in N_2O and CO_2 emissions from crops which similarly to this study had a narrow C/N ratio (c. 10-20) and high moisture content (> 80%). Other studies support the theory that crops with low C/N ratios tend to cause greater CO_2 and N_2O emissions (e.g. Loecke and Robertson, 2009), as well as tending to decompose more quickly (Henderson *et al*., 2010), in keeping with this study's results. The total available C and NO_3-N provided by residue inputs both enhance CO_2 and N_2O emission, irrespective of C/N ratio (Henderson *et al*., 2010; Loecke and Robertson, 2009). In this study, the proportion of labile C and N in residues was not measured, but lettuce residues tend to be readily decomposable (De Neve and Hofman, 1996), so a high proportion of labile organic matter can be expected. With these properties of lettuce residue in mind, the emissions observed in this study are lower than expected, and may be explained by the relatively low total quantity of residue C and N added to each core in our experiment (c. 746 mg C, c. 73 mg N) in comparison with other studies (e.g. Velthof *et al*., 2002).

In this study, soil moisture appeared to impact on surface-applied residue CO_2 and N_2O fluxes. The high moisture content of the lettuce residue may have provided a small amount of additional moisture to the soil surface during the first week after application, enhancing microbial activity. Additionally, a significant positive correlation was found between soil respiration and both daily rainfall and total N concentration (Table 5.3.): microbial respiration may have been further enabled by rainfall washing soluble residue N into the soil surface layer. In contrast, there was a significant negative correlation between N_2O flux and daily rainfall (Table 5.3.), and since soil NO_3-N was not limiting (Fig. 5.4.e.), this potentially indicates that the slightly compacted soil surface in this treatment (Table 5.2.) was creating surface WFPS conditions more conducive to N_2 than N_2O production when it rained. A similar effect of pore space size on residue-related N_2O production has been observed by Velthof *et al*. (2002). There was no significant positive correlation between soil temperature and CO_2 or N_2O production, contrary to the relationship observed within other treatments (Table 5.3.). It is likely that the surface residue cover at least temporarily insulated and moistened the soil surface, suppressing the extremes of temperature experienced in the surface soils of other treatments. Novel to the surface residue treatment, N_2O emission showed a negative correlation with air temperature, possibly reflecting an interaction with soil surface compaction and moisture, whereby denitrifier activity was increased at higher air temperatures, but producing N_2 rather than N_2O.

Methane fluxes from the surface-applied residue treatment were highly variable (section 5.3.3.6., Fig. 5.4.d.), and were within the upper range of those observed by Elder and Lal (2008) and Kasimir-Klemedtsson *et al*. (2009) from residue addition to peat soil. The small positive peak on day three may have been a response of residue application interacting with soil surface moisture and compression, as was suggested in relation to N_2O emissions. Prominent CH_4 uptake episodes occurred on days 17, 44, and 81 (Fig. 5.5.d.), and were significantly associated with low soil and air temperature (Table 5.4.).

The overall effect of surface residue application on cumulative net emissions was a significant increase (Table 5.5.). Although few other studies are available for direct comparison of GWP_{100}, the net effect of surface residue addition appears to vary depending on the total quantity added (e.g. Weslien *et al*., 2012), and the quality of residue (C/N ratio; total labile C and N; Velthof *et al*., 2002). Particularly relevant to organic soils, residue addition may result in “priming”: the enhanced decomposition of soil organic matter when nutrient (C or N) amendments are added to the soil (Bingeman *et al*., 1953). The mean cumulative C loss from the surface-applied residue cores was 0.31 t C ha^{-1} 81 d^{-1} greater than from the control, and the quantity of residue C added to the cores was only the equivalent of 0.90 t. The equivalent of 88.1 kg N was added to the residue cores, but only 1.85 kg N ha^{-1} 81 d^{-1} was lost. In this experiment, the relative carbon budgets of the C1 and R treatments suggest that adding lettuce residue to the soil did not induce priming when residues were surface-applied, although it is unclear whether the C and N loss through GHG emission was wholly from the residue itself, or partly from the soil as well.

5.4.4.3. Effect of incorporated residue application

In this study, residue-induced CO_2 and N_2O peaks were generally smaller when residues were incorporated than when they were surface applied, although the decomposition and mineralisation profiles are different to the R treatment, with initial response peaks (again, within 1-6 d of incorporation) greater than when residue was surface applied (section 5.3.3.6.; Figs. 5.5.b., c.). For the remainder of the experimental period, patterns of soil respiration and nitrous oxide were very similar to those of the surface-applied residue treatment (Figs. 5.5.b., c.).

Peak soil respiration effluxes associated with residue incorporation (193.23 ± 3.54 mg CO_2-C m^{-2} h^{-1}) were low compared to incorporated residue in a mineral soil (c. 50 to 500 mg CO_2-C m^{-2} h^{-1}, Loecke and Robertson, 2009) and in a peat soil (c. 625 mg CO_2-C m^{-2} h^{-1}, Elder and Lal, 2008), although they were similar to post-harvest soil respiration rates (92.79 mg CO_2-C m^{-2} h^{-1}) measured by Morrison *et al*. (2013). Peak N_2O emissions observed in the incorporated residue treatment (413.33 ± 168.46 µg N_2O-N m^{-2} h^{-1}) were also low compared to those found in the literature (c. 300 to 4375 µg N_2O-N m^{-2} h^{-1}, Elder and Lal, 2008; Kasimir-Klemedtsson *et al*., 2009; Loecke and Robertson, 2009), although Weslien *et al*. (2012) reported a minimal additional effect of residue incorporation at harvest compared to fluxes during a carrot crop.

The environmental variables significantly associated with incorporated-residue CO_2 and N_2O emissions were similar to those associated with surface-applied residue fluxes (Table 5.3.), indicating that within the RC treatment, the residue component may contribute more prominently to flux patterns than the ploughing component. However, soil bulk density (0-20 cm) was similar to that of the ploughed treatment by the end of the experiment, and significantly lower than that of the R cores’ bulk density, with a corresponding low soil surface moisture content, suggesting an additional influence on emissions from the RC cores. This interaction effect is substantiated by the incorporated-residue peak and cumulative soil respiration and N_2O emissions, which are all intermediate between the corresponding ploughed and surface-applied residue values (Fig. 5.4.b., c., Table 5.5.). The cumulative effect on CO_2 and N_2O fluxes of surface applying or incorporating residue vary in the literature, from a negligible effect (e.g. Elder and Lal, 2008) to incorporation giving a significantly higher flux than surface application (e.g. Giannopoulos *et al*., 2010). It is likely that incorporated residue fluxes in this study were limited by soil moisture content in the top 5-10 cm of the soil, unlike other studies where soil moisture tended to be more similar between treatments.

The variability of RC treatment CH_4 emissions over time was of a similar magnitude to CH_4 emissions associated with other treatments. This treatment yielded the largest effluxes of CH_4 from all cultivation and residue treatments (up to 496.64 ± 485.55 µg CH_4-C m^{-2} h^{-1}), some 17-20 d after incorporation. Again this may reflect the combination of ploughed treatment soil characteristics (a similar pulse of methane from lower soil layers following ploughing, section 5.3.3.6.) and residue application (potential anaerobic decomposition of residue at depth and subsequent release through soil pore space to the surface).

Similarly to the surface-applied residue treatment, incorporating residue into the soil does not appear to result in a priming effect. Mean cumulative carbon loss (0.10 t C ha^{-1} 81 d^{-1} greater than from the ploughed control, from 0.90 t C added), and a mean cumulative N loss (0.6 kg N ha^{-1} 81 d^{-1} compared to the ploughed control, from 88.1 kg N added), were smaller than when residue was surface applied.

5.4.5. Comparison of cumulative emissions with IPCC emission factors

It is difficult to directly compare the cumulative emissions observed over the summer season in this study with other absolute values from the literature, as a variety of experimental time scales and different up-scaling methods have been employed in other studies; consequently, all of the comparisons made here are relative. The pattern of cumulative emissions observed in all of the treatments (dominated by soil respiration, with N_2O less important but still significant, and CH_4 emissions negligible or a small negative flux) is similar to that found in other studies that have simultaneously measured these three GHGs from cropped peat soils (e.g. Elder & Lal, 2008; Furukawa *et al*., 2005; Maljanen *et al*., 2004), and also reflect the IPCC (2000) recommendations for calculating emissions from cultivated Histosols (Webb *et al*., 2014). The IPCC (2000) annual values for GHG emissions from cultivated Histosols are the equivalents of 3.99 to 46.90 t CO_2-e ha^{-1} yr^{-1} carbon dioxide, 2.48 t CO_2-e ha^{-1} yr^{-1} nitrous oxide, and “negligible” methane emissions. Using a simple multiplication to estimate annual values, the range of emissions in this study are within a similar range for CO_2 and CH_4, but up to fifteen times higher for N_2O, although the control cumulative N_2O emission (c. 2.37 t CO_2-e ha^{-1} yr^{-1}) is very similar to the IPCC default value. Refinements to EFs according to the IPCC 2006 and 2013 guidelines give a higher estimated EF for N_2O emission, so are closer to the values observed here. If the effect of draining after water table raising to 15 cm below the soil surface is also taken into account, then even the most recent cultivated Histosol EF is likely to underestimate annual CO_2 emission.

The IPCC inventory method provides emission factors (EFs) for N_2O fluxes from cultivated soils, additional to the annual value for cultivated peats (Webb *et al.*, 2014). Amongst the treatments, the only EF provided relates to residue addition (1.25 % of N added, equating to a predicted loss of 11.0 kg N_2O-N ha^{-1} yr^{-1}). Subtracting this EF from the total N_2O emission from each residue treatment's emissions indicates whether residue addition has altered the underlying default EF corresponding to temperate drained and cultivated Histosols. Surface residue application resulted in an underlying EF of 2.85 kg N_2O-N ha^{-1} yr^{-1}, and residue incorporation gave a value of -3.33 kg N_2O-N ha^{-1} yr^{-1}, both of which are lower than the IPCC Tier 1 EF for temperate drained and cultivated Histosols (8 kg N_2O-N ha^{-1} yr^{-1} under the IPPC 2000 and 2006 guidelines, and 13 kg N_2O-N ha^{-1} yr^{-1} under the IPPC 2013 guidelines). Residue treatment also affected the underlying cultivated Histosol EF for CO_2 emission. Subtracting the applied residue C (0.90 t C ha^{-1} yr^{-1}) from the annual emission estimate from the surface-applied and incorporated residue treatments gives an underlying EF of 7.90 and 6.56 t CO_2-C ha^{-1} yr^{-1} respectively. While these fall within the range suggested by the Tier 3 IPCC 2000, and Tier 1 2006 and 2013 guidelines (1 to 10, 5.00 ± 4.50, and 7.90 ± 1.5 t CO_2-C ha^{-1} yr^{-1} respectively), they are closest to the most recent Tier 1 estimates.

Pronounced differences in estimated annual emissions from the different treatments explored in this study, and those recommended by IPCC guidelines, suggest that further refinement of EFs according to differences in cropping and crop management (particularly in relation to water table regulation) should form the foundation of appropriate higher tier EF definition.

5.4.6. Implications for horticultural management

The primary objective of this study was to evaluate the effect of manipulating water table depth, fleece application, ploughing, and residue application on net GHG emissions, thereby identifying farming operations which may be effective in mitigating emissions and promoting SOC retention in horticultural peat soils. This was a novel study as it simultaneously tested a number of mitigation options on peat soils, for all three GHGs.

The results of this study suggest that raising the water table to the soil surface, minimising the period over which fleece is applied, and incorporating residues into the soil to a depth of 30 cm instead of leaving them on the soil surface post-harvest have the potential to mitigate overall GHG emissions. However, the results also illustrate that the environmental conditions under which mitigation operations are implemented may affect their efficacy, so these recommendations are offered with some caveats. Further, it is likely that each mitigation measure will have wider agronomic implications in terms of economic cost, effect on crop yields, and practicality of implementation; these considerations are discussed here.

5.4.6.1. Implications of seasonal variation in weather conditions

The mean CO_2 and N_2O fluxes from this experiment are of a comparable magnitude to the fluxes observed between May and November on the 70% SOM soil in the field study (Chapter 2), disregarding the fluxes from WL and WH treatments, which were not encountered under field conditions. Methane flux variation fell within a similar range to the field study, but had greater variability. Mean and extremes of temperature and rainfall experienced during May-November during the field study in East Anglia (2011-2012) and this cores experiment in North Wales (2013) were comparable, and given that CO_2 and N_2O fluxes were both associated with temperature and rainfall, probably accounts for the similar range of CO_2 and N_2O fluxes in this experiment and the field study. The greater variability observed in CH_4 fluxes may be a result of using more frequent sampling dates in relation to farm operation events in this controlled experiment, with the associated greater chance of capturing extremes in emission than the more sporadic measurements taken in the field.

Since higher temperature and lower rainfall conditions seem to be associated with higher GHG emissions and intensified differences between treatments and controls, it is recommended that mitigations should be implemented in warmer, drier weather (i.e. late spring and summer in the UK), when they are likely to yield the greatest reduction in net emissions.

5.4.6.2. Implementation of water table raising

This study simulated raising the water table during late spring followed by draining in late summer, mimicking the water management regime commonly employed by farms in the study area to enable sub-surface irrigation and minimise peat loss through wind erosion (Dawson *et al*., 2010). Under normal conditions, the water table would only be raised to 50 cm below mean field surface level (Dawson *et al*., 2010), but this study investigated the effects of an extreme version of this intervention, in order to investigate the sensitivity of the soil surface layers to flooding.

One limitation of this study is that it was conducted on a small scale, without the presence of crops. In practice, raising the water table to within 15 cm of the soil surface would not be practised while a crop was in place, as it would likely kill the crop and make tractor operations impossible. Instead, this intervention would probably be implemented between summer crops, possibly over quite short fallow periods. Therefore, this experiment may give a reasonably accurate representation of the relative effects of the water table treatments.

The relative efficacy of flooding in practice may be limited by a number of factors. Some weed growth on these highly fertile sites is likely even during relatively short fallow periods, which could further reduce net GWP through elevated net primary productivity (Morrison *et al*., 2013). Conversely, both the presence of weeds and labile organic matter input from post-harvest crop residues could result in substantial emissions of N_2O and CH_4 in particular (e.g. Le Mer and Roger, 2001). The net effect of vegetation merits further investigation at the field scale.

Maintaining the water table at the correct level and ensuring it drains quickly enough post-flooding could be challenging. Kechavarzi *et al*. (2007), suggest that close spacing of sub-surface drainage pipes (≤ 10 m) would be required to maintain a consistent water table level in a sub-irrigated field. Some fields are not equipped with closely spaced drainage pipes, and not all peat soils are sub-irrigated. Surface-level flooding may be easier to implement via direct flooding from drainage ditches, but maintaining the water level may be difficult in hot weather due to evaporation or statutory water use restrictions. Fluctuation of the water level between 0-15 cm of the soil surface, either through poor water level maintenance or slow draining post-flooding, is likely to result in large pulses of GHG emission, as was observed in the 15 cm water table treatment, negating the effect of flooding altogether. This effect may be minimised if draining were done in cooler weather.

Flooding poses a number of difficulties both agronomically and in the context of the wider landscape. Implementation would require careful timing so that after flooding, soil had time to dry sufficiently before subsequent in-field machinery operations. Yields of subsequent crops could be reduced after flooding, or the costs of mineral fertiliser increased: this study's results strongly implied that much of the soil nitrate was leached from the soil columns during draining. In terms of wider landscape effects, leaching of nitrate into watercourses poses a severe pollution risk, with associated costs for the grower. Further, if flooding were to be implemented on a widespread scale, regulation would be required to ensure that it did not adversely impact on flood risk and response across the region, which would be challenging across areas of flat topography.

5.4.6.3. Implementation of fleece application

The intention in this experiment was to observe the effect of fleece on GHG emissions over a range of weather conditions. The greatest emissions from the fleece treatment were observed when the air temperature was highest. In practice, fleece would be usually only be applied to early crops, to minimise the risk of frost damage and encourage early crop development (Hamouz *et al*., 2006). The presence of fleece did increase net emissions under cooler as well as warmer temperatures, but by a smaller amount (Fig. 5.3.). It is important therefore, to restrict fleece application to as short a period as possible during cooler weather, as is common under current practice.

As with the water table treatments, the effect of fleece application in the presence of a crop should be investigated at the field scale, to compare crop growth and associated net ecosystem exchange between fleece and control treatments, as this may further reduce the difference in emissions. It would also be of interest to consider the effect on net emissions when fleece is applied over recently-fertilised peat, as the results suggest that N_2O emissions may substantially increase when fertilised soil is subjected to the warmer soil temperatures associated with fleece application. Although the soil nitrate concentrations were relatively high (Fig. 5.3.e.), they did not reflect the high levels commonly found directly after mineral N application (e.g. Bouwman *et al*., 2002). The results show significantly better retention of both NO_3-N and available P by the end of the experiment (Table 5.2.). It would be useful therefore, to examine at the field scale, for typical time-periods of fleece application, the balance between the benefits of apparent nutrient retention and the problematic potential increase in GHG emissions under fleece.

5.4.6.4. Implementation of ploughing

Similarly to some other studies, this study found a minimal effect on emissions of ploughing, probably related to the fact that these are pre-ploughed fields. It is therefore recommended that when peats are being deep-ploughed into soil layers that are not already well-mixed and largely decomposed, that this is done in cooler weather, and to further reduce the disruption caused by ploughing by reducing the ploughing depth and the frequency of deep ploughing if possible.

One limitation of this study was the difference in surface soil moisture observed between the control and ploughed treatments (Table 5.2.). This may be less likely under field conditions, but comparison of GHG emissions and associated soil characteristics from ploughing vs. minimal- or no-till treatments, at different times of the year, warrants further investigation.

5.4.6.5. Implementation of residue incorporation

This study found that net cumulative GHG emissions from incorporated lettuce residue was significantly lower than from surface-applied residue (section 5.3.5.3.). However, there are a number of aspects of these interventions that may differ from this mesocosm study when applied in the field.

Net emissions from the incorporated residue treatment were found to be intermediate between those from ploughed and surface-applied residue treatments. As recommended for the ploughing treatment, this mitigation measure should be tested at the field scale at the appropriate time of year, as ploughing under different soil and weather conditions may result in the net emission from incorporated residue exceeding that of surface-applied residue, possibly with an associated priming effect not seen in this experiment. This should be investigated at different points during the summer season to reflect the practice of sometimes double- or triple-cropping fields (Chapter 2), with associated differences in harvesting times.

Net emissions from residue application may differ greatly according to crop dry matter content, C/N ratio, availability of labile C and N, and the total quantity of residue applied (Velthof *et al*., 2002; Webb *et al*., 2014). The range of crops grown at the study site varied in each of these characteristics, so a larger-scale study exploring the differences in emissions from a number of commercially important crops would be useful to gauge the range of expected emissions from surface-applied or incorporated residues on horticultural peat soils. Further, the particle size and distribution of residues across or within the soil can significantly influence overall emissions (Loecke and Robertson, 2009). It would be worth considering if these factors could be varied to reduce emissions further, being mindful of balancing reduced emissions with the increased economic costs associated with ploughing in, aggregating or spreading residues after harvesting.

5.5. Conclusions

This study has identified a number of important factors that drive on-farm GHG mitigation efficiency. Effective mitigation implementation is contingent on careful evaluation of a range of climatic and soil related co-variables. For example, flooding the soil surface might be advisable and practicable under restricted circumstances; a less effective option might be to maintain the water table at a lower depth, but above the level of drainage pipes on the outside of the field to prevent emissions escaping via that route. Most importantly, avoiding a water table level at c. 15 cm below soil surface is paramount in minimising emissions. Flooding should be practised during summer fallow periods for maximum mitigative effect.

The findings from this study suggest the use of horticultural fleece during the shortest possible period, and in cooler weather only, which should pose few practical difficulties as it in effect maintains the management *status quo*.

Recommendations regarding ploughing and residue application are difficult to make with the present available evidence, which may not reflect typical field conditions. Generally, ploughing and harvesting operations should be conducted during cooler or damper weather to minimise the initial peak in emissions, although this is likely to be somewhat impractical in relation to harvesting operations.

The relative efficacy of the limited number of potential GHG mitigation options investigated here is influenced by the weather and soil conditions at the time of implementation and usually for approximately one month following implementation. Mitigation measures hold the greatest potential efficacy if applied during the main cropping season. In addition to the efficacy of mitigation measures considered here, future research should focus on the practicality of implementation, economic cost, and effects on crop quality and yield. Serious consideration should also be given to mitigation involving changes to less intensive land uses, such as extensively managed pasture with a high water table.

Chapter 6

Discussion

The primary objectives of this study were to quantify GHG emissions from peat soils under horticultural vegetable production in the UK, and to investigate a number of candidate measures with the potential to reduce net GHG emissions from these soils. These research aims were met using a mixed methods approach. Two methods were explored for quantitatively estimating emissions: a field-based study using gas collection chambers, and a computer modelling study. Additionally, two methods were used to explore GHG mitigation options: an online questionnaire, and a field-based study of soil core mesocosms.

This study had four specific research aims:

1. To produce a full annual GHG budget for a number of commercially important vegetable crop rotations, grown on soils of contrasting, medium-to-high soil organic matter content (‘horticultural peat soils’).
2. To assess the performance of a biogeochemical model (the DNDC model) in predicting GHG emissions from horticultural peat soils.
3. To identify candidate mitigation options with the potential to reduce net GHG emissions from UK horticultural peat soils.
4. To evaluate the effectiveness at reducing emissions, and practicality of implementation, of a selected shortlist of potential GHG mitigation options.

The thesis explored a number of novel applications of existing methods. In Chapter 2, the first known annual GHG budget for UK peat soils under horticultural production is estimated. The DNDC model is used to predict GHG emissions, in a first application to UK horticultural peat soils in Chapter 3. Further innovations included the use of a sensitivity analysis to inform model calibration, and the use of fit statistics to identify sources of error within sub-models. Chapter 4 explored the perceptions of experts and farmers concerning GHG mitigation in the UK horticultural sector, through a novel application of a Best-Worst Scaling survey technique. In Chapter 5, the efficacy of a number of GHG mitigation options were simultaneously evaluated on GHG emissions for a peat soil, including the previously unexplored effect of horticultural fleece application.

In order to place the thesis within a wider context, this chapter discusses the key strengths and weaknesses of GHG measurement methodologies, the importance of horticultural peatland emissions within the entire UK agricultural and land use emissions sector, and the implications of the key findings for policy-makers and other stakeholders.

6.1. Strengths and weaknesses of the methodology

6.1.1. Upscaling of site-scale GHG measurements

Both the field-based chamber method used in Chapter 2, and the computer modelling approach employed in Chapter 3, explore GHG emissions at the site scale, while the mitigation study in Chapter 5 investigates effects on fluxes at a scale of less than one square metre. Small-scale observations enable an understanding of the environmental factors influencing GHG fluxes at a relatively high resolution (Chapter 1), but emissions estimates used for the purposes of national inventory need to be up-scaled appropriately. The IPCC three-tiered system of emissions estimation, recommends using the highest spatial and temporal resolution data available, which can include both direct measurement and mathematical modelling approaches (IPCC, 2006; Webb *et al*., 2014). National estimates are disaggregated into a number of source components, each of which comprises an emission factor (EF, indicating an estimated rate of GHG flux associated with a particular activity), and an area or quantity multiplier (indicating the extent of that activity over different spatial scales). The accuracy of aggregated national estimates is therefore contingent upon the availability of high quality data, representing a reasonable estimate of the range of spatial and temporal variability found within the system of interest.

Many reviews of potential sources of error arising from the use of closed chambers and IRGAs have been published (e.g. de Klein and Harvey, 2013; Levy *et al*., 2011; Pumpanen *et al*., 2004). The accuracy of chamber-derived emissions estimates are dependent on a multitude of factors relating to chamber design and deployment; gas sample collection, storage and analysis; and flux calculation methods (de Klein and Harvey, 2013; Chapter 1). Capturing a representative range of spatial and temporal heterogeneity requires optimal allocation of limited research resources over a sufficient range of spatial and temporal conditions. For the purposes of compiling a national inventory estimate of horticultural peat soil emissions, spatial heterogeneity may exist at the national, regional, local, field, and within-field scales. Temporal heterogeneity in the national inventory context occurs at scales from inter-annual to diurnal. Both temporal and spatial heterogeneity derive from variation in soil, climatic, and crop management conditions (e.g. Chen *et al*., 2008; Li, 2007).

In this thesis, attempts were made during the experimental design stage to account for common sources of variation. Current best practice was considered in chamber design, deployment protocol, gas sample collection, storage and analysis; and flux calculation methods (Chapter 2; Chapter 5). Spatial variation at the within-field to local scale was incorporated using randomised plot placement within each field, and by sampling emissions from a range of crops; while regional variation was considered to some extent by sampling across a range of soils of differing SOM contents. The national representativeness of chamber measurements made in Chapter 2 is discussed below (Section 6.2.1.). Addressing temporal variation was a greater challenge, particularly with respect to the field study (Chapter 2). Seasonal variation in GHG fluxes was represented relatively well, by sampling over the full annual cycle. A measure of diurnal flux variation was provided using a proxy from the literature, but only for CO_2 (due to a lack of robust information relating to N_2O and CH_4 diurnal correction indices). Consequently, diurnal variation may constitute a considerable source of error in this study, and is an issue that merits further investigation. A further weakness in the field study, common to manually closed chamber studies, is the infrequency of GHG observations (c. once per month). This may have resulted in the accidental omission of important emission peaks, particularly short-lived N_2O pulses (e.g. after fertiliser application), with resultant inaccurate annual estimates calculated from the available data points. This issue could be overcome to some extent by combining data from field studies with calibrated and validated computer-simulated data (Chapter 3), or by augmenting the information provided in this thesis with data provided by finer temporal resolution techniques (e.g. micrometeorological or automatic chamber techniques, Chapter 1).

Process-based computer models have the potential to provide insight into short-lived emission events within a full annual cycle that sporadic field measurements often cannot. This study demonstrated that for models such as DNDC to provide effective GHG estimates, high quality input data, and accurate model calibration and validation are required (Chapter 3). This is a matter that requires further attention with regards to DNDC’s modelling of horticultural crops on peat soils. Subject to successful calibration and validation using field data, upscaling to a regional or national level could be achieved using the regional-scale function available in DNDC and some other models (e.g. Brown *et al.*, 2002; Chen *et al.*, 2008; Smith *et al.*, 2010). As for observed data, scaling up site-scale modelled data to the regional or national level requires a robust understanding of sources of variation. Methods for estimating uncertainty associated with simulated data at the country-wide scale are relatively well developed, and broadly comply with IPCC recommendations for uncertainty estimation (e.g. Giltrap *et al.*, 2013; Hastings *et al.*, 2010; IPCC, 2006). Combining observed and modelled estimations of

GHG fluxes is a key step in meeting the requirements of higher resolution UK estimates (IPCC Tiers 2 and 3; IPCC, 2006).

6.1.2. Identifying suitable mitigation measures

This thesis used two approaches to identifying candidate measures with the potential to reduce GHG emissions from peat soils under horticultural production in the UK. A broad range of large- to small- scale MMs were evaluated using expert and grower opinion (Chapter 4), and a selected small number of potential field-scale MMs were assessed in a soil core mesocosm experiment (Chapter 5).

A particular strength of the expert and grower consultation approach is its capacity to assess a large number of nominated interventions concurrently, within a relatively short time frame (Sawtooth, 2013). This is important where the issue under discussion requires an urgent mitigation response, as it quickens the process of implementation. Robust experimental evaluation of all of the candidate interventions identified in the initial literature search, would substantially delay action to mitigate GHG emissions. Further, some of the landscape-scale interventions (e.g. reversion of prime agricultural land to semi-natural habitats) would be difficult to assess experimentally without overcoming considerable practical, economic and social barriers. Evaluation through expert and grower opinion allows immediate mitigation implementation, while the efficacy of measures can be confirmed through field monitoring of implemented measures in the longer term.

Some caution should be applied when using this semi-quantitative opinion-based approach. While this method enables a relatively rapid response to the pressing issue of climate change, it should be noted that it is a subjective process. Prioritisation of measures by the scientific community in the face of conflicting evidence may be driven by somewhat circular modes of information gathering. Subject knowledge derives from exposure to published literature and peer opinions, a process that is unlikely to present an entirely comprehensive overview of the issue, being subject to the current scientific paradigm and possibly driven by the theories or opinions of a small number of individuals.

Soil mesocosm studies provide another means of evaluating GHG mitigation measures simultaneously, and with relatively limited time and funding resources. This approach is subject to the same sources of error as those discussed in section 6.1.1.1., including those related to upscaling. At the relatively small spatial scale of mesocosm studies, upscaling is potentially more problematic than in field scale studies, since not all parts of the system are sufficiently replicated at the small scale. For example, one study on peat soils found differing responses to wetting and drying in mesocosm and field conditions (Dinsmore *et al*., 2009). When using an expert consultation approach, the mesocosm approach could be used to short list (or eliminate) some of the smaller-scale candidate interventions for implementation, with field- and landscape-scale studies augmenting knowledge and understanding in the longer term.

6.2. Magnitude of GHG emissions from horticultural peat soils

6.2.1. Comparison of measured and modelled emissions to IPCC estimates

A comparison of UK inventory method GHG estimates (IPCC, 2000; Webb *et al*., 2014) with observed estimates from Chapter 2 and DNDC-simulated estimates from Chapter 5 is shown in Table 6.1. Despite some variation between soil types, measured and modelled CO_2 emissions from both cropped and bare soils fall within the range suggested by the IPCC guidelines for historically drained and cultivated organic soils, for Tier 1 temperate zone estimates (IPCC 2000; 2006; 2013c) and for Tier 3 UK (2000) estimates. Similarly, IPCC CH_4 emission estimates are confirmed by field observations in this study, and their magnitudes are adequately simulated by DNDC. Measured N_2O emissions were typically two to four times that of the earlier Tier 1 IPCC default EF, and simulated emissions were between approximately half and three times that of the Tier 1 IPCC default EF (2000; 2013c), for cultivated temperate Histosols. The more recent Tier 1 IPCC (2013c) EF has improved this estimate considerably. As previously discussed in Chapter 2, this is probably at least partially because measured emissions estimates represent ‘cultivated Histosol’ emissions plus several additional sources that should be taken into account when compiling a full GHG inventory from an agroecosystem. For the fields observed in this thesis, further emissions sources include direct N_2O emissions from mineral fertiliser application and crop residue application to fields, and indirect emissions from atmospheric N deposition.

6.2.2. Assessing emissions across the UK horticultural sector

A full assessment of emissions across the whole UK horticultural sector, would include all relevant emissions sources within the ‘Agricultural Soils’ category and other inventory emissions categories, and would consider within-sector variation in agricultural practices on peat soils. Additional agricultural N_2O sources might include biological N fixation by leguminous crops, or for some perennial horticultural crops such as fruit trees, application or deposition of animal manures (Webb *et al*., 2014). Additional LULUCF CO_2 emissions could include those associated with applying lime amendments to the soil, while emissions from the Energy sector could include those associated with tractor fuel combustion, and electricity use in agricultural buildings (Webb *et al*., 2014).

Table 6.1. Emissions of CO_2, N_2O and CH_4 from horticultural peat soils, estimated by the UK 2014 National Inventory method (IPCC, 2000), the IPCC (2006) and IPCC (2013) methods, directly measured from soils, and modelled using DNDC.

Source	**Site, (soil type)**	**Cropping**	**Annual emission (t CO_2-e ha^{-1} yr^{-1})**			
			CO_2	**N_2O**	**CH_4**	**GWP_{100}**
IPCC (2000)	**Histosol**	Cultivated	3.99 , 46.90 [1]	3.90	‘negligible’	7.89 to 50.8
IPCC (2006)	**Histosol**	Cultivated	18.32 ± 16.49	3.90 (0.97 to 11.69)	‘negligible’	22.22 (2.8 to 46.50)
IPCC (2013)	**Histosol**	Cultivated	28.95 ± 5.13	6.33 ± 2.34	0 ± 0.06	35.28 (27.75 to 42.81)
Measured	**L20,**	Cropped	19.29 ± 2.76	6.90 ± 2.07	-0.01 ± 0.09	26.19 ± 3.35
	(c. 20% SOM)	Bare	13.04 ± 2.39	4.94 ± 0.77	-0.02 ± 0.08	17.96 ± 3.10
	M35,	Cropped	30.85 ± 2.54	7.93 ± 0.78	0.04 ± 0.02	38.82 ± 2.38
	(c. 35% SOM)	Bare	21.52 ± 1.13	13.88 ± 1.91	-0.004 ± 0.01	35.39 ± 1.96
	S70,	Cropped	28.26 ± 2.25	7.90 ± 2.23	0.01 ± 0.04	36.17 ± 2.14
	(c. 70% SOM)	Bare	26.03 ± 2.50	6.66 ± 1.49	0.04 ± 0.05	32.73 ± 2.30
Modelled [2]	**M35**	Cropped	35.53	7.56	0.00	43.09
	(c. 35% SOM)		(33.12 to 39.43)	(3.84 to 12.09)		(36.96 to 51.52)
	S70,	Cropped	29.63	3.67	0.00	33.30
	(c. 70% SOM)		(17.95 to 38.11)	(1.16 to 8.22)		(19.11 to 46.33)

All IPCC quoted values refer to Tier 1 estimates for temperate cultivated Histosols, unless stated otherwise. [1] *Corresponding to Tier 3 estimates for English ‘shallow peats’ (< 1 m deep, with c. 12% SOC content) and* ‘deep peats’ *(> 1 m deep, with c. 21% SOC content) respectively (IPCC, 2000; Webb, 2014).* [2] *No S.E. available; range provided refers to minimum and maximum modelled emissions from individual fields.*

Successful estimation of GHG emissions across all UK horticultural peat soils requires a sufficiently accurate estimate of the total area of peat soils under horticultural production in the UK. There is currently considerable uncertainty over the geographical extent of this specific soil type, so it is difficult to make a reasonable estimate of the contribution of horticultural peat soil emissions to the UK GHG budget (Chapter 1). Accurate estimates of peat depth for the UK would facilitate assessment of the extent of the remaining soil carbon stock reserve and the urgency of mitigation efforts required.

Evaluating the variability in GHG emissions across the UK horticultural sector as a result of different cropping and farm management practices is also important. This study included a representative selection of the range of crops grown on peat soils in the East Anglia region, but did not measure emissions from all possible UK crops (notably, *Alliums*, carrots, and fruit crops). Studies on cultivated peats in other countries indicate that crop choice can significantly affect net GHG budgets (e.g. Weslien *et al.*, 2013; Chapter 1). Sensitivity of emissions to crop choice and management practices was confirmed in this thesis by the results of the DNDC modelling exercise, the field study, and the mesocosm experiment, and is discussed further in Chapters 2, 3, and 5. Characterising this source of variability could be achieved using a calibrated mathematical model such as DNDC, which has been validated against other crops in the UK or on horticultural peat soils in other temperate zones. Globally however, complete GHG budgets on Histosols under horticultural production still remain scarce.

If the IPCC guidelines estimate of 158,500 ha of UK arable peat soils is used (Chapter 1), total annual UK emissions are estimated at between 2,847 and 6,153 kt CO_2-e yr^{-1} (measured), or between 3,029 and 8,166 kt CO_2-e yr^{-1} (modelled). This calculation assumes that the range of emissions measured or modelled in this work is representative of all arable crops grown on drained peatlands, and provides a maximum estimate. Conversely, by assuming that the horticultural area on peat soils is proportional to the share of UK fen peat occupying grade 1 agricultural land (Chapter 1), a minimum estimate is provided. Using the same emissions values from Table 6.1., this gives a minimum UK annual estimate of 1,395 to 4,025 kt CO_2-e yr^{-1} (measured), or 1,484 to 5,341 kt CO_2-e yr^{-1} (modelled).

More comprehensive estimates of emissions budgets for comparison within and between agricultural and land-use sectors can be compiled using carbon footprinting (CF), and life-cycle assessment (LCA) protocols (e.g. Edwards-Jones *et al.*, 2008; Jones *et al.*, 2014). While precise application of each technique varies between studies, standardised guidance on application across the agricultural sectors is available through international and national certification schemes such as ISO standards and PAS 2050 (e.g. BSI, 2011; ISO 2006a; 2006b; 2013). Application to the UK horticultural sector has included studies of a range of field-grown and glasshouse crops, including apples, broccoli, green beans, lettuce, sugar beet, and tomatoes (e.g. Edwards-Jones *et al.*, 2008; Mila í Canals *et al.*, 2008). These studies form a reasonably robust basis on which to build future comparisons, although some refinement of input data and methods would be required. For example, a complete evaluation of on-farm net emissions could include carbon sinks such as carbon sequestered in farm hedgerows and windbreaks, and emissions of CH_4 and N_2O from field drainage pipes and ditches (IPCC, 2013c; Jones *et al.*, 2013). It would also be useful to quantify soil losses via wind erosion and crop adherence routes, as current data relating to cultivated lowland peats are almost entirely lacking.

6.3. Emissions mitigation and policy implications

This thesis advocates a mixed methods approach to facilitate a swift response to GHG mitigation; this is particularly important in the UK horticultural sector, which has a narrower evidence base compared to other agricultural sectors (Chapter 4). Implementing GHG mitigation measures impacts on more than simply the net GHG output or uptake of a system (as accounted for using LCA or CF approaches). For policy-making purposes, evaluation of a range of alternative management practices or land uses should also address the potential wider environmental effects, and social, cultural and economic implications of implementation (DeFries *et al.*, 2004; Norse, 2012). The range of ecosystem services provided by peatlands under different land use scenarios may differ in their compatibility, so that careful consideration should be given to the potential trade-offs required when applying interventions. This might be a particular problem when balancing the carbon stock and food provision functions of peatlands. For example, in this study, the measures perceived as being the most effective at reducing net emissions from horticultural peat soils involved major land use changes – chiefly, reversion to semi-natural wetland ecosystems or low-intensity pasture (Chapter 4 and 5). Conversely, such measures were perceived as least practical by farmers, presumably because applying land use changes would clearly have serious implications for farming livelihoods in their current form.

One potential solution for farmers in the case of a national drive to move horticultural production away from peatlands, would be to only farm horticultural crops on mineral soils. This shift could present GHG emissions displacement issues at the national or international level, rather than resulting in a net emissions reduction, but it does offer the major advantage of preserving the peatland carbon stock. Although mineral soils tend to yield CO_2 emissions that are significantly lower than those from peat soils, this is not always the case, and N_2O emissions may under some circumstances be considerably higher from mineral soils. Net GWP_{100} depends on factors such as annual cropping intensity, rate of residue return to the field, and fertiliser application rates (Chapter 1). Estimates in the literature of CO_2 emissions from temperate vegetable crops grown on mineral soils vary from 3.8 to 50.9 t CO_2-e ha^{-1} yr^{-1}, with the UK estimates at the upper end of the range (Bessou *et al.*, 2010; Jia *et al.*, 2012; Koerber *et al.*, 2009; Koga *et al.*, 2006). Temperate vegetable N_2O emissions vary from 0.1 to 1.9 t CO_2-e ha^{-1} yr^{-1} (Bessou *et al.*, 2010; Burton *et al.*, 2008; Smith *et al.*, 1998; Vermeulen and Mosquera, 2009; Xiong *et al.*, 2006), while CH_4 emissions are negligible, at < 0.01 t CO_2-e ha^{-1} yr^{-1} (Bessou *et al.*, 2010). The potential for emissions mitigation is therefore uncertain, and would be contingent upon minimising emissions associated with fertiliser applications, crop residue breakdown, and plant respiration.

Shifting intensive cultivation from peat to mineral soils may become less feasible in the future, as pressure for food production increases in the face of climate change and world population growth (Godfray *et al*., 2010). Other potential mitigation options, which would avoid farmer displacement, might include a change of agricultural land use to low-intensity grassland, with maintenance of a high water table for part of the year. While this might help preserve the soil carbon stock, emissions of CO_2 and N_2O can still be substantial, with the additional consideration from a whole-farm perspective of introducing livestock-induced emissions (e.g. Jones *et al*., 2014). From a national perspective, exchanging horticultural crops for low-intensity livestock grazing on peat soils, and livestock for horticulture on mineral soils where possible, may reduce emissions – both livestock and horticultural emissions would simply been displaced, whilst removing the problematic high emissions associated with intensively cultivated peat soils. Again, a full assessment of net GHG emissions arising from imposing such far-reaching land uses changes should be compared to baseline emissions, before implementation, as well as considering other environmental and socio-economic effects.

Ideally, mitigation efforts should recognise opportunities within the whole food chain (Garnett, 2011). Reducing food waste is crucial, including addressing consumer and retailer attitudes to food quality so that waste at the growing stage might be minimised (e.g. encouraging sales of food produce that is less than visually perfect). Mitigating the environmental effects of food waste would also be required – for example, crops that are not marketable could be ploughed back into the field with the aim of enhancing SOC conservation. The opportunities to reduce food waste between the crop production and consumption stages of the crop life cycle may be substantial, given that in the UK up to an estimated 33% of fresh fruit and vegetables produced may be wasted even before retail (Parfitt *et al*., 2010), with a further 22% wasted by households after purchase (DEFRA, 2014b). Much of the waste created prior to retail is thought to comprise out-graded produce rejected by buyers, with the remainder being unsuitable for harvest due to crop damage (Parfitt *et al*., 2010). If the majority of waste between planting and consumption could be reduced, horticultural production on mineral soils may be able to supply the requirements of consumers, with a subsequent shift to lower-intensity systems on peat soils.

The UK is legally bound to reducing national emissions from all sectors under the EU’s commitment to the Kyoto Protocol, and the UK Climate Change Act (UK Parliament, 2008).

At the level of UK or devolved government administrations, no overarching policy or clearly defined strategy exists in relation to maintaining and restoring peatlands, including their specific contribution to climate change issues (Bain *et al*., 2011; Reed *et al*., 2010). While some agri-environment schemes, or policies relating to biodiversity conservation, contain elements aimed at reducing peatland degradation, these elements have tended to be secondary components (Bain *et al*., 2011). Future policies may need to focus on integrating the national and international needs for peatland services with those of agriculture and other land uses (e.g. similar in principle to the Glastir Agri-Environment Scheme in Wales; Reed *et al*., 2014; Wynne-Jones *et al*., 2013). This would be further aided by development of methods for valuing non-market goods and services (e.g. clean water and carbon storage), so that equivalent comparisons between alternative mitigation scenarios may be made. In the wider context of the food production chain, policies to reduce waste at all stages, and incentivise supermarkets and consumers to view food quality differently, are also critical.

6.4. Recommendations for future research

Based on the points discussed in this section and throughout the thesis, the following research priorities are recommended:

- Quantification of the extent and distribution of horticultural peat soils over the entire UK, followed by upscaling of UK GHG emissions in this sector, encompassing known sources of within-sector variation. Given current and predicted emission rates, this could be extended to forecasting of peatland degradation rates under baseline conditions.
- Field-scale measurements over finer time scales surrounding particular farm operations (e.g. tillage and fertiliser application), to capture rapid responses to operations. This might be achieved using frequent automatic chamber measurements, or micrometeorological techniques.
- Field-scale trialling of mitigation measures previously tested at the mesocosm scale. Measures should ideally be tested within the context of a ‘typical’ annual management cycle, for a range of commercially important crops, on a range of Histosols from minimum to maximum SOM content. The results would provide information on the mitigation potential range of priority interventions.
- In the longer term, interventions could be trialled over multiple years and cropping cycles, to evaluate the cumulative effect of implementation, particularly with regards to SOM accumulation or degradation.
- Formulation of a policy framework incorporating the following: identification of key peatland ecosystem services and stakeholders; valuation of non-market ecosystem services; evaluation of trade-offs between different ecosystem services under both current and potential future management practices and land uses.
- Expansion of the BWS survey to capture a wider farmer respondent base. This may require a broadening of strategies for approaching respondents, e.g. the use of face-to-face paper-based questionnaires.
- Evaluation of the potential uptake rates of shortlisted mitigation measures, through an additional BWS survey, perhaps combined with interviewing farmer focus groups. Potential uptake rates would be compared with farmer views on the practicality of interventions, aiding identification of important components of, and barriers to, MM uptake.

- Construction of MACC (Marginal Abatement Cost Curves) for selected priority MMs identified from the BWS survey results, in order to further enhance understanding of components of and barriers to uptake. This would first require robust carbon accounting of net GHG emissions associated with each measure.

Appendix A

Description of crop management operations for field L1

(Chapter 2)

Table A.1. Crop management operations implemented on field L1 (site L20) during 2011 and 2012 cropping rotations.

Field, year	Crop	Growth period [1]	Tillage	Fertiliser	Ground cover	Irrigation
		dd/mm, (days)	dd/mm, method [2], (depth)	dd/mm, type, (N applied, kg N ha^{-1})	dd/mm, (days), type [3]	dd/mm, (rate, cm)
L1, 2011	Lettuce (Iceberg)	07/06 - 21/07, (44 d)	27/05, PH, (5 cm) 06/07, PH, (5 cm) 09/08, D, (10 cm)	06/06, NPK, (32.6) 08/06, N37, (71.6) 08/06, NPK, (27.7) [4] 23/06, N37, (45.3) [4]		07/06, (1.5) 15/06, (1.5) 07/07, (2.5) 14/07, (2.5)
L1, 2012	Winter wheat	25/10 - 28/08, (308 d)	13/09, SS, (30 cm) 23/10, SS, (30 cm) 25/10, PH, (5 cm)	24/03, N37, (77.8) 17/04, N37, (39.1)		

[1] Growth period is calculated from date of planting or drilling, to date of harvest. [2] BF = bed-forming; D = disked; DP = deep ploughing; MH = mechanical hoeing; P = ploughing; PH = power harrow; SS = sub-soiling. [3] HF = horticultural fleece; IM = insect mesh. [4] Injected, to a depth of 5 cm. [5] Author's estimate (farm records not available).

Appendix B

Details of field sampling block layout and chamber construction

(Chapter 2)

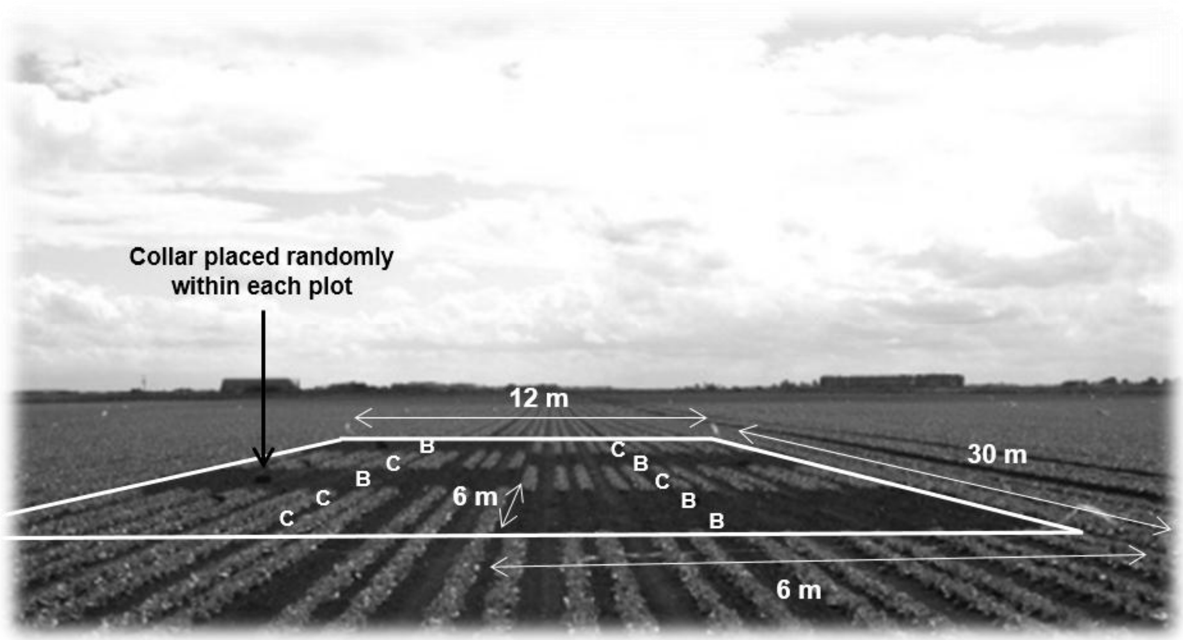

Figure B.1. Typical layout of a field sampling block, containing five randomised pairs of 6 x 6 m cropped (C) and bare soil (B) plots. Each plot contains an inner 4 x 4 m area into which a chamber collar is randomly positioned.

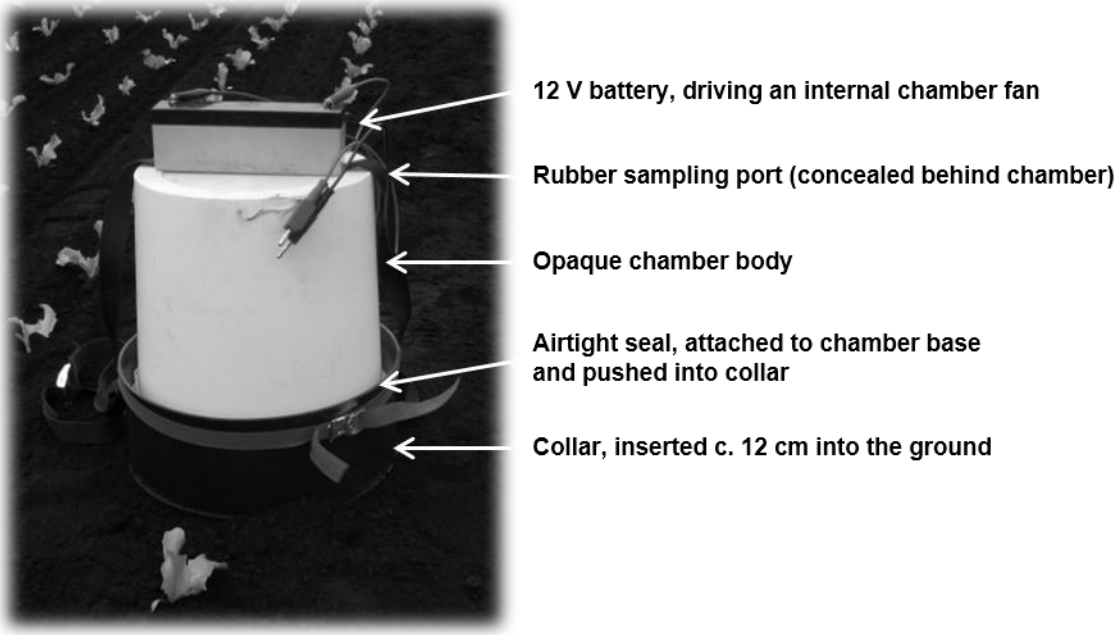

Figure B.2. Static opaque GHG sampling chamber, inserted into a collar, and comprised of a sealed chamber, 12 V fan, and rubber sampling port.

Appendix C

DNDC model pre-simulation results

(Chapter 3)

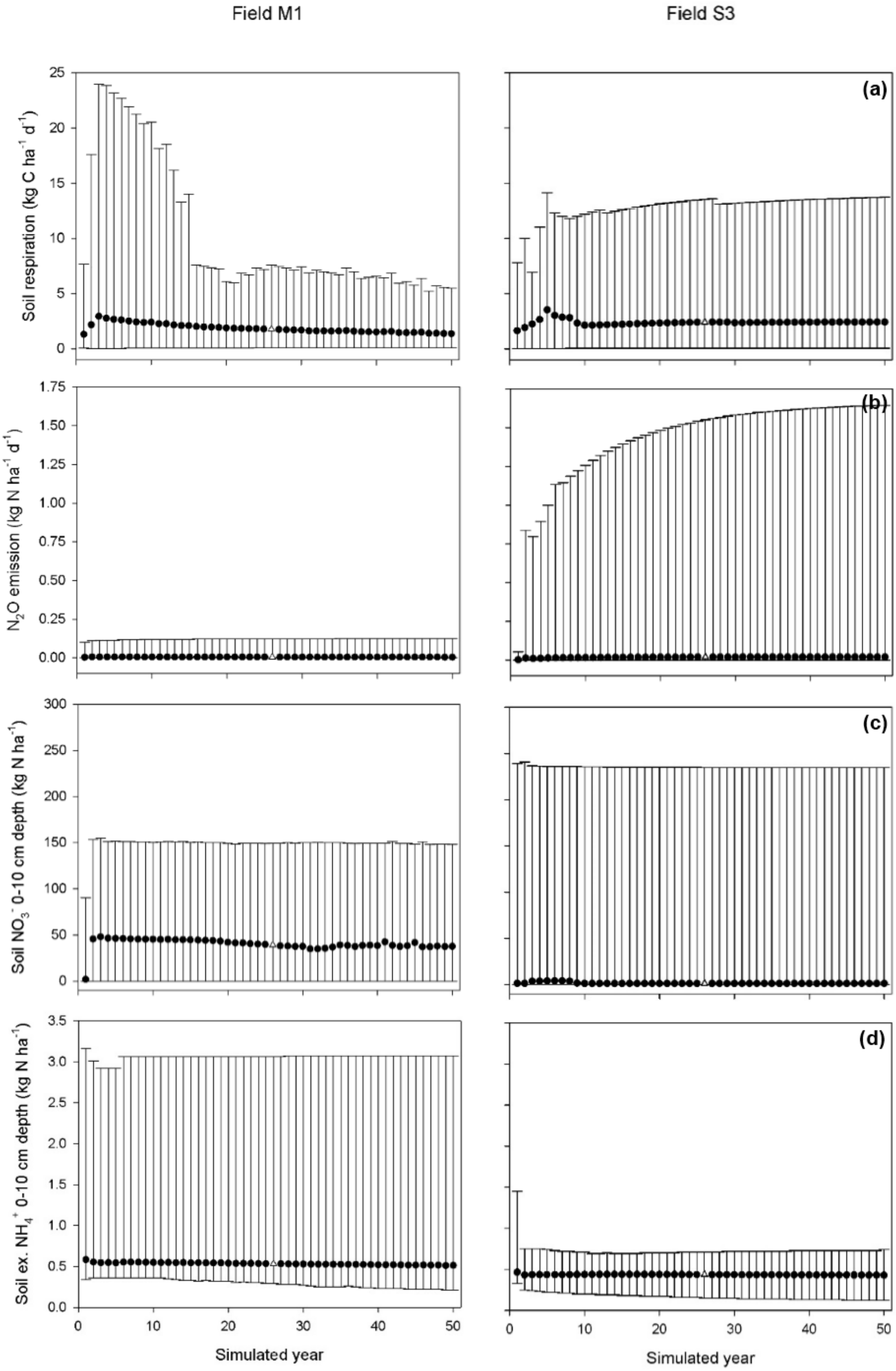

Figure C.1. DNDC-modelled mean daily values and annual range, using ≤ 49 years of pre-simulation model runs, for (a) soil respiration, (b) N_2O emission, (c) soil NO_3^-, and (d) soil exchangeable NH_4^+, for fields M1 (c. 35% SOM content) and S3 (c. 70% SOM content). Year 26 mean value is indicated with a white triangle. Error bars indicate annual minimum and maximum daily values.

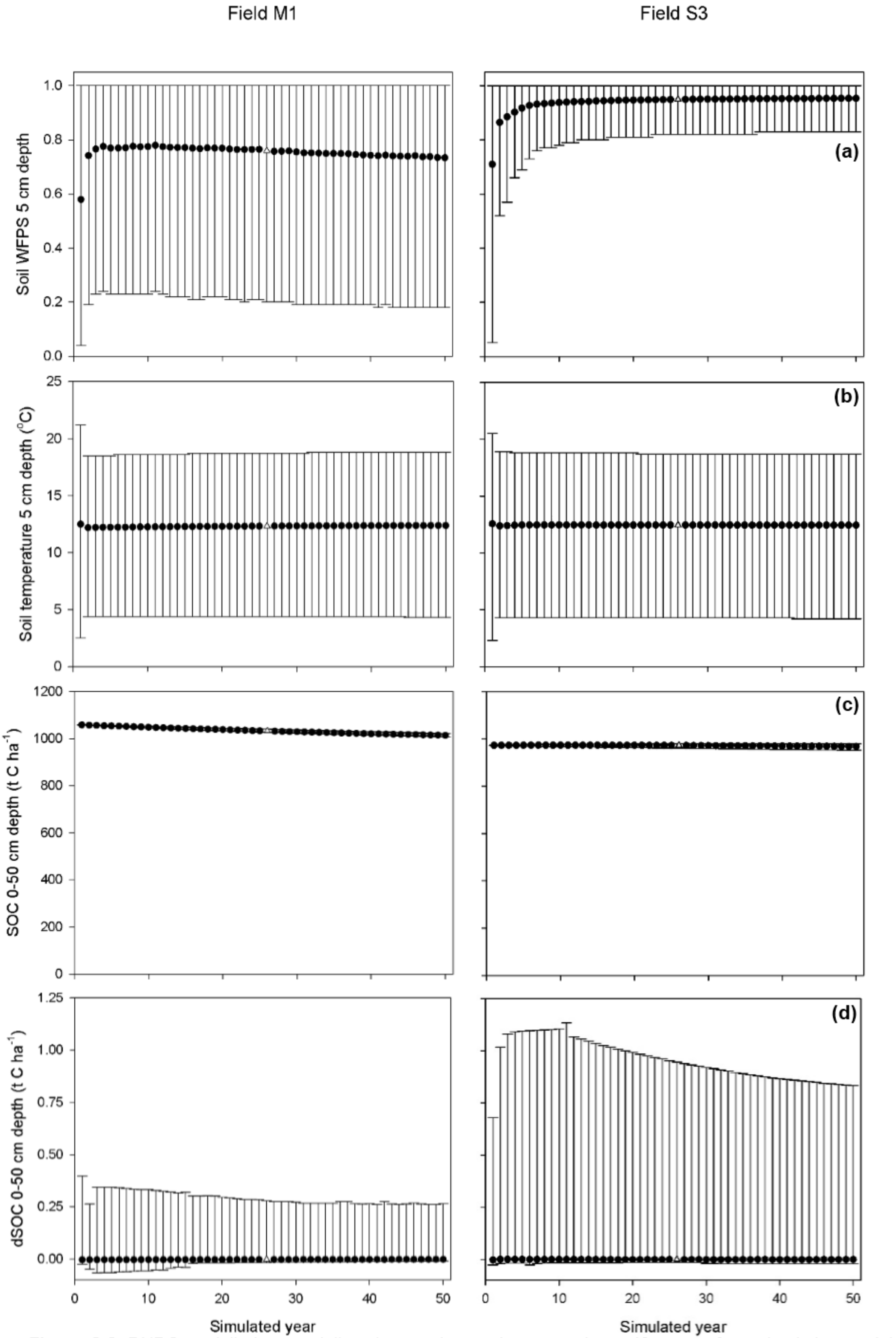

Figure C.2. DNDC-modelled mean daily values and annual range, using ≤ 49 years of pre-simulation model runs, for (a) soil WFPS fraction, (b) soil temperature, (c) SOC, and (d) *d*SOC, for fields M1 (c. 35% SOM content) and S3 (c. 70% SOM content). Year 26 mean value is indicated with a white triangle. Error bars indicate annual minimum and maximum daily values.

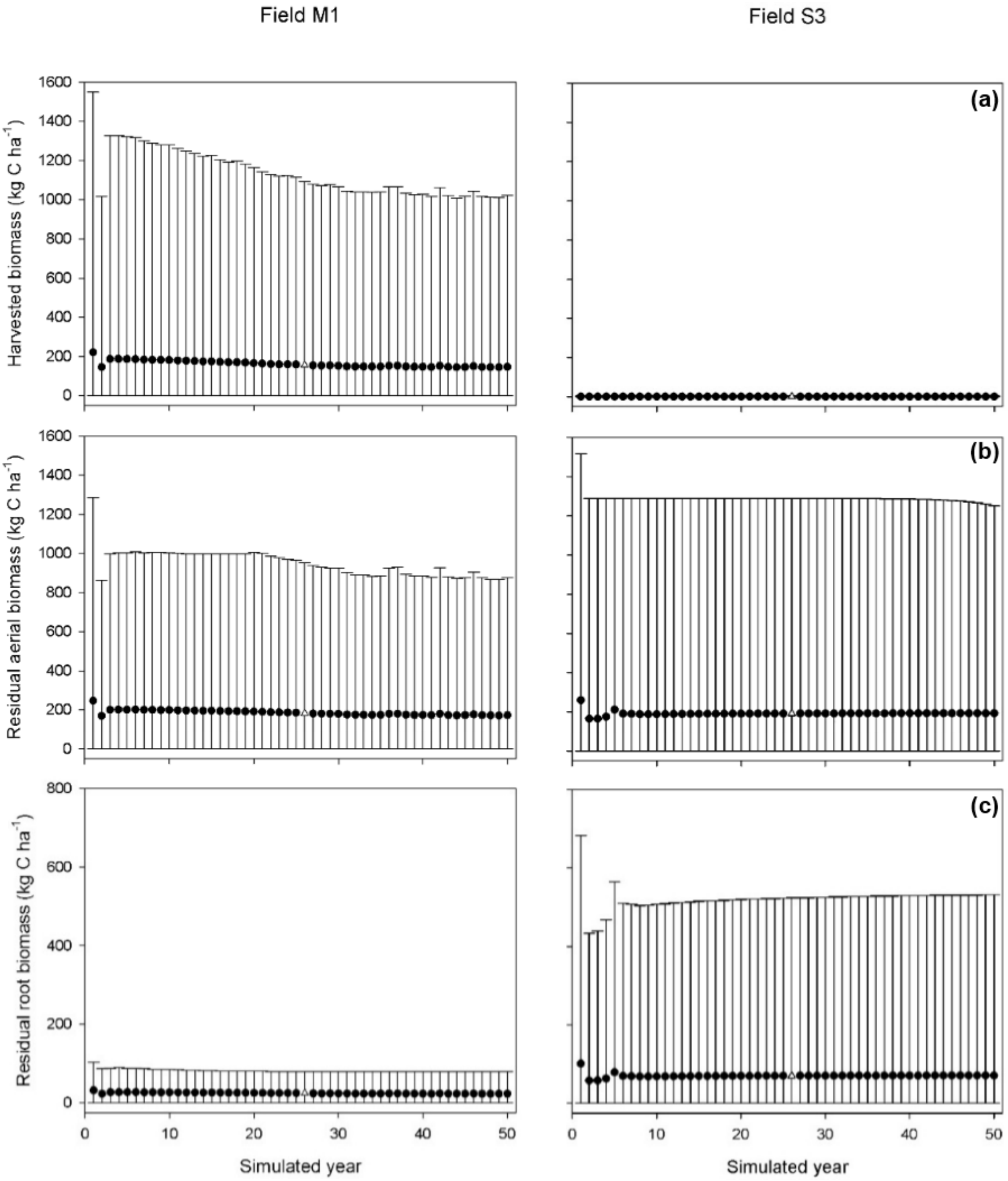

Figure C.3. DNDC-modelled mean daily values and annual range, using ≤ 49 years of pre-simulation model runs, for (a) harvested crop biomass C (‘grain’), (b) residual crop aerial biomass C (‘leaf’ + ‘stem’), and (c) residual crop root biomass C (‘root’), for fields M1 (c. 35% SOM content) and S3 (c. 70% SOM content). Year 26 mean value is indicated with a white triangle. Error bars indicate annual minimum and maximum daily values.

Appendix D

Additional results for validation of DNDC for fields M2, M3, S1 and S2

(Chapter 3)

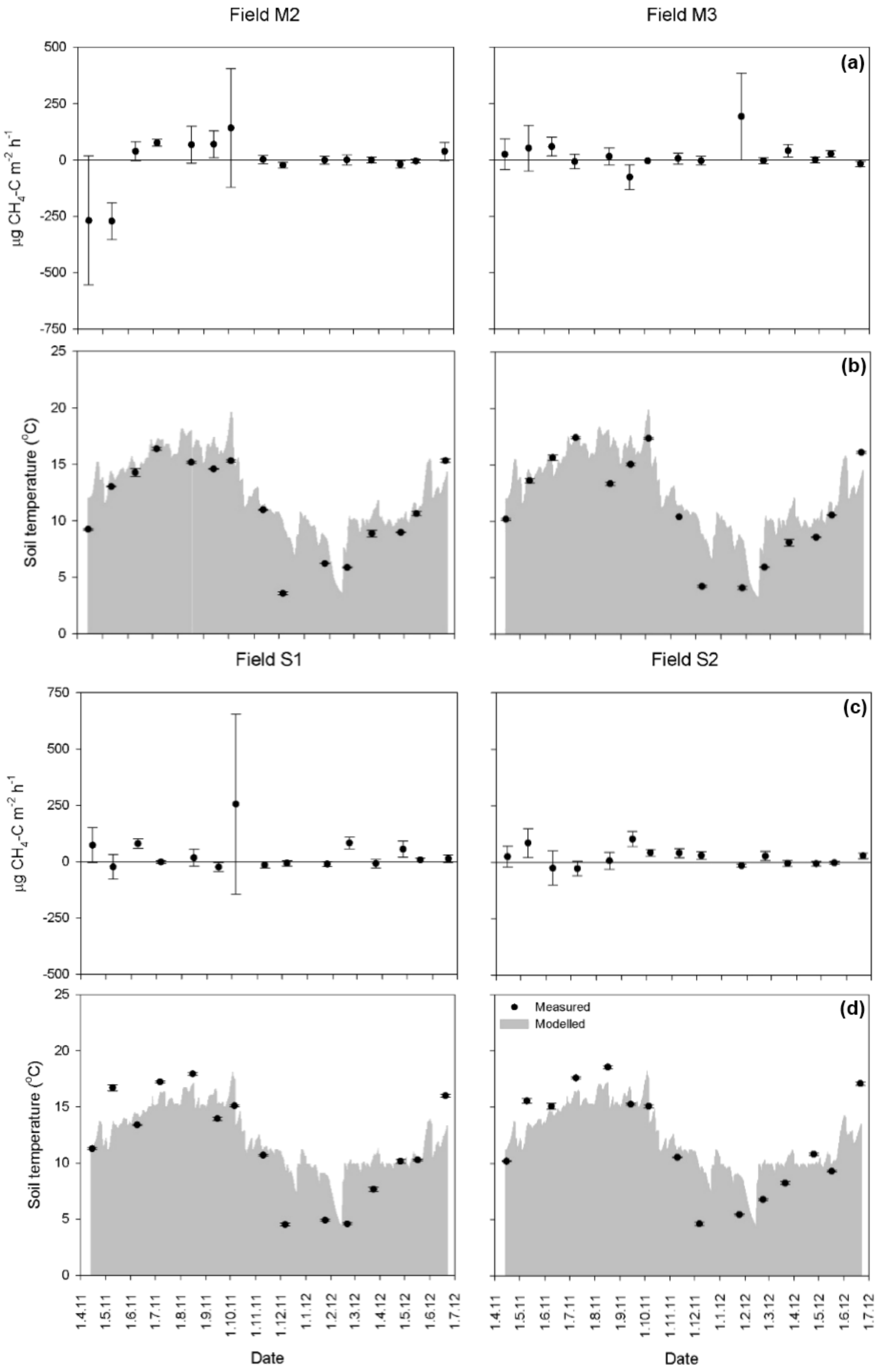

Figure. D.1. Performance of DNDC at predicting episodic CH_4 emission (a) and (c); and soil temperature (b) and (d); 14th April 2011 to 22nd June 2012: cropped soils – on fields M2 and M3 (c. 35% SOM content) and S1 and S2 (c. 70% SOM content). Error bars represent ± 1 S.E. of the mean value.

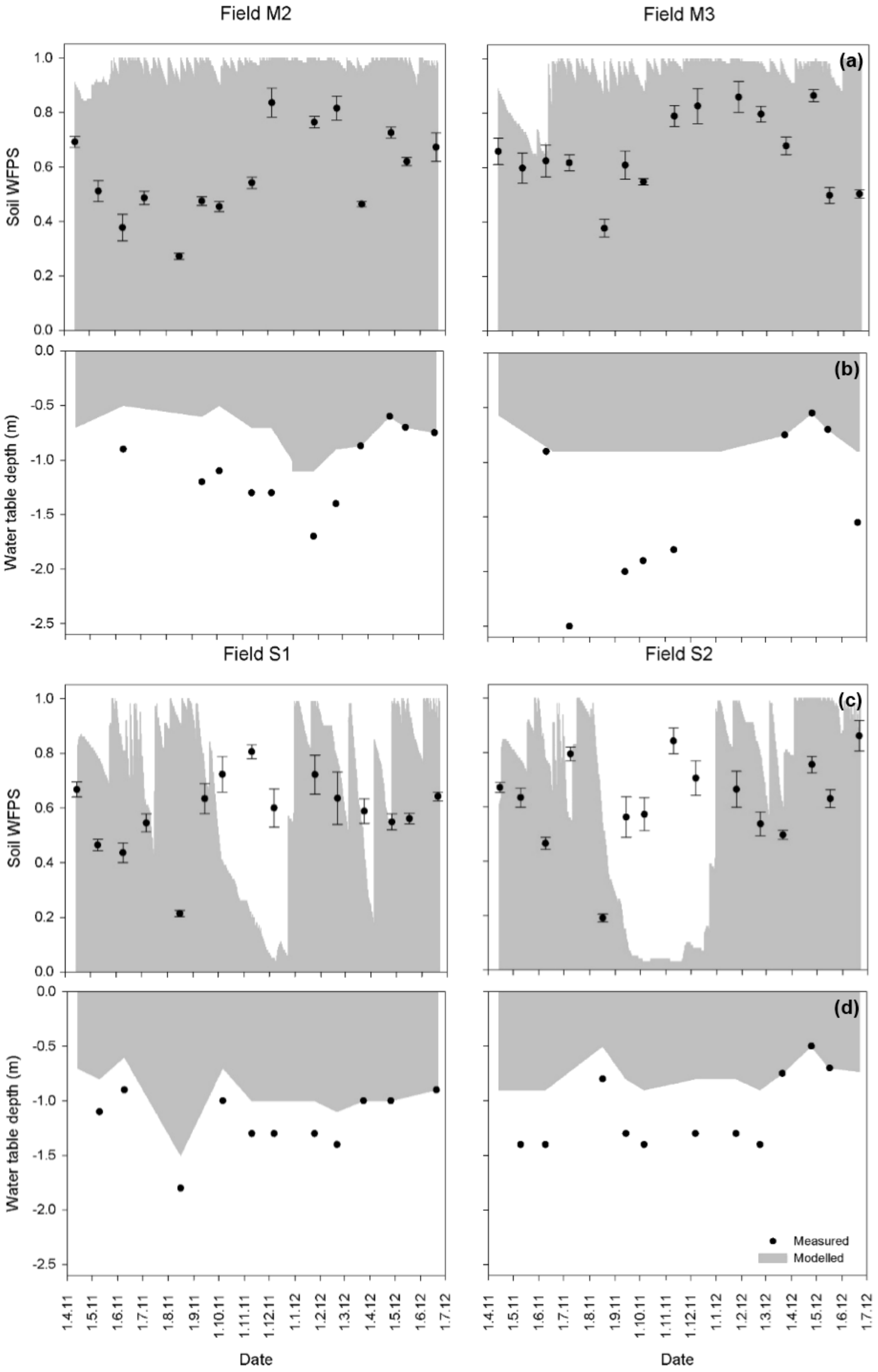

Figure. D.2. Performance of DNDC at predicting episodic soil WFPS (a) and (c); and water table depth (b) and (d); 14th April 2011 to 22nd June 2012: cropped soils – on fields M2 and M3 (c. 35% SOM content) and S1 and S2 (c. 70% SOM content). Error bars represent ± 1 S.E. of the mean value.

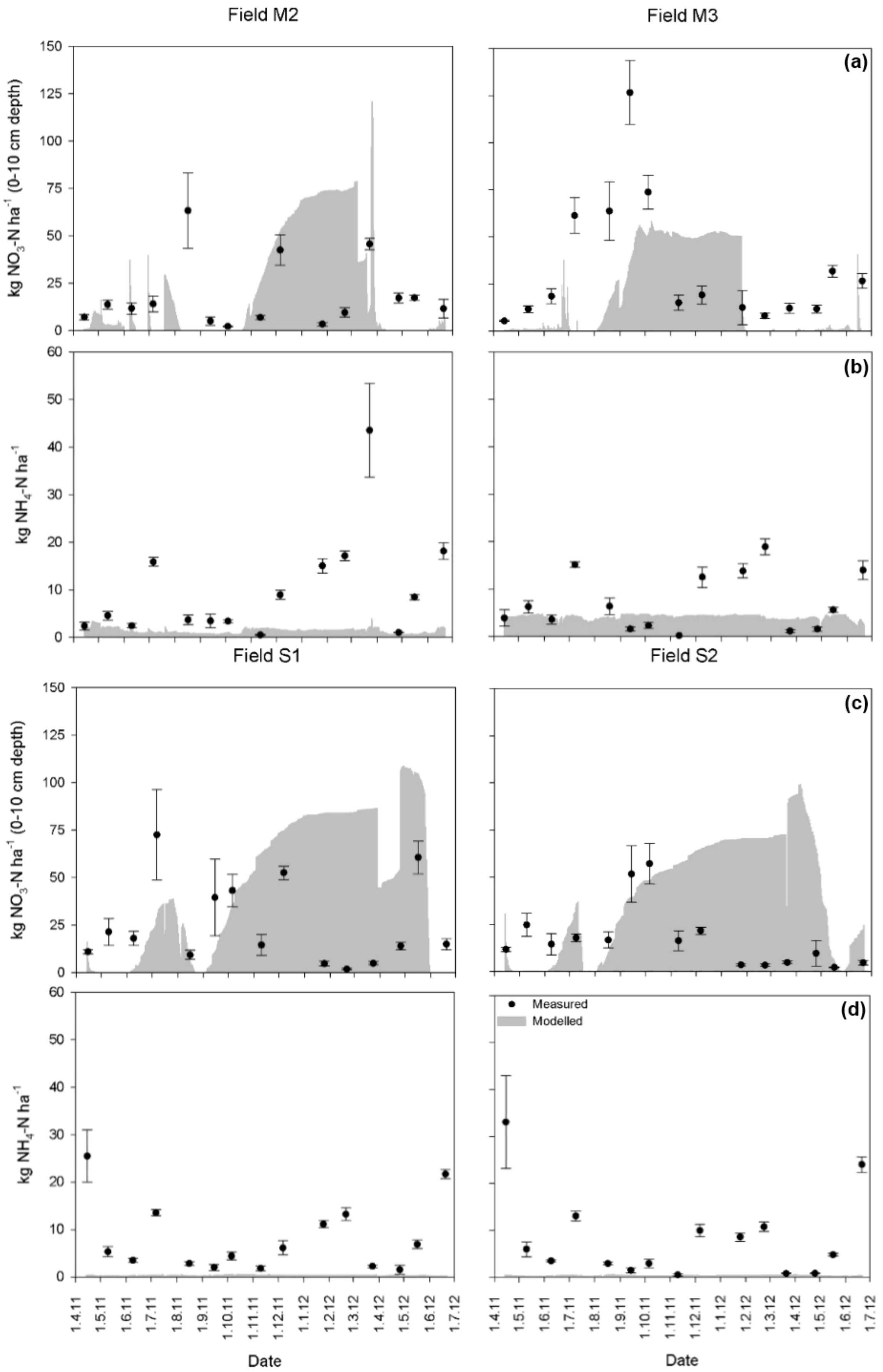

Figure. D.3. Performance of DNDC at predicting episodic soil nitrate (a) and (c); and soil exchangeable ammonium (b) and (d); 14th April 2011 to 22nd June 2012: cropped soils – on fields M2 and M3 (c. 35% SOM content) and S1 and S2 (c. 70% SOM content). Error bars represent ± 1 S.E. of the mean value.

Appendix E

Details of soil mesocosm assemblage

(Chapter 5)

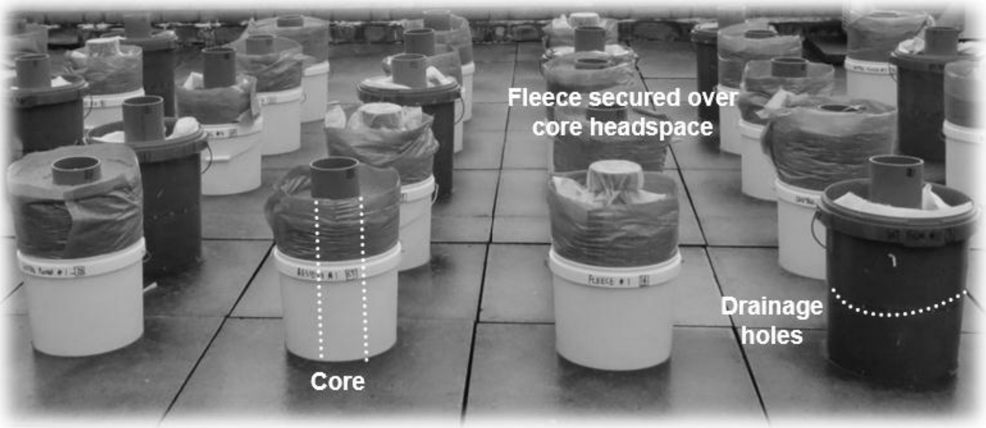

Figure E.1. Examples of intact soil cores used in the mesocosm mitigation experiments. Front row, from left to right: ploughed (C); surface residue application (R); fleece application (F); and water table maintained at 15 cm below the soil surface (WL) treatments. All cores are installed within a sand-filled bucket for stability, soil temperature regulation, and to allow for water table depth control in WL and WH treatments.

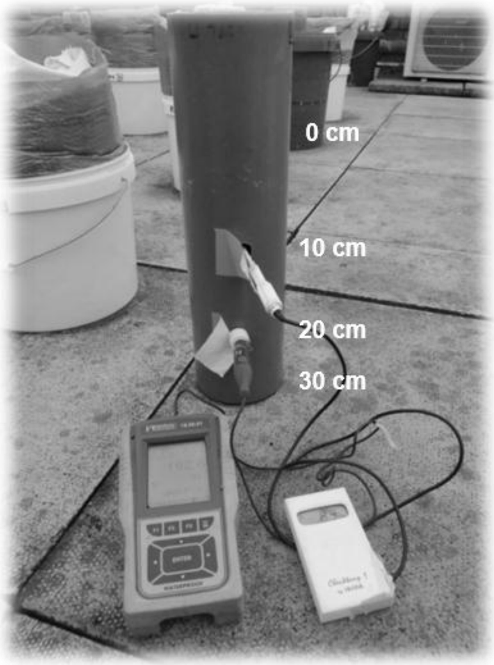

Figure E.2. Example of an intact soil core used in Experiment phase (II). Sampling holes allow access for soil redox (Eh) and soil temperature measurements.

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