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Mitigating the impacts of intensive agriculture on lowland organic soils

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Mitigating the impacts of intensive agriculture on lowland organic soils

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Declaration and Consent

‘I hereby declare that this thesis is the result of my own investigations, except where otherwise stated. All other sources are acknowledged by bibliographic references. This work has not previously been accepted in substance for any degree and is not being concurrently submitted in candidature for any degree unless, as agreed by the University, for approved dual awards.

I confirm that I am submitting this work with the agreement of my supervisor(s)’.

‘Yr wyf drwy hyn yn datgan mai canlyniad fy ymchwil fy hun yw’r thesis hwn, ac eithrio lle nodir yn wahanol. Caiff ffynonellau eraill eu cydnabod gan droednodiadau yn rhoi cyfeiriadau eglur. Nid yw sylwedd y gwaith hwn wedi cael ei dderbyn o’r blaen ar gyfer unrhyw radd, ac nid yw’n cael ei gyflwyno ar yr un pryd mewn ymgeisiaeth am unrhyw radd oni bai ei fod, fel y cytunwyd gan y Brifysgol, am gymwysterau deuol cymeradwy.

Rwy'n cadarnhau fy mod in cyflwyno'r gwaith hwn gyda chytundeb fy Ngoruchwyliwr (Goruchwylwyr)

Signed: ***Samuel Musarika*** Date: 10th of October 2022

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Summary

As the global climate faces immense pressure because of anthropogenic activity, questions have arisen regarding how to minimise the impact of anthropogenic forcing on the climate system. Global greenhouse gases (GHG) are created by a variety of sources, e.g., transport, industry, and agriculture. While we can innovate or make improvements to different industries and the transport sector to reduce the amount of GHG they produce e.g., using electric vehicles, reducing GHG emissions from agriculture is challenging as it risks creating food security issues, especially with some of the mitigation measures that would require land to be taken out of agricultural production. The need to find reliable ways to reduce GHGs without damaging food security is made more pressing by global population growth which means that more food needs to be produced with finite resources. Land required for arable agriculture will increase with population growth and increased food demand. Countries in the northern hemisphere, such as the UK, produce substantial amounts of food from cultivated peatlands. Consequently, these peatlands have lost their carbon storage capacity and if current practices persist, they will be lost entirely in under one hundred years. To preserve cultivated peatlands or at least slow down their degradation, this thesis examines a range of potential mitigation measures which could address the loss of cultivated peatlands. First, the use of fresh organic matter was investigated, finding that this shows potential, especially, if applied in conjunction with an elevated water table. Secondly, it observed the long-term effects of fresh organic matter to assess if there any negative effects occur in the long term. Further, this study looked at the effects of iron chloride, iron sulphate and calcium sulphate on GHG fluxes in cultivated peatlands. It shows that these can potentially be beneficial to peatland preservation, but that they have limitations and potential drawbacks e.g., there is a risk of making the soil acidic, and build-up of sulphides in the soil which can get re-oxidised when the soil is drained can lead to the release of sulphuric acid. What the results in this thesis confirm is that there is no single solution to the issue of preserving peatlands whilst producing enough food to satisfy the projected population. There are some benefits to the mitigation measures, but they do have associated drawbacks which bring their feasibility into question. Stakeholders will need to explore this mitigation measure further, especially on a large scale as the results might not mirror what the controlled mesocosm showed.

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Acronyms

C	carbon
$CaSO_4$	calcium sulphate
CFC	chlorofluorocarbons
CH_4	methane
Cl	chlorine
ClO	chlorine monoxide
$ClONO_2$	chlorine nitrate
cm	centimetre
CO_2	carbon dioxide
DEFRA	Department for Environment, Food and Rural Affairs
EEA	European Environment Agency
ER	ecosystem respiration
EU	European Union
FAO	Food and Agriculture Organization
$FeCl_2$	iron (II) chloride or ferrous chloride
$FeSO_4$	iron (II) sulphate or ferrous sulphate
FOM	fresh organic matter
GHG	greenhouse gas
GWP	global warming potential
IPCC	Intergovernmental Panel on Climate Change
IUCN	International Union for Conservation of Nature
km	kilometre
km^2	square kilometre
KP	Kyoto Protocol
m	metre
mg	milligram
Mha	million hectares
mm	millimetre
MP	Montreal Protocol
N	nitrogen
N_2O	nitrous oxide
NOx	nitrogen oxides

NFU	National Farmers Union
NH_3	ammonia
NPOC	non-purgeable organic carbon
NPPF	National Planning Policy Framework
ODP	ozone depletion potential
ODS	ozone-depleting substances
OM	organic matter
PE	priming effect
PES	Payment for Ecosystem Services
Pg	petagrams
SOC	soil organic carbon
SOM	soil organic matter
TC	total carbon
UNFCC	United Nations Framework Convention on Climate Change

Chapter 1. Introduction

1.1 Background

Anthropogenic activities, such as industry, transport, and agriculture, produce significant amounts of greenhouse gas (GHG) emissions that are released into the atmosphere. The atmospheric concentration of GHGs including carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O) have increased and are now the main driving force of climate change (IPCC, 2013). The levels of atmospheric CO_2 concentrations have increased from a pre-industrial concentration of ~260 ppm to the current levels of over 400 ppm (Wigley, 1983; IPCC, 2014). Atmospheric CH_4 concentrations are 150% more than before the year 1750 (IPCC, 2013). Anthropogenic pressure on the environment will continue to increase as the world's population grows and increasing pressure occurs on the finite resources the planet has to offer. The world's population could exceed 10 billion in this century (Godfray et al., 2010; Hickey et al., 2019). Food demand and production are projected to inevitably increase with population growth. In Europe, agriculture is the second-largest contributor to emissions (EEA, 2016; EEA, 2021). As agriculture is a major contributor to GHG, there are concerted efforts to mitigate the amount of GHG emissions from livestock and arable farming (Foresight, 2011; Gilbert, 2012; Taft et al., 2017).

Arable agriculture on both mineral soils and organic soils has a substantial impact on the climate as soils are important stores of carbon (C). However, organic soils (specifically cultivated peatlands) are faced with the greatest challenge. Peatlands worldwide cover an estimated area of 400 Mha, which is 3% of the global land surface but play a vital role in C sequestration (Frolking et al., 2011; Joosten & Couwenberg, 2009). Since the last glaciation, peatlands have sequestered a significant amount of carbon, estimated to range from 400 to 500 Pg (Belyea & Clymo, 2001). 30% of the global soil carbon storage is present as soil organic matter (SOM), which is a crucial C reservoir (Fontaine et al., 2003; Berglund & Berglund, 2010; Frolking et al., 2011). In the UK, peatlands cover 10% of the total land area but they store a considerable proportion of C. Dawson & Smith (2007) estimated the total amount of C in UK peatlands (upland blanket, lowland raised bogs, and fen peats) was 9.8 billion tonnes, the majority of this (about 6.9 billion tonnes) being in Scotland (Cruickshank et al., 1998; Dawson & Smith, 2007). Peatlands are able store C due to the presence of water which creates anoxic conditions, therefore, leading to reduced organic matter (OM) decomposition. However, due to drainage for agriculture, cultivated peatlands no longer represent sinks of C. Drainage has led to increased oxidation of the organic soils which in turn make them large emitters of

both CO_2 and N_2O (Foresight, 2011; Gilbert, 2012; Parish et al., 2008). In the period between 1978 and 2003, UK soil C loss occurred at an average rate of 0.6% yr^{-1}. More alarmingly, a higher rate of 2% yr^{-1} was recorded on soils with C levels of more than 100 g kg^{-1} e.g., parts of the East Anglian Fens (Bellamy et al., 2005; Holman, 2009). Evans et al. (2017) reported that arable peatlands produce an estimated 7,600 kt CO_2e yr^{-1}, which is 32% of total UK peatlands.

In general, soil is not only a crucial component of the food system but also important for water regulation, and carbon and nitrogen cycles. Whilst it is hard to put a monetary value on peat soil, the UK government estimates that the degradation of UK soils in general, costs over £1 billion a year (Parliamentary Office of Science and Technology, 2015). Unfortunately, there are no figures for peat degradation alone, but in 2019 the National Farmers Union (NFU) reported that the total agricultural output from organic soils in The Fens in East Anglia alone is worth £1.23 billion.

Peatlands are irreplaceable in our lifetime because their formation takes many years due to low productivity rates (Moore, 1987). The rate of peat formation varies in different ecosystems, and the rates are linked to the prevailing hydrological conditions (Moore, 1987). The hydrological conditions are subsequently affected by climatic factors such as precipitation, evaporation, and temperature. Furthermore, factors including topography, land use, and vegetation cover also affect peat formation rates (Belyea & Clymo, 2001; Moore, 1987). Measurements made on the net primary productivity of distinct types of wetlands from various parts of the world show a wide variation. Wetlands comprising of cattail (*Typha spp.* L.) are amongst the most productive ecosystems, producing biomass in general at rates of 5-6 kg m^{-2} yr^{-1} and in extreme cases rates of 15 kg m^{-2} yr^{-1} have been recorded. Wetland forests in Florida achieve 2-3 kg m^{-2} yr^{-1} but the peatlands in the temperate and boreal regions rarely exceed 1 kg m^{-2} yr^{-1} (Moore, 1987).

A range of studies have proposed and explored mitigation measures to protect cultivated peatlands such as i) longer crop rotation (Locke & Bryson, 1997; Triplett Jr & Dick, 2008) ii) no or reduced tillage (Locke & Bryson, 1997; Triplett Jr & Dick, 2008) iii) use of winter cover crops (or green manures) such as wheat (*Triticum aestivum* L.), clover (*Trifolium spp.* L.), and vetch (*Vicia spp.* L.) (or other native legumes) (Wen et al., 2019) and iv) rising water tables (Parliamentary Office of Science and Technology, 2015; Renger et al., 2002; Matysek et al., 2019). In the UK, Germany, and the Netherlands, water table management is used to reduce peat loss and lower GHG emissions. However, the water table levels used are sometimes not

enough. UK farmers maintain low water table levels due to concerns about the impact higher water tables could have on productivity, thus effectively over-draining the peatland (Musarika et al., 2017). Water tables closer to the surface could impact farm machinery (Musarika et al., 2017). However, of most concern is the potential of the prevalence of plant fungal diseases, such as water moulds (*Oomycetes*) e.g., *Aphanomyces*, *Pythium*, and *Phytophthora*, which could considerably impact crop yield (Katan, 2000). From 1845 *Phytophthora* caused potato blight leading to the Great Famine in Ireland which resulted in starvation, death, and mass migration (Geber & Murphy, 2012). Therefore, the threat of plant disease should not be taken lightly.

A no-tillage or reduced tillage approach incurs higher equipment costs. For example, using expensive seed drills and strip-tillers (Parliamentary Office of Science and Technology, 2015), and the use of cover crop (green manure) i.e., fresh organic matter (FOM) applied to organic soil risks priming the release of GHG, therefore defeating their purpose as mitigation against soil C loss. In addition, crops, such as root vegetables including carrots (*Daucus carota* L.), can lead to increased N_2O release from the soil (Weslien et al., 2012).

There is a need to increase food production while simultaneously protecting the soil, therefore there is an increased drive towards sustainable intensification to meet the challenges of food production and preservation of soils across the world. Aspects of soil preservation consider how to improve soil health thus building resilience in farming, with particular attention to both crop yield and environmental impact. The aims of sustainable intensification are ideally; increasing yields whilst also reducing impacts on the environment, although it may be more realistic to either i) maintain current yields whilst reducing impacts on the environment, or to ii) increase yields whilst maintaining the current impacts on the environment. Decreasing GHG emissions from arable agriculture while maintaining current yields would be ideal and could contribute significantly to the UK's GHG emission reduction targets. However, each aspect of any proposed action needs to be thoroughly investigated and assessed, and needs to be scalable nationally and if possible, globally.

1.2 Aim of thesis

The principal aim of this thesis was to find mitigation measures that could be effective at reducing GHG emissions from cultivated peatlands in the UK, especially measures which do not make the soil unsuitable for horticultural production. It is based on an analysis of four

studies designed to investigate the impact of different mitigation methods on reducing GHG emissions from cultivated peatlands.

The specific research aims were to:

1. Assess the effects of additions of FOM on the release of GHGs such as CO_2, CH_4, and N_2O in currently cultivated peat soils in the UK.
2. Evaluate the response of GHG emissions when FOM is added into the peat soil and assess this response in conjunction with the water table.
3. Assess how CO_2 from cultivated peatlands will respond to added FOM of a commercially important crop in the long term after the initial application and a season of being left fallow and the legacy of an elevated water table.
4. Assess the effectiveness of calcium sulphate ($CaSO_4$), iron (II) chloride ($FeCl_2$), and iron (II) sulphate ($FeSO_4$) in suppressing CO_2 and CH_4 and the added effect of water table depth.

1.3 Structure of thesis

The individual chapters of this thesis focus on addressing the research aims.

Chapter 1 is the general introduction to the thesis and set outs the aims of the thesis and how it is structured.

Chapter 2 - is a review of the published literature. It begins by providing a broad overview of climate change and the effects of anthropogenic activity. The literature review then narrows down to soils and their importance to the climate system as they are significant sinks of C. Emphasis is then placed on arable agriculture on cultivated peatlands. The use of cultivated peatlands is problematic as the stored carbon is being emitted into the atmosphere, driving climate change. However, cultivated peatlands are crucial for food production. This chapter outlines the options that are available to mitigate the release of C from the cultivated peatlands.

Chapter 3 – this chapter aims to address the first objective. It is based on analysis of a mesocosm experiment with intact cores that were collected from a cultivated peatland in The Fens. The setup of this study included two water table treatments of 20 cm and 50 cm from the ground level. Half of the cores in each treatment had FOM added in the form of barley (*Hordeum vulgare* L.) straw. Throughout the experiment, fluxes of CH_4, CO_2, and N_2O were recorded using LI-COR automated chambers (Model 8100-104), LI-COR Infra-Red Gas

Analyser (IRGA) LI8100A, and a Picarro Greenhouse Gas Concentration Analyser (G2508). The experiment was conducted for 27 weeks between November 2018 and May 2019.

Chapter 4 – the scope of this chapter was limited due to the COVID-19 pandemic which meant that it was not possible to change the parameters of the experiment as the facility was locked down. However, this study managed to assess and observe how CO_2 from a cultivated peatland will respond to added FOM of a commercially important crop in the long term after the initial application and a season of being left fallow. The long-term effect of FOM was studied in conjunction with the additional impact of an elevated water table depth.

Chapter 5 – this chapter provides evidence on the effectiveness of $CaSO_4$, $FeCl_2$ and $FeSO_4$ in suppressing CO_2 and CH_4 from cultivated peat soils and their potential as amendments for GHG mitigation. In addition to the amendments, it explored contrasting water table depths, reflecting different drainage and rewetting conditions. Two experiments were conducted as part of this study. After the first experiment, it was clear that there was no CH_4 production at the set water table. A follow-up experiment used a 0 cm water table depth (flooded). Both experiments provided insight into the dynamics of cultivated peatlands when the water table is altered. The experiment also provided insight into the importance of substrate for CH_4 production in flooded peat soils.

Chapter 6 – provides a general discussion for the thesis, summarising all the findings from the thesis and makes suggestion of what further research could be conducted to answer some of the questions that were raised by this thesis.

Chapter 2. Reduction of greenhouse gas emissions from cultivated peatlands whilst maintaining food production

Abstract

Agriculture is one of the biggest producers of greenhouse gas (GHG) emissions globally - responsible for a third of GHG emissions. In Europe, agriculture is the second largest GHG source after transport representing an estimated 11% of total emissions. The amount of GHG emissions from agriculture are projected to rise even further as the global population reaches a new record of 10 million in the next 30 years. Arable agriculture on drained peatlands is damaging the soil and making them large sources of CO_2. If this risk is not mitigated, productive peatlands could be lost completely by 2050. This chapter reviews the history of cultivated peat soils and their potential to sequester atmospheric C. It also investigates other important GHGs such as N_2O and CH_4 and the consequences of mitigation measures on these GHGs. Food production is commercially important; therefore, this review responds to this by identifying ways that could both be beneficial for food production and GHG reduction. Such research is important as, in the UK, studies exploring GHG mitigation measures and food production, as well as crop studies, are currently limited.

2.1 Introduction

The earth's climate system is under increasing pressure from anthropogenic activities, e.g., transport, industry, and food production, which are releasing enormous quantities of greenhouse gas (GHG) such as carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O) into the atmosphere (IPCC, 2013). Further, anthropogenic pressure on the environment will continue to increase as the world's population grows. Estimates indicate that the population could exceed 9 billion within this century (Godfray et al., 2010). Hickey, et al. (2019) estimated that the population could be over 10 billion people in the next 30 years. The level of atmospheric CO_2 concentrations has increased significantly from a pre-industrial concentration of about 260 ppm to the current levels of over 400 ppm (Wigley, 1983; IPCC, 2014). Simultaneously, CH_4 concentrations have increased by over 150 times (IPCC, 2013). This increase will continue as the human population grows.

Agriculture is a major source of anthropogenic GHG and there are concerted efforts to reduce its impact, i.e., by reducing the sources of GHG from livestock and arable farming (Foresight, 2011; Gilbert, 2012; Taft et al., 2017). In Europe, agriculture is the second-largest sector contributor to emissions with 11% of GHG emissions; up from 9.9% in 2016 (European Environment Agency, 2016; 2021). Arable agriculture on both mineral soils and organic soils has a significant impact on the climate. However, organic soils (specifically cultivated peatlands) face the greatest challenge. Over 20% of peatlands around the world are exploited for arable agriculture (Strack, 2008).

Peatlands worldwide cover an estimated area of 400 Mha, which is 3% of the global land surface but plays a significant role in C sequestration (Frolking et al., 2011; Joosten & Couwenberg, 2009). Due to the presence of water and the absence of oxygen, peatlands are natural accumulators of carbon (C) derived from organic plant material. Peatlands represent about 30% of the global soil C storage (Berglund & Berglund, 2010; Frolking et al., 2011). Due to drainage, cultivated peatlands may no longer represent sinks of C but have become significant emitters of both CO_2 and N_2O (Foresight, 2011; Gilbert, 2012; Parish et al., 2008).

In the UK, cultivated peatlands produce substantial amounts of CO_2, estimated to be about six times that of restored peatlands (Natural England, 2010; Parish et al., 2008). Evans et al. (2016) reported that in East Anglia (a region in the east of England), intact fen was a sink of C at -5.13 t CO_2 ha^{-1} yr^{-1} whereas cultivated fen was a source of about 25 to 28 t CO_2 ha^{-1} yr^{-1}. Peatlands

in the UK cover 10% of the total land area but they store a considerable proportion of C. Dawson & Smith (2007) estimated that the total amount of C in UK peatlands (upland blanket, lowland raised bogs, and fen peats) was 9.8 billion tonnes, with the majority of this (about 6.9 billion tonnes) being in Scotland (Cruickshank et al., 1998; Dawson & Smith, 2007). Peat soils form when partly decomposed plant matter accumulates under waterlogged conditions, which limit the normal decomposition process. This is achieved by numerous pathways such as lack of oxygen, nutrient deficiency, low average temperature, and high acidity. However, the key factors that influence the development of peat are climate, topography, and geology (Smith et al., 2007). Peatlands occur in two distinct types, fens, and bogs depending on where they are formed (Dawson & Smith, 2007). Fens have a high pH as they get most of their water from mineral-rich groundwater (minerotrophic) whiles bogs have low pH and are acidic because they get most of the water from rain (ombrotrophic) (Griffiths et al., 2019; Pedrotti et al., 2014). Naturally, peatlands are covered in dense moss. In the case of bogs, the moss tends to be peatmoss (*Sphagnum spp.*) while in fens it is usually mosses in the Amblystegiaceae family (Emsens, et al., 2020; Vitt, 2008). Fresh organic matter is constantly replenished by the mosses, and it provides a suitable environment for many microbes e.g., nitrogen-fixing and methanotrophic bacteria (Emsens, et al., 2020). As bogs are acidic, they are not as suitable for agricultural crop production as fens so not all peatlands are exploited for agriculture, therefore, there is greater pressure on fens.

In the period between 1978 and 2003, UK soil C loss occurred at an average rate of 0.6% yr^{-1}. More alarmingly, a higher rate of 2% yr^{-1} was recorded on soils with C levels of more than 100 g kg^{-1} e.g., parts of the East Anglian Fens (Bellamy et al., 2005; Holman, 2009). From the mid-19th century, ~84% of peat in East Anglia has been lost. In some areas, peatlands that were over four metres deep have disappeared completely (Holman, 2009). Consequently, there is growing concern that over two-thirds of productive peatlands might be lost by 2050 (Dawson et al., 2010). Loss of productive peatlands will negatively affect agricultural productivity and thus threaten food security. It is important to note that soil C is not only lost to the atmosphere, but also significant amounts are leached deeper into the soil layers and lost to freshwaters and groundwater.

2.2 Material and methods

The initial step in this literature review was to read a collection of literature that was already acquired from previous studies such as (Musarika et al., 2017). From this set of literature,

gaps were identified and areas where further understanding was needed. Literature searches were then carried out using a variety of tools but mainly Web of Knowledge, Google Scholar, university library search tools and search engines such as Bing and Google. For general peatlands, terms such as peat, peatland, fens, and bogs were used. Other terms there were searched for were fresh organic matter, priming effect, and FOM. The terms were either searched as a single word or a combination with other terms such as peatland CO2 and FOM priming effect.

The literature search was focused on peatlands in the mid-latitudes as these were the peatlands of concern for this thesis. However, relevant literature from other parts of the world was sought. The literature search brought many results but not all of them were relevant.

2.3 Implications of drying and wetting of peatlands

Peatlands contain a diverse microbial population that has varied metabolic diversity (Andersen et al., 2013). The community of organisms that can be found in natural peatlands consist of fungi, bacteria, and archaea (Andersen et al., 2013; Dedysh et al., 2004). These organisms have developed metabolic and physiological adaptations to cope with the limiting conditions found in peatlands such as low availability of oxygen, low temperature, acidity, and lack/reduced availability of nutrients (Peltoniemi et al., 2012). These diverse microorganisms in peatlands are responsible for the turnover of organic carbon and are responsible for the mineralisation and uptake of nutrients (Andersen et al., 2013). Fungi are the main aerobic decomposer in peatlands as well as actinobacteria which breaks down cellulose in a comparable way to fungi (Peltoniemi et al., 2012).

A water table close to the surface limits the amount of available oxygen which reduces the rate of decomposition. However, when the peatlands are drained, more oxygen becomes available for the aerobic decomposition of organic matter to CO2 (Peltoniemi et al., 2012). In addition to the aerobic environment created when peatlands are drained, litter quality has a strong influence on both fungal and actinobacterial communities (Peltoniemi et al., 2012).

The drainage of peatlands does not only increase the rate of organic C decomposition, but it influences N mineralisation and its subsequent availability for plants (Prévost et al., 1999). Prévost et al. (1999) showed that draining a peatland can lead to increased temperature and the amount of mineral N available. This increase in mineralised N can lead to an increase in N2O emissions from peatlands into the atmosphere (Frolking et al., 2011).

2.4 Unsustainable use of peatlands

Globally, due to unsustainable farming practices, 12 million ha of farmland is abandoned every year (Pimentel et al., 1995). Soil protection is important as it can address challenges such as climate change, water security, and biodiversity. However, due to the growing population, questions about food security need to be addressed together with soil protection (Godfray et al., 2010). In the UK, the preservation and protection of peatlands and other stores of SOC have been identified as a vital component in the maintenance of the C balance due to their ability to sequester C (Parliamentary Office of Science and Technology, 2015; Ostle et al., 2009). Nevertheless, there is currently no specific legislation on soils only directives, such as the directives on water, nitrate, and biodiversity, which require a reduction in agricultural soil and fertiliser that run off into the water, have addressed soil management indirectly (Parliamentary Office of Science and Technology, 2015). The lack of legislation on soils is due to political issues when it comes to land management. In the European Union (EU), a proposed EU Soils Framework Directive was repeatedly blocked by member states notably, the UK together with Austria, Germany, France, and the Netherlands (Chen, 2019). Nevertheless, there are efforts by the EU Member States to create a framework that relates to soils. Even though the UK is no longer a member of the EU, such a framework could still have an impact on UK policy. The current policies on soils include the National Planning Policy Framework (NPPF) and DEFRA's Payment for Ecosystem Services (PES) (Parliamentary Office of Science and Technology, 2015). The NPPF is not legislation on agriculture but sets out the UK government's planning policies for England. The PES covers a wider range of natural resources than just soil (Smith et al., 2013). Between 1998 and 2007, policies such as the EU Common Agricultural Policy and UK agri-environment stewardship schemes have led to exceedingly small decreases in cropland which has led to modest decreases in the area of peatlands exploited for agriculture (Parliamentary Office of Science and Technology, 2015).

The UK has signed up to treaties, such as the Kyoto Protocol (KP) and the United Nations Framework Convention on Climate Change (UNFCCC), that aim to reduce the negative anthropogenic impact on the climate. The protection of peatlands will help to increase the UK's C sequestration goal, which will be crucial to maintaining the C balance (Smith, 2004; Nabuurs et al., 2000).

Studies on cultivated peatlands have focused on the emissions of CO_2, while CH_4 and N_2O have not been given as much prominence (Taft et al., 2017). Studies on CH_4 have shown no

discernible variation to CH_4 fluxes on cultivated peatlands (Musarika et al., 2017; Taft et al., 2017; Matysek et al., 2019). This lack of variation could explain the scarcity of studies of CH_4 fluxes on cultivated peatlands. CH_4 production in peatlands is dependent on communities of methanogens, a longer-term study could show different results as methanogens are bound to be abundant in higher water table environments. Whilst N_2O has lower fluxes than CO_2, it has a Global Warming Potential (GWP) 273 times more than that of CO_2 (IPCC, 2021) and it has been shown to have seasonal variations, lower in winter and higher in summer (Taft et al., 2017). N_2O is a damaging GHG and similarly needs to be addressed.

The UK is committed to reducing the impact of anthropogenic activity on cultivated peatlands (DEFRA, 2018). Consequently, several mitigation measures have come to the forefront, such as i) longer crop rotation, ii) use of no-tillage or reduced tillage methods, iii) growing winter cover crop (or green manures) e.g., wheat (*Triticum aestivum* L.), clover (*Trifolium spp.* L.) and vetch (*Vicia spp.* L.) (or other native legumes) instead of leaving it fallow, and iv) the raising of water tables to reduce peat oxidation (Locke & Bryson, 1997; Triplett Jr & Dick, 2008; Ostle et al., 2009; Parliamentary Office of Science and Technology, 2015; Musarika et al., 2017).

There is a drive towards sustainable intensification to meet food production challenges and to support the preservation of soil across the world (Cassman & Grassini, 2020; Franks, 2014; Godfray et al., 2010). Optimally, sustainable intensification aims to increase yield while also reducing environmental impact. However, it may be more realistic to either i) maintain the current yields while simultaneously reducing environmental impact, or ii) increase the yields while maintaining the current environmental impact. Achieving sustainable intensification is important given the projected growth in population. To be able to achieve sustainable intensification, there is a need to adopt farming practices that do not damage the soil, especially its ability to store C as this will build resilience in farming. Attention should be on both crop yield and environmental impact.

The reduction of GHG emissions from arable agriculture is a global challenge. In the UK context, decreasing GHG emissions from arable agriculture will contribute significantly to the UK's GHG emission reduction targets (Taft et al., 2017). However, there are knowledge gaps on how to achieve this, especially without destabilising food production. The proposed mitigation measures need thorough investigation and assessment, and they need to be scalable nationally, if possible, globally.

2.5 Preserving peatlands worldwide challenge

The preservation of peatlands on a global scale is challenging because the definition of what a peatland is varied from country to country and within countries. There are different terminologies used to refer to peatlands, sometimes interchangeably, such as histosols, peat soils, peatlands, and organic soils. The term 'organic soils' is best avoided because it can create confusion as it is sometimes used to define soils that do not meet the definition of true peat soil. Peatlands are soils that contain primarily organic matter (Berglund & Berglund, 2010); in the UK, peatlands are soils that contain at least 50-60% organic matter and have a thickness of more than 40-50 cm (Smith et al., 2007). These numbers vary between Scotland and Wales due to the development of different classification schemes. Regardless of the agreed threshold of organic matter, a soil classed as a peatland must contain have substantial depth of organic C.

2.6 Nitrogen (N) and nitrous oxide (N_2O) emissions from peatlands

In addition to releasing the C stored in the peatlands, agricultural activity on peatlands is producing substantial amounts of N_2O; between 65-80% of anthropogenic N_2O emissions are from agriculture (Maljanen et al., 2007). N_2O is a very potent GHG that has a GWP 273 times higher than that of CO_2 on a 100-year basis and is responsible for about 6% of observed anthropogenic radiative forcing (IPCC 2021). N_2O is produced naturally under suboxic conditions as a by-product of nitrification and denitrification (Kasimir-Klemedtsson et al., 1997). Organic soils (including peatlands) produce ten times more N_2O than mineral soils; the drainage of peatlands leads to the release of substantial amounts of N_2O as the decomposition of SOM is amplified by the presence of oxygen and nitrate production (Maljanen et al., 2007). Furthermore, the application of nitrogen-based fertiliser has led to an exponential increase in the production of N_2O (Signor et al., 2013). Additionally, nitrogen input has been shown to facilitate microbial decomposition of SOM, which leads to increased CO_2 emissions (Ostle et al., 2009).

2.6.1 Benefits of nitrogen (N)

The use of nitrogen can increase primary productivity in N-limited ecosystems leading to increased C sequestration into the soil (Evans et al., 2006). In forest biomass soils, atmospheric N deposition has been shown to increase C storage. However, its benefits are often negated as it can facilitate microbial decomposition, leading to the release of soil C. Furthermore, these benefits of enhanced growth are only relevant in natural and semi-natural ecosystems; due to

harvesting in arable agriculture. Once harvesting has taken place, only small amounts of crop biomass are left behind. The continuous addition of N can lead to N saturation which is then leached into surface water instead of being immobilised in the soil, therefore, defeating the C accumulation benefits (Evans et al., 2016).

2.6.2 The impact of nitrous oxide (N_2O) as an Ozone Depleting Substance (ODS)

In addition to being a potent GHG, N_2O has been identified as a potential Ozone Depleting Substance (ODS) (Ravishankara et al., 2009). The Montreal Protocol (MP) was set up to reduce the atmospheric concentration of ODS, specifically chlorine- and bromine-containing halocarbons, by restricting their emission and production. The MP has successfully led to the recovery of the ozone layer. However, the anthropogenic release of N_2O might impede ozone recovery. Globally, N_2O emissions (~10 t yr^{-1}) are about ten times higher than all the chlorofluorocarbons (CFCs) combined during their peak emission (Ravishankara et al., 2009). Nonetheless, unlike chlorine- and bromine-containing source gases, N_2O is not regulated, and it is not considered an ODS under the MP, even though it has a positive Ozone Depletion Potential (ODP). The ODP of N_2O is about 0.017, which is comparable to the ODP of hydrochlorofluorocarbons (HCFCs), such as HCFC-123 (0.02), HCFC -124 (0.022), HCFC -225ca (0.025) and HCFC -225cb (0.033) that are currently being phased out under the MP. Furthermore, N_2O behaves similarly to CFCs, it is stable in the troposphere but its by-products, nitrogen oxides (NOx) enable catalytic destruction of the ozone in the stratosphere. The ODP of N_2O is lower than that of CFCs because only 10% of it is converted to the more damaging NOx.

Without action and regulation, the ODP of N_2O will continue to increase throughout this century and is set to potentially overtake the ODSs currently regulated by the MP. The MP has set targets to gradually phase out ODSs and return their atmospheric concentrations to pre-industrial levels. Conversely, nitrogen oxides can have a positive effect of dampening chlorine-catalysed destruction of ozone through the formation of chlorine nitrate ($ClONO_2$), which renders chlorine benign. However, this dampening effect is offset by reactions such as the conversion of chlorine monoxide (ClO) to chlorine (Cl) by NO. Even though the MP does not regulate it, N_2O is currently the largest emitted ODS today and its emissions are unlikely to fall given the reliance of agriculture on nitrogen-based fertiliser and the burning of fossil fuels. N_2O will remain an important gas affecting the ozone layer and radiative forcing if action is not taken.

2.7 How to improve peat soil health

The health of peatlands in an agricultural context can best be described as its ability to sequester C whilst maintaining food production. Whilst abandoning production can be seen as good for C sequestration, completely abandoning cultivated peatlands when they are no longer in agricultural use is not the best solution as significant CO_2 and N_2O emissions from abandoned cultivated peatlands can persist for 20 to 30 years after abandonment (Maljanen et al., 2007). Therefore, when peatlands are no longer productive, they should be rewetted so that they cease to be a large source of CO_2. Rewetting ensures maximum protection of the remaining peat via inducing anoxia and slowing microbial activity allowing C to start accumulating again (Peh et al., 2014). Water table depth is a key regulator of organic matter (OM) decomposition, GHG emissions, and soil subsidence in peatlands (Couwenberg et al., 2010; Malone et al., 2013). The best example of a restored peatland is Wicken Fen, which has shown a reduction in peat loss, increased C sequestration, and biodiversity increase (Peh et al., 2014). Wicken Fen is a nature reserve owned by the National Trust, it is part of the East Anglian Fen and is a remnant of a landscape that has almost disappeared (McCartney & de la Hera, 2004). Rewetting peatlands can potentially sequester significant amounts of CO_2 from the atmosphere. Whilst rewetting provides the best solution, it is not an ideal and feasible option on current productive agricultural peatlands because it requires the cessation of agricultural production. Food production needs to be maintained and improved where possible (Department for Environment, Food and Rural Affairs, 2008).

2.7.1 Water table management

A key strategy in the protection of peatlands and mitigation against CO_2 loss that allows continued farming is the use of elevated water tables, intermittent flooding, and intermittent drainage (Hu et al., 2017). As a compromise, farmers in the UK, Germany, and the Netherlands use elevated water tables to reduce the loss of release of CO_2 from peatlands (Matysek et al., 2019; Musarika et al., 2017). The rate of peat mineralisation and CO_2 fluxes are reduced when the water table is high because water reduces the amount of oxygen available (Oechel et al., 1998). Studies have shown that increasing the water table leads to an immediate reduction in CO_2 emissions (Ballantyne et al., 2014; Chivers et al., 2009; Couwenberg et al., 2010; Moore & Knowles, 1989; Matysek et al., 2019; Musarika et al., 2017; Taft et al., 2017; Zeitz & Velty, 2002). Increased water availability has also been shown to reduce N_2O emissions from peatlands; average N_2O emissions from drained peatland can be three times higher than in

flooded peatland (Hu et al., 2017). Frolking et al. (2011) and Zou et al. (2005) reported that daily average N_2O emissions were three times higher when peatland soil was drained. The increase in N_2O under drained conditions has been attributed to the higher decomposition rate of peat under the oxic conditions which lead to high mineral N content. Mineral N is an essential substrate to produce N_2O (Maljanen et al., 2007).

Whilst increased water table is effective at reducing CO_2 loss from peatlands, water table management is not currently used effectively. Farmers in the UK maintain water tables that are too low, effectively over-draining the peatlands, because there are concerns that higher water tables will have an impact on plant productivity and the use of heavy machinery (Matysek et al., 2019; Musarika et al., 2017). However, these concerns might be ill-founded, Musarika et al. (2017) showed that not only did a higher water table of 30 cm reduce CO_2 loss, but it increased crop productivity, demonstrating that the water table can be raised without loss of crop production. Nevertheless, this study was only conducted on radish. Other studies have shown that raising the water table can negatively impact crop productivity. Renger et al. (2002) showed a 10% productivity reduction on grassland when the water table was elevated. Matysek et al. (2019) showed water table close to the soil surface significantly reduced the growth of celery (*Apium graveolens* L.). There is a scarcity of studies that have included food crops in elevated water table experiments. Stanley and Harbaugh (2002) conducted a study on *Caladium x hortulanum*, a poisonous perennial bulb that flowers in the summer; their study showed increased water table has no impact on plant productivity.

2.7.1.1 Negative effects of elevated water tables

Higher water tables can affect heavy farm machinery and can also lead to increased disease prevalence (Katan, 2000). Furthermore, elevated water tables can lead to increased CH_4 emissions as the increased water availability leads to an abundance of methanogens (Yang et al., 2014; Moore & Knowles, 1989). However, in the short term, the CH_4 emissions will be lower than in natural peatlands and in rice paddies (Hu et al., 2017). Hu et al. (2017) studied CH_4 emissions under flooded conditions for 24 weeks but methanogenesis remained suppressed throughout the experiment.

2.7.2 No-tillage, reduced tillage

For millennia, tillage has been synonymous with agriculture, and it was perceived as improbable to grow crops without tillage (Triplett Jr & Dick, 2008). However, the adoption of

alternative methods such as reduced tillage or conservation tillage and the use of cover crops can reduce the impact of agriculture on cultivated peatlands (Locke & Bryson, 1997). Reduced tillage or no-tillage makes use of advances in weed control through modern herbicides that have proved highly effective. Due to the successful use of herbicides, tillage is only needed occasionally to smooth out the ruts created during harvest (Locke & Bryson, 1997). Using reduced tillage or no-tillage, it is possible to grow crops with a reduced impact on peatlands (Triplett Jr & Dick, 2008). Reduced tillage or no-tillage agriculture can lead to the accumulation of SOC, which improves soil aggregation, water holding capacity, and nutrient cycling (Teasdale et al., 2007).

2.7.2.1 Issues with no-tillage and reduced tillage

No-tillage or reduced tillage presents higher equipment costs (e.g., seed drills and strip-tillers), which are beyond the economic reach of most farmers in the UK (Parliamentary Office of Science and Technology, 2015). Further issues include that continued use of herbicides will potentially lead to the development of herbicide-resistant weeds that could compete with crops and that these herbicides will also be left in the soil where they can cause environmental damage (Locke & Bryson, 1997).

2.7.3 Use of cover crops

The growing of winter cover crops can lead to increases in SOC; winter cover crops ideally should be a mixture of legumes (e.g., vetch and clover and non-legume crops e.g., wheat, oats (*Avena sativa* L.), and rye (*Secale cereale* L.) with a diverse rotation with perennial crops e.g., ryegrass (*Lolium spp* L.) during fallow periods (Locke & Bryson, 1997; Teasdale et al., 2007). Legumes used as cover crops protect the soil and will provide green manure when residue management is in effect. Residue management is a practice that leaves at least 30% of crop residue on the soil surface which will protect the soil from erosion and loss of moisture (Locke & Bryson, 1997; Triplett Jr & Dick, 2008). Furthermore, if cover crops are used in conjunction with reduced-tillage or no-tillage, they can potentially facilitate the degradation and biochemical transformation of herbicides (Locke & Bryson, 1997).

2.7.4 Rewetting peatlands for paludiculture

Paludiculture is the cultivation of biomass on wet peatlands which enables the conservation of peatlands, therefore mitigating the negative environmental impact of drained peatlands and

leading to the preservation of unique wetland species and habitats (Abel et al., 2013; Joosten et al., 2016). It has been described as an innovative land use concept that allows for the cultivation of crops that can either be used for food, energy, or other industrial use (Knox et al., 2015; Joosten et al., 2016; Wichtmann et al., 2016). The core objective of paludiculture is the maintenance and restoration of ecosystem services that are provided by wet peatlands (Joosten et al., 2016). Whilst paludiculture benefits peatlands by reducing the environmental impact of crop production, it presents a challenge as economically important food crops such as potatoes (*Solanum tuberosum* L.), carrots (*Daucus carota* L.), wheat, sugar beet (*Beta vulgaris* L.), salad crops, and maize (*Zea mays* L.) are not suitable for growth under paludiculture (Joosten et al., 2016). The production of such crops necessitates the drainage of peatlands due to the crops being intolerant of elevated water tables which can lead to reduced nutrient uptake, therefore causing stunted growth and, in most cases, exposing the crops to plant diseases such as *Aphanomyces*, a genus of water mould that can damage crop yield (Hossain et al., 2012; Musarika et al., 2017). Due to many commercially grown crops being intolerant of excess water, rewetted peatlands tend to be taken away from agricultural production and become reserved for nature conservation (Joosten et al., 2016; Karki et al., 2019; Knox et al., 2015).

2.7.4.1 Alternative crops to grow on peatlands

To overcome the limitations of unsuitable commercially grown crops, the use of alternative crops in paludiculture has been advocated for as a viable solution to the problem (Joosten et al., 2016; Karki et al., 2019; Knox et al., 2015). However, a move away from these ‘tried and tested’ crops to alternative crops represents a fundamental challenge for food production in countries such as the UK. Whilst there is a wide range of crops that can be grown through paludiculture, only a few of them are suitable food crops. Farmers in the Eurasian boreal zone produce a range of berries and mushrooms on wet peatlands, crops such as wild rice (*Zizania aquatica* L.) are grown in the USA, Canada, Hungary, and Australia as a viable alternative to rice (*Oryza sativa* L.), and bogbean (*Menyathes trifoliata* L.). However, the suitability of bogbean as food is debatable due to toxicity if consumed in substantial amounts (PFAF, 2020). The young stems of calamus (*Acorus calamus* L.) can be used for food but mature plants are toxic, therefore calamus is banned commercially in the USA. Sweetgrass (*Hierochloe odorata* (L.) P.Beauv.) is grown for food in Europe, and sago palm (*Metroxylon sagu* Rottb.) in Malaysia (Joosten et al., 2016). Agricultural producers in Belarus grow cranberry (*Vaccinium*

macrocarpon Aiton.) and has been commercially successful (Joosten et al., 2016). Rice is the most widely consumed staple food in most parts of the world, feeding a third of the world's population, it is the most successful paludiculture crop (Ghadirnezhad & Fallah, 2014; Khush, 1997). However, rice is intolerant to frequently low temperatures, therefore, making it unsuitable for locations such as the UK (Ghadirnezhad & Fallah, 2014). Nevertheless, with technological advancements i.e., genetically modified organisms (GMO), it may be possible to breed species of rice suitable for a range of abiotic stresses (López-Arredondo et al., 2015). Whilst wild rice grows well in the temperate regions of the USA and Canada where it is a commercially successful crop, it is not related to *O. sativa*, and it requires different growing methods. Furthermore, wild rice has issues, such as susceptibility to disease, slow maturation, and seed shattering which means greater care needs to be taken when handling plants. Furthermore, yield is significantly lower thus requiring more land to grow enough crops to be economically competitive and to ensure food security (Hauan, 2015). In the future, these issues could be addressed by either breeding or genetic modification (Hauan, 2015; Knox et al., 2015).

2.7.4.2 Greenhouse emissions from crops grown on wetlands.

The cultivation of rice on wetlands limits ecosystem respiration (ER) and the rice paddy becomes a sink for CO_2. Whilst paludiculture can successfully sequester CO_2 from the atmosphere, it can lead to increased CH_4 fluxes. CH_4 has a GWP 25 times greater than CO_2 over a 100-year time scale (Forster et al., 2007). Therefore, even low rates of CH_4 emissions can potentially offset the benefit of CO_2 sequestration in terms of the net GHG effect (Knox et al., 2015). Rice paddies cover 11% of the global cropland area and account for 11% of global anthropogenic CH_4 emissions. The GWP of GHG emissions from rice cultivation is more than three times higher than those from either wheat or maize (Jiang et al., 2019). Genetic manipulation techniques are currently being researched that would allow rice to reallocate photosynthate from root exudates to the grain, therefore limiting the activity of methanogens which will reduce CH_4 emissions from the soil (Jiang et al., 2019). An alternative to growing crops on wetlands could be meat production e.g., game fowl and fish, which will not require the seasonal drainage of the wetlands for harvest as is the case with rice paddies (Joosten et al., 2016).

Paludiculture has been successful in countries, such as Poland, where common reed (*Phragmites australis* (Cav.) Trin. ex Steud) is grown and harvested for thatching. In Canada, significant GHG mitigation potential has been shown by growing cattail (*Typha spp.* L.) for

fuel (coal displacement in power stations), nutrient removal, and reclamation (Grosshans et al., 2011). Other plants such as black alder (*Alnus glutinosa* (L.) Gaertn.) can be grown for raw materials such as timber for furniture and construction, and fuel. Furthermore, peatmoss (*Sphagnum spp.* L.) can be grown for use as an alternative for white peat/peat moss in horticulture (Joosten et al., 2016). The extraction of white peat damages important ecosystem services (Bustamante et al., 2008; Joosten et al., 2016). Moreover, white peat is a finite resource that is close to depletion in western and central European countries (Joosten et al., 2016). As an alternative, fresh peatmoss biomass has similar physical and chemical properties to white peat, therefore, making it a suitable alternative (Joosten et al., 2016). Sphagnum farming has been successful on previously drained and wasted peatlands. Importantly, replacing white peat with sphagnum peatmoss has no impact on the quality of cultivated plants (Joosten et al., 2016).

Energy crops on wet peatlands can become an important substitute for fossil fuels (Joosten et al., 2016). There is a drive towards the use of biofuels, with most of these crops being cultivated on drained peatlands. Unfortunately, the production of biofuel crops such as maize and oil palm (*Elaeis guineensis* Jacq.), on drained peatland is favoured by the KP, the EU, and other governments who do not account for massive peat carbon losses associated with these crops (Joosten et al., 2016). Growing biofuel crops on drained peatland produces more emissions than burning fossil fuels, 100 t CO_2e compared to 800 t CO_2e for generating the same amount of energy from biofuels (Joosten et al., 2016).

While paludiculture has been recommended in recent reports by the Food and Agriculture Organization (FAO) and the Intergovernmental Panel on Climate Change (IPCC) as a probable management option for drained peatlands (Biancalani & Avagyan, 2014; IPCC, 2014), there is limited data on the long-term effectiveness of paludiculture to mitigate CO_2 emissions in the long term (Günther et al., 2015; IPCC, 2014; Karki et al., 2016). The few studies that have focused on paludiculture, have primarily focused on rewetted peatlands growing natural vegetation and low-input management (e.g., no fertiliser and on harvest annually (Beetz et al., 2013; Günther et al., 2015). Günther et al. (2015) argues that for paludiculture to become socioeconomically viable, more intensive management and dedicated biomass crops will be needed. The range of non-food crops that can be grown through paludiculture is extensive. However, as there are a few food crops that can be grown through paludiculture compared to non-food crops; this makes paludiculture an unfeasible option for food security and sustainability in countries such as the UK. Nevertheless, it can be commercially viable thus

allowing farmers to continue to make a living from their land whilst simultaneously protecting it and reducing the GHG burden.

Whether paludiculture can or cannot provide food sustainably and be commercially successful whilst ensuring food security leads to multifaceted answers. In countries like the UK, in temperate zones, food production through paludiculture is going to be more challenging given that few food crops can be grown through paludiculture. There is a limited list of crops that can be grown for food e.g., cloudberry (Rubus chamaemorus), cranberry, water chestnut (*Trapa bicornis* Osbeck. & *Trapa natans* L.), and wild rice. From the above list, wild rice stands out as one crop that could become a staple. For paludiculture to be truly viable in the UK, for example, it will require significant diet modifications.

2.7.4.3 Negative effects of cover crops

Whilst cover crops can be beneficial for soil health, they can inhibit crop emergence, host pathogens, and insects (Koch, et al., 2012; Pullaro, et al., 2006). Furthermore, cover crops can compete for soil moisture and nutrients (Locke & Bryson, 1997). The use of cover crop residue as green manure introduces fresh organic matter (FOM) into the soil. FOM applied to organic soil risks priming the release of GHG emissions, therefore this could make them unsuitable as a mitigation measure against soil C loss (Bingeman et al., 1953; Bader et al., 2018; Wang et al., 2015). FOM provides an additional C source for soil microorganisms which leads to the increased decomposition of existing SOM i.e., a priming effect (PE) (Bingeman et al., 1953). Older SOC has a lower decomposition rate than younger SOC because it is devoid of oxygen-rich compounds such as polysaccharides and amino acids (Bader et al., 2018; Wang et al., 2015). Consequently, this older SOC is not a preferred energy source for the microorganisms found in peatlands and remains largely untouched as it requires more energy to breakdown that is provides to the microorganisms. However, FOM provides an energy source for soil microorganisms that can enables them to degrade recalcitrant older SOC, in effect lowering the energy requirements needed to break the older SOC (Bader et al., 2018; Wang et al., 2015). Wang et al. (2015) found that the quality of FOM will influence the effect of the PE in the soil. However, this experiment was conducted on mineral soil, so the results might be different on organic soils such as peatlands. In addition to the PE of FOM, there is an issue of rhizodeposition also inducing SOM priming. Specifically, amino acids typically found in root exudates can increase the decomposition of peat (Bader et al., 2018; Norberg et al., 2016). Crops such as root vegetables (e.g., carrots) have been shown to increase N_2O release from the

soil (Maljanen et al., 2007; Weslien et al., 2012). There is no recommended crop as mitigation on organic soils, however, grasslands have shown more resilience than croplands (Norberg et al., 2016).

2.7.5 Benefits of peatland restoration

Raising the water table to the surface can have some drawbacks including increased CH_4 fluxes; since restoration, Wicken Fen produces annually an average of 3.8 to 4.0 g CH_4 m^{-2} yr^{-1} (Kaduk et al., 2015). However, there are benefits to the restoration of peatlands, the biodiversity at Wicken Fen has increased exceptionally providing a home to over 9000 species of flora and fauna (Jones & Comfort, 2021). In addition to increased biodiversity, restored peatlands can enhance the ecosystem services they provide including the mitigation of climate change through the sequestration of C. In addition to C sequestration, Worrall, et al. (2019) observed that when peatlands are restored, they can potentially lead to the cooling of the local environment. However, for these benefits to be fully realised, water tables closer to the surface will be required, which is not compatible with current agricultural practice and types of crops. Furthermore, restored peatlands can help improve water quality and rewetting prevents declines of water quality (Martin-Ortega, et al., 2014). Another benefit of restoring peatlands is cultural. Restored peatlands can improve local tourism and benefit the local economy (Martin-Ortega, et al., 2014).

2.8 Measuring greenhouse gas (GHG) fluxes

Weather a peatland is restored or under arable agriculture various gases are emitted from the soil. There are different ways these gases can be measured. However, these are generally classed into two main techniques, i) micrometeorological techniques e.g., eddy covariance and ii) chamber techniques. Each method has its advantages and disadvantages depending on where it is deployed.

2.8.1 Micrometeorological techniques

Micrometeorological techniques offer many advantages as they can be deployed at a landscape level and provide continuous measurement (Baldocchi et al., 1988; Zamolodchikov et al., 2011). Eddy covariance is commonly used for direct vertical flux measurements that are independent of atmospheric stability (Abalos et al., 2016). Because eddy covariance is placed in situ, it has no impact on the studied environment. However, the use of eddy covariance has

drawbacks associated with it. The level of air turbulence can affect the measured flux; the level of CO_2 flux can approach zero if the level of turbulence that is measured by the friction velocity drops down to zero (Goulden et al., 1996). For this reason, micrometeorological techniques can lead to a significant underestimation of CO_2 fluxes. Nevertheless, equations can be used to correct this. However, different users of micrometeorological techniques use different values which can lead to inconsistency in the reported figures (Goulden et al., 1996). The biggest drawback of micrometeorological techniques is replication as it measures on a field scale (di Perta et al., 2020). Due to the prohibitive cost of equipment, most users only have access to one setup.

Historically, eddy covariance has issues when it comes to the fluxes of CH_4 and N_2O as the techniques could not reliably measure the fluxes. However, thanks to recent advancements in narrowband infrared technology, it is now possible to reliably measure the fluxes of N_2O and CH_4 using eddy covariance (Pan et al., 2022). Nonetheless, the instrumentation needed to measure N_2O and CH_4 fluxes reliably is expensive and has high energy requirements which make it challenging to deploy in areas that have no mains power supply.

2.8.2 Chamber methods

Chamber techniques are commonly used for gas flux measurements because they are cheaper to run and require less technical expertise to deploy compared to micrometeorological techniques. Furthermore, they are suited for finer scale measurements, which is not possible with micrometeorological techniques. There are two types of chamber techniques i) open dynamic chambers (steady-state flow-through chambers), and ii) closed dynamic chambers (non-steady-state flow-through chambers). Flux concentration in open dynamic chambers is calculated from the concentration difference between the air inflow and outflow. In closed dynamic chambers, the flux is calculated from concentration changes within the chamber's headspace.

Whilst chamber techniques can be cheaper and suitable for fine-scale spatial measurement, opponents argue that they can alter the local environment of the subject being studied if left on too long and they are extremely localised, therefore not truly representative of the wider landscape. Manual chambers are often difficult to deploy for long-term studies (Baldocchi et al., 1988). However, this can be overcome by deploying automated chambers. Automated chambers are widely available such as the LI-COR autochambers (Long Term Chamber –

Model 8100-104). These chambers can run automatically, throughout the day for the duration of a study. This allows for higher resolution measurements which is one of the problems of manual chamber operations that tend to be used with a reduced frequency which leads to fine flux variations being completely missed. Whilst automated chambers allow for a higher frequency of data collection, they can be costly to acquire, which is the reason manual chambers are still common. Furthermore, there is a limit to the number of autochambers that can be deployed, the LI-COR (LI 8150) multiplexer only allows up to 16 autochambers and the latest model, the LI-COR (LI 8250) multiplexer can accept a maximum of 36 ports. Chamber techniques are more suited for mesocosm studies as measurements are localised within that chamber.

2.8.3 Measurement of priming effect (PE)

In addition to measuring the GHG fluxes from the soil, the PE of added organic matter to the soil can be measured. The measurement of the PE of FOM is done using 13C or 14C labelled plant material (Bader et al., 2018; Wang et al., 2015). Wang et al. (2015) used maize which they incubated with 50 L of 99.3 atom % $13CO_2$ for 50 to 66 days. After that, the crops continued their growth until being harvested 28 days after labelling. Bader et al. (2018) used similar methods to label their plants. After the application of 13C or 14C labelled FOM, CO_2 concentration and $\delta 13CO_2$ or $\delta 14CO_2$ values can be analysed using a C isotope analyser.

2.9 Conclusions

There is a pressing need to mitigate GHG emissions from cultivated peatlands. However, challenges remain in terms of how this can be achieved without affecting food production, especially given the global population is growing and therefore food production will need to increase. Emissions from arable peatlands differ from country to country, farm to farm, and field to field due to different management regimes, soil properties, local climatic conditions, and government-level politics and policies. Nevertheless, a universal and coherent mitigation solution is necessary unless reasonable alternatives to peatland farming can be found. All soils, both organic and mineral have the potential to store organic C, but to achieve this, changes in agricultural practices are needed.

Chapter 3. Effects of crop residue addition on carbon dioxide, methane, and nitrous oxide emissions from cultivated peat soils

Abstract

Due to intensive agriculture, cultivated peatlands no longer function as C sinks but typically become major greenhouse gas (GHG) emission hotspots (i.e., CO_2 and N_2O). In 2021, agriculture was the second-largest sector in Europe in anthropogenic GHG emissions contributing 11% of the total. This study aimed to evaluate the response of GHG fluxes in cultivated peat soils in response to the addition of fresh organic matter (FOM) and water table manipulation. The study was conducted using large intact cores collected from a commercial farm in the East Anglian Fens. The cores were divided into two water table treatments, 20 and 50 cm from the surface of the soil. Half of the cores had added FOM in the form of barley (*Hordeum vulgare* L.) straw. There were significant differences between the cores with or without FOM in both water table treatments. The CO_2 fluxes in the 20 cm water table were lower than in the 50 cm water table. Overall, the cores with FOM had higher CO_2 fluxes. However, it was not clear whether this was due to the increased decomposition of the peat or due to the decomposition of the added FOM. The addition of FOM is beneficial against the fluxes of CH_4 and N_2O, in the current 'business as usual' set up of 50 cm water table or greater. The addition of FOM could help reduce atmospheric CH_4 and N_2O as the addition of FOM led to the consumption of both GHGs. Focusing on CO_2 alone, then the addition of FOM might not be the best option. Nevertheless, if an integrated approach is to be taken, then adding FOM might have benefits as it can lead to reduced N_2O fluxes and the consumption of CH_4.

3.1 Introduction

Peat soils are defined as high quality due to their capacity to accept, store, and recycle water, nutrients, and energy (Gregorich et al., 1994; Sanchez et al., 2003). For this reason, peat soils are excellent at supporting primary productivity, and consequently, they are extensively exploited for food production. Globally, over 20% of peatlands are currently used for arable agriculture (Strack, 2008). In the UK, peat soils, such as those in East Anglia Fens, are commercially important for food production, and they form the backbone of the UK's agricultural output. The Fens account for close to 50% of the most productive ("grade 1") agricultural land in England and produce more than 37% of the total vegetable production in England (NFU, 2019). If this is further broken down, a third of fresh vegetables and a fifth of crops such as potatoes (*Solanum tuberosum* L.), sugar beet (*Beta vulgaris* L.), flowers, and bulbs, are produced on fenland (NFU, 2019). The East Anglia Fens includes a series of lowland peatlands that have been intensively drained since the 17th century (NFU, 2019; McCartney & de la Hera, 2004). Whilst it is impossible to put a monetary value on peat soil, the UK government estimates that the degradation of UK soils in general, costs over £1 billion a year (Parliamentary Office of Science and Technology, 2015). Unfortunately, there are no figures for peat degradation alone, but the NFU (2019) reported that the total agricultural production from The Fens alone is worth £1.23 billion.

3.1.1 The carbon (C) storage capability of peat soils and status

Globally, peatlands cover an estimated area of 400 Mha representing 3% of the global land surface, but they play a significant role in carbon (C) sequestration (Frolking et al., 2011; Joosten & Couwenberg, 2009). Since the last glaciation, peat has sequestered a significant amount of C, estimated to range from 400 to 500 Pg (Belyea & Clymo, 2001). 30% of the global soil C storage is present as soil organic matter (SOM), which is a significant C reservoir (Fontaine et al., 2003). Peatlands are natural accumulators of C from organic plant material which stays intact under anaerobic conditions, due to the presence of waterlogged conditions (Berglund & Berglund, 2010). In the case of bogs (raised and blanket), acidic conditions are another factor that retards plant matter from decaying because they are rain-fed (ombrotrophic). Fens, on the other hand, are not acidic as they get their water primarily from mineral-rich groundwater (minerotrophic) (Griffiths et al., 2019; Pedrotti et al., 2014).

Due to intensive agriculture, cultivated peatlands are no longer C sinks but are significant sources of anthropogenic C and dominate carbon dioxide (CO_2) and nitrous oxide (N_2O) emissions from agricultural land (Foresight, 2011; Gilbert, 2012; Parish et al., 2008). In the UK, arable peatlands produce an estimated 7600 kt CO_2e yr^{-1}, 32% of total UK peatland greenhouse gas (GHG) emissions (Evans et al., 2017). In East Anglia, substantial losses of over 3370 km^2 of lowland fens have been recorded since drainage began in the 17th century, (McCartney & de la Hera, 2004). Since drainage of the East Anglian Fens began, it is estimated that between 84% and 95% of the East Anglian Fens have been lost through subsidence and oxidation of the peat (Frolking et al., 2011; McCartney & de la Hera, 2004). Whilst a considerable proportion of the peatlands have been converted for agricultural use, some of the land has been reclaimed for domestic and commercial properties and other infrastructure such as roads.

The rate of peat formation varies in different ecosystems, the rates are linked to the prevailing hydrological conditions. Hydrology is subsequently affected by climatic factors such as precipitation, evaporation, and temperature. Furthermore, factors including topography, land use, and vegetation cover also affect peat formation rates (Belyea & Clymo, 2001; Moore, 1987). Measurements of the net primary productivity of distinct types of wetlands from various parts of the world show a significant degree of variation. Wetlands of cattail (*Typha spp.* L.) are amongst the most productive ecosystems that produce organic matter at rates of 5-6 kg m^{-2} yr^{-1} and in extreme cases 15 kg m^{-2} yr^{-1} has been recorded (Belyea & Clymo, 2001; Moore, 1987). Wetland forests in Florida achieve 2-3 kg m^{-2} yr^{-1} but the peatlands in the temperate and boreal regions rarely exceed 1 kg m^{-2} yr^{-1} (Moore, 1987). Therefore, peat soils are not an immediately replaceable resource, therefore they need preservation if continued agricultural use is to persist whilst also being important reservoirs of C (IPCC, 2013). Peat soils have played a vital role in our climate system for centuries and will continue to play a significant role in the mitigation of climate change.

3.1.2 Impact of agriculture on climate and mitigation

The European Environment Agency (EEA) (2021) reported that agriculture in Europe was the second-largest sector contributor to GHG emissions, accounting for 11% of the emissions, whilst N_2O emissions accounted for 41% of this figure (European Environment Agency, 2021). In the UK, there has been a commitment from successive governments to reduce the anthropogenic impact on cultivated peatlands. Unfortunately, due to the devolved nature of UK

governance, there is no single authority responsible for *Peatland Policy* at the UK level (IUCN UK, 2021). In 2021, the Department for Environment, Food & Rural Affairs (DEFRA) launched the *England Peat Action Plan* which sets out the government's long-term vision for the management, protection, and restoration of peatlands (DEFRA, 2021). Several mitigation measures are explored such as i) longer crop rotation, ii) use of no-tillage or reduced tillage methods, iii) growing winter cover crop e.g., wheat (*Triticum aestivum* L.), barley (*Hordeum vulgare* L.), clover (*Trifolium spp.* L.), and vetches (*Vicia spp.* L.) or other native legumes instead of leaving it fallow, then ploughing in the crop residue as green manure or fresh organic matter (FOM), and iv) the raising of water tables to reduce peat oxidation (Locke & Bryson, 1997; Musarika et al., 2017; Triplett Jr & Dick, 2008; Ostle et al., 2009; Parliamentary Office of Science and Technology, 2015; Wen et al., 2019)

3.1.3 Water table as mitigation of greenhouse gas (GHG) flux

Bringing the water table closer to the surface can lead to a significant decrease in the fluxes of CO_2 as anoxic conditions are created. Raising the water table allows the reduction of peatland degradation by decreasing oxidation of the soil, subsequently changing the microbial community towards methanogens (Moore & Dalva, 1993). Water tables that are 30 cm or more were reported to have no substantial effect on the fluxes of methane (CH_4) (Matysek et al., 2019; Musarika et al., 2017; Taft et al., 2017). However, with a water table 20 cm or less from the surface, higher methane fluxes are expected as the anoxic environment is brought closer to the surface (Evans et al., 2021). Furthermore, water table depth will have an impact on the fluxes of N_2O. A water table closer to the surface will reduce the amount of oxygen (O_2) in the soil, therefore denitrification will become prevalent leading to N_2O fluxes as microbes use nitrate (NO_3^-) as a terminal electron acceptor instead of O_2 (Robertson & Groffman, 2015; Morley & Baggs, 2010; Russow et al., 2009; Wen et al., 2019).

Increasing water table depth has been shown to either increase crop yield or decrease it depending on the type of crop. The elevated water table in grassland led to a 10% drop in crop yield while in horticultural studies, celery (*Apium graveolens* L.) shoots both wet and dry weight were on average 19% lower when the water table was set at 30 cm compared to that at 50 cm (Renger et al., 2002; Matysek et al., 2019). However, increased crop yield was observed when radish (*Raphanus sativus* L.) was grown in elevated water table depth of 30 cm and ryegrass (*Lolium perenne* L.) grown at an elevated water table led to significantly greater yield (Musarika et al., 2017; Berglund & Berglund, 2010). Regardless of the reported minor losses

in crop yields, raising water table depths is a viable long-term mitigation measure if crop production is to be maintained on peatlands.

3.1.4 Green manure or fresh organic matter (FOM) and the status of its application on peat soil (directly or leftover FOM)

The growing of winter cover crops has the potential to increase soil organic carbon (SOC) as the crop can provide cover to the soil and the residue can be incorporated into the soil before the next growing season. There are a variety of crops that can be grown as winter cover crops, and each has its perceived benefits. Locke & Bryson (1997) and Teasdale et al. (2007) suggested that the crops grown should be ideally a mixture of legumes such as vetch and clover and non-legume crops like wheat, oats (*Avena sativa* L.), and rye (*Secale cereale* L.). In addition, there is a need for a rotation with perennial crops e.g., ryegrass (*Lolium spp.* L.) during fallow periods. Furthermore, as an added benefit, cover crops used in conjunction with reduced tillage, or no-tillage can potentially facilitate the degradation and biochemical transformation of herbicides and molluscicides that would have been long-lived in the soil (Locke & Bryson, 1997).

Maintaining and improving soil quality is crucial for the sustainability of future agricultural systems (Gregorich et al., 1994). The incorporation of crop residue from the cover crop could help improve or maintain the soil quality and further prolong its suitability for agriculture by adding new C to the soil. However, crop residue or FOM is an essential source of energy for soil microbes. Therefore, its addition to the soil can potentially lead to the rapid growth of specialised microorganisms that can decompose the FOM, consequently increasing the rate of SOC mineralisation i.e., a priming effect (PE) (Fontaine et al., 2003). Bader et al. (2018) and Wang et al. (2015) showed FOM can stimulate a PE on existing SOC i.e., a change in the decomposition rate of existing SOC. The absence of FOM limits available energy for soil microorganisms which can help the soil by increasing the stability of SOM in the deeper layers of soil. Even though the incorporation of FOM, normally from the previous crop is a widespread practice, the effects of FOM on GHG emissions and soil quality are not widely studied.

Existing studies provide limited insight into the effect of FOM on GHG emissions. Bader et al. (2018) conducted their study on peatland that used to be cultivated but had since been converted to other types of land use i.e., forest, grassland, and cropland. The study by Wang et al. (2015)

used forest soils, as opposed to cultivated peatland. Whilst these studies highlight the issues and benefits of FOM, there is no information on how this would apply to cultivated peatlands in the UK such as The Fens in East Anglia.

The PE in the study by Bader et al. (2018) was shown to be dependent on the age of the SOC. Older SOC did not respond to the PE but the effect was more pronounced in young SOC especially from pristine peatlands (Bader et al., 2018). However, the PE effect was not evaluated against commonly used mitigation measures on cultivated peatlands e.g., elevated water table. In the study conducted by Wang et al. (2015), the FOM was not directly applied as would happen on an active farm. The maize (*Zea mays* L.) crop was separated into stalks and leaves before being dried and milled before being applied to the soil. Furthermore, these severely controlled studies were not conducted under prevailing field conditions. Wang et al. (2015) bulked their soil together and sieved it to remove any existing plant matter whilst Bader et al. (2018), used two types of corers for the distinct types of soil in the experiment, undoubtedly creating different soil bulk densities.

To address these gaps in understanding, in this study, the experimental setup resembled the field conditions as closely as possible by using intact cores. This study used a common cereal crop grown in the East Anglian Fens. The crop chosen was barley. This study did not use 13C/14C labelled plant material, instead, it assessed the fluxes resulting from the application of the FOM and compared the resultant fluxes to the fluxes from the cores without FOM by calculating the mass balance due to loss over time. The mass balance calculation can show if any of the added FOM is either lost as CO_2 or remains in the soil.

3.1.5 Specific aims and objectives

This study aimed to assess GHG emissions (CO_2, CH_4, and N_2O) from cultivated peat soils, evaluate the response of these emissions when FOM is added into the peat soil, and assess this response to water table manipulation as a mitigation measure. The study specifically targets the FOM of a commercially important crop that is extensively used as a cover crop in winter in the UK. The hypotheses are: i) a water table depth of 20 cm will lead to a reduction in CO_2 fluxes but, ii) it will stimulate methanogenesis by increasing the anoxic environment, therefore an increase in CH_4 fluxes is expected, iii) CO_2 fluxes will be high in the water table depth of 50 cm due to a large amount of soil exposed to an oxygen-rich environment, iv) FOM will lead to increased CO_2 in both water table depths, this is most likely for the 50 cm water table depth,

and v) increased N_2O fluxes are expected in the 20 cm water table as denitrification becomes dominant in the absence of O_2, but vi) as no additional N was added, large N_2O flux differences are not expected between the treatments.

3.2 Materials and Methods

3.2.1 Study site

3.2.1.1 Location of sampling site and experimental site

The experiment was conducted at the UK Centre for Ecology & Hydrology (UKCEH) facilities located near Bangor University's Henfaes Farm in Abergwyngregyn (Grid Reference: SH 65486 73156, UTM: 30U 432126,5899267). This study utilised soil samples collected from Rosedene Farm in Methwold, Figure 3.1, which is located on the edge of the Norfolk Fens in East Anglia, UK (Grid Reference: TL 68424 95144, UTM: 30U 329151, 5822762).

Figure 3.1 Rosedene farm is in the Village of Southery in Norfolk, England, 250 miles to the east of Bangor and about 90 miles to the north of London.

Rosedene Farm and similar farms in the area are lowland fen that is characterised by a flat topography of between 0-1%. These farms were created after extensive post-war drainage as part of an agricultural expansion program from the late 1930s and 1940s (Musarika et al., 2017; Short, 2007).

3.2.1.2 Climatic conditions of the farm compared to the experimental setup site

The mean annual rainfall for Rosedene Farm, from where the soil samples were collected, and surrounding areas from the years 1981 to 2010 is 665 mm, with an average of 121 days of rainfall per year and the mean annual maximum temperature is 14.5 °C, while the mean annual minimum temperature is 4.8 °C (Met Office, 2020a). At Henfaes Farm where the experiment took place, the mean annual rainfall from 1981 to 2010 is higher at 1100 mm, with an average of 151 days of rainfall, making it wetter than Rosedene Farm. The mean annual maximum temperature is not too dissimilar at 13.7 °C and the mean annual minimum temperature is warmer at 7.6 °C (Met Office, 2020b).

3.2.1.3 Field conditions at Rosedene Farm

The fields at Rosedene farm have boundaries of willow bushes (*Salix spp.* (L.) nom. cons.) forming dense hedges that were planted to minimise loss of soil through wind erosion. Each field is separated by dykes that are connected to water reservoirs and are used for water table management, Figure 3.2. The *Downham & Stow Bardolph Internal Drainage Board* is responsible for flood protection and land drainage; they conduct continuous maintenance and surveys to ensure that the water levels are kept at safe levels (Downham Market Group of Internal Drainage Boards, 2019).

Figure 3.2 Field covered in stinging nettles (Urtica dioica L.) where the samples were collected showing (A) a dyke filled with water which is used for water table manipulation and (B) the Willow (Salix spp. (L.) nom. cons.) borders that are used for protection against wind erosion.

Currently, Rosedene Farm is owned by G.S. Shropshire & Sons Ltd, and they specialise in growing a variety of crops such as lettuce (*Lactuca sativa* L.), celery, potatoes, onions (*Allium cepa* L.), sugar beet (*Beta vulgaris* L.), maize, wheat and barley which are amongst the most commonly grown crops in The Fens (G's Growers Ltd, 2018; NFU, 2008; Wood et al., 2019).

3.2.2 Collection of soil samples

The soil cores were collected using similar methods to previous work conducted on these cultivated peatlands from Rosedene Farm e.g., Musarika et al. (2017) and Matysek et al. (2019). The intact cores for this experiment were collected using PVC pipes with a diameter of 200 mm and a length of 500 mm. The samples were collected from the farm on the 4th of October 2018. The cores were collected from random locations within a single field. At the time of soil sample collection, there were no crops in the field, but it was covered in weeds, most notably stinging nettles (*Urtica dioica* L.), Figure 3.3.

Figure 3.3 The PVC pipe used as a core is shown at the time of soil sample collection with the prominent stinging nettles (Urtica dioica L.) weeds in the background.

The core collection method ensured that there was minimal disturbance to the soil samples. In summary, the PVC pipes were driven into the soil covering the whole pipe before being excavated. Overall, 16 intact cores were extracted and transported to the UKCEH facilities in Abergwyngregyn. The cores were stored outside covered in black bin bags to preserve their moisture for 3 days before they were buried into the soil for the mesocosm setup.

3.2.2.1 Soil characteristics

This study used soil samples identical to Wen et al. (2019) from the same field. The soil is classified as Earthy Sapric Fen Soils under the UK soil taxonomy or under the US soil taxonomy, they are a Typic Haplosaprist with a humification score of H9 on the von Post scale (Wen et al., 2019). The soil characteristics of the soil are summarised in Table 3.1.

Table 3.1 Soil Characteristics for the lowland organic cultivated soil.

Depth (cm)	**Bulk density (g cm^{-3})**	**Organic matter (OM) (%)**	**Total carbon (C) (%)**	**Total nitrogen (N) (%)**	**C:N**	**pH (H_2O)**	**Electrical conductivity (EC) (µS/cm)**
0-10	0.32 ± 0.03a	78.5 ± 0.5b	50.7 ± 0.4b	2.71 ± 0.04a	18.7 ± 0.2b	6.45 ± 0.10	394 ± 120
10-30	0.31 ± 0.01a	78.7 ± 0.7b	50.5 ± 0.3b	2.71 ± 0.03a	18.6 ± 0.1b	6.29 ± 0.08	558 ± 119
30-50	0.22 ± 0.02b	84.1 ± 1.2a	54.8 ± 0.7a	2.45 ± 0.07b	22.4 ± 0.9a	6.09 ± 0.11	377 ± 30

All values mean ± standard errors (n = 4). Different letters indicate statistically significant differences, after a one-way (Analysis of Variance) ANOVA test with Tukey's pot-hoc test at $p \leq 0.05$).

3.2.3 Experiment equipment setup

The experiment was conducted in mesocosms made up of intact PVC pipe cores (200 mm x 500 mm). The PVC pipes were inserted into a 200 mm PVC coupler that was capped at one end (Figure 3.4A). The coupler was connected to a drainage pipe (blue pipe in Figure 3.4A, 3.4B, 3.4C), which was connected to a smaller PVC pipe (110 mm x 700 mm) that functioned as a dip well to maintain and observe the water table (Figure 3.4A, 3.4C). A filter made of a fine mesh was inserted onto the drainage pipe (Figure 3.4D) to stop the ingress of fauna through the dip well and to ensure that no soil washed away through the drainage pipe.

The mesocosms were buried into the ground leaving only the top 15 cm exposed (Figure 3.5A). Below ground (Figure 3.5B), the dip well was placed on top of loose gravel to allow the free draining of excess water from the soil cores. Figure 3.5C shows the cross-section of a single mesocosm setup buried in the soil. The dip well had a water table management system in place as shown in Figure 3.5C; this system comprised of a removable depth adjustment pipe. The bottom of the dip well was capped but water could flow out of the dip well through small holes on the depth adjustment pipe. The overflow of the depth adjustment pipe was through the middle of the cap at the bottom of the dip well. The small holes in the dip well were at a specific height which allowed the water table to be adjusted to any desired depth. The cores were buried in the soil to minimise temperature and moisture fluctuations caused by atmospheric conditions.

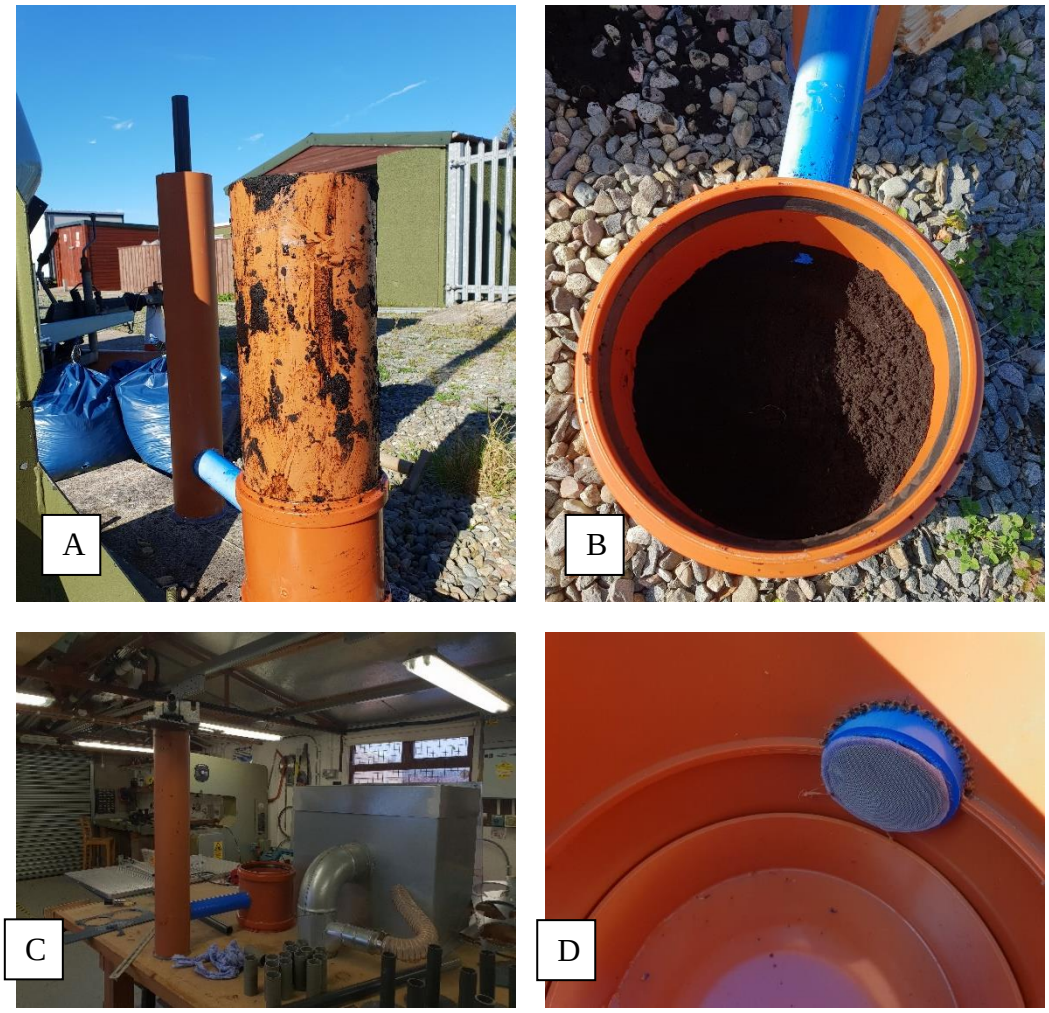


Figure 3.4 Setup of a single mesocosm (A) before it was buried into the ground. The soil core was inserted into the coupler, which was connected by a drainpipe (blue PVC pipe) to the dip well which was used to maintain and observe the water table (C). A fine mesh was placed on the drainage pipe (D) to stop the ingress of fauna into the cores through the dip well.

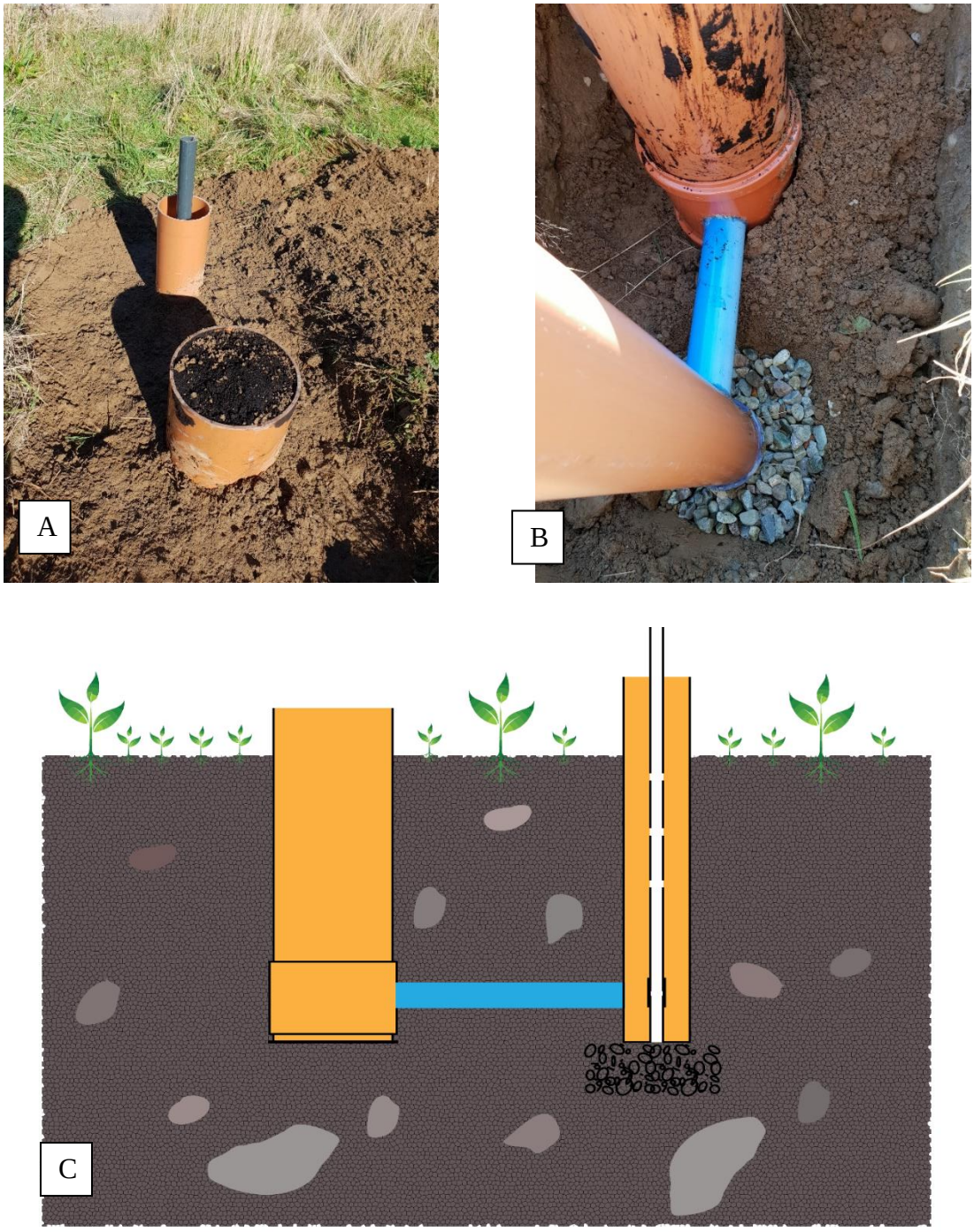


Figure 3.5 Cross-section of a single mesocosm buried in the soil, showing the soil core (left) and the dip well used for water table (WT) management (right). The soil core and the dip well were connected by a 40 mm drainage pipe.

3.2.4 Greenhouse gas (GHG) analysers

LI-COR autochambers (Long Term Chamber –Model 8100-104) were placed on top of the exposed 15 cm of the soil core (Figure 3.6). The autochambers were controlled by the LI-COR 16 port Multiplexer (LI 8150). The multiplexer only allows up to 16 autochambers to be connected. Therefore, the number of replicates in this experiment was limited to 16. When the autochambers were placed onto the collars, their offsets were calculated and input into the LI-COR LI8100A Infra-Red Gas Analyser (IRGA). The offsets were calculated following instructions from the LI-COR manual.

The LI-COR autochamber setup was connected to a Picarro Greenhouse Gas Concentration Analyser (G2508). The G2508 provides real-time measurements of N_2O, CH_4, CO_2, ammonia (NH_3), and water (H_2O). However, in this experiment, the G2508 was only used to collect fluxes of CH_4, N_2O, and CO_2. The G2508 is not capable of controlling the LI-COR autochambers, therefore, having this two-analyser setup was necessary.

The G2508 is easy to integrate with the LI-COR autochamber system. It was connected in series with the LI-COR autochambers, i.e., the G2508 was extracting gases from the exhaust of the LI-COR LI8100A using a 17.2 m gas line. A similar length exhaust gas line was connected to the G2508 which fed back to the LI-COR LI8100A exhaust. Due to the length of the gas line, there was a 30-second delay in flux responses on the G2508. This delay was accounted for when the LI8100 data was compared to the G2508 data. The sampling rate for both analysers was set at a rate of 1 Hz. This setup took measurements on each core, every hour, and every day for the whole duration of the experiment. At the time of the study, the analysers were both new, less than a year old, so calibration was unnecessary as they still had their factory calibration.

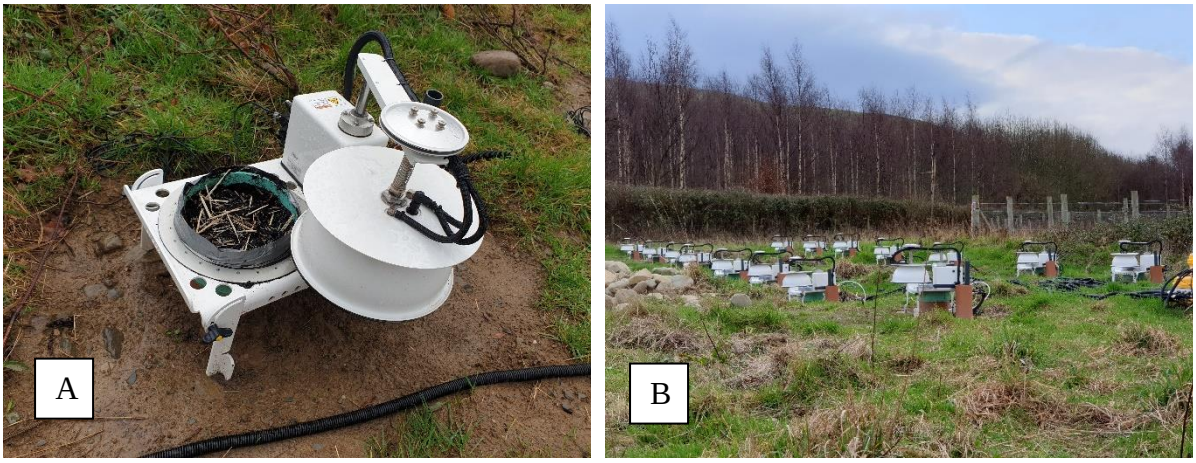


Figure 3.6 A LI-COR autochamber (Long Term Chamber –Model 8100-104) is shown here placed on top of the exposed 15 cm of the soil core (A). All 16 of the cores (B) had an autochamber fitted that was connected to the LI-COR 16 port Multiplexer (LI8150).

3.2.5 Experiment layout

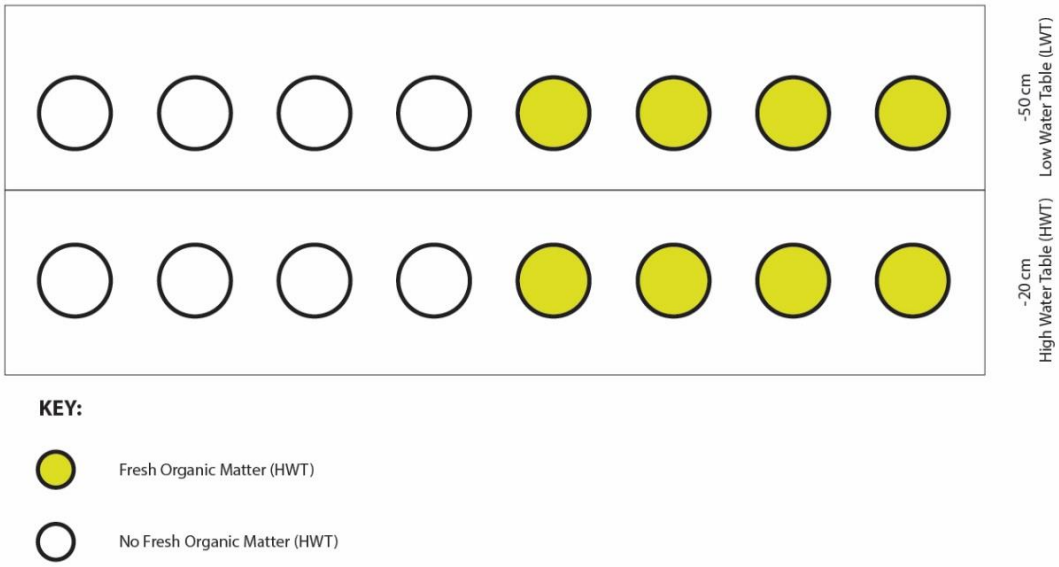


Figure 3.7 The layout of the mesocosm during the experiment. The cores were divided into two water tables, 50 cm from the surface and 20 cm from the surface.

3.2.6 Barley (*Hordeum vulgare* L.) straw as fresh organic matter (FOM)

Barley was chosen as a suitable FOM because it and wheat are common cereal crops grown on the peatlands in The Fens. Both barley and wheat belong to the *Poaceae* or *Gramineae* family of plants. However, unlike wheat straw, barley straw has antialgal properties. Studies have shown that in the presence of barley straw, algal biomass decreased whereas, in the presence of wheat straw, a catalytic effect was identified as the presence of the wheat straw led to algal biomass increase (Ball et al., 2001). Ball et al. (2001) found that decomposed barley straw inhibits the cyanobacteria *Microcystis spp.* and the algal species *Scenedesmus*. Whilst this study did not explicitly investigate the antialgal effects of barley straw, it is important to note as algal blooms are common on wetted peatlands. In this study, 10 g of shredded barley straw was used as the FOM amendment. This was applied to an area of 0.031 m^2, this amounts to 318 g of barley straw per m^2. According to Sedmihradská et al. (2020) barley has a similar C content to that of wheat, which is between 50% and 67%. Using the lowest estimate of 50%, the calculated amount of C applied per square metre is equal to 159 g m^{-2}.

3.2.7 Water table and soil temperature

3.2.7.1 Water table management

The experiment was split into two different water table management regimes. In the business-as-usual cores, the water table was maintained at 50 cm in line with what the farmers have reported as ideal, both for crop production and the use of heavy machinery (Renger et al., 2002). The maintenance of the water table was performed using the dip wells (shown in Figure 3.5C) to mimic the sub-irrigation method used at the farm where the soil samples were collected. The other half of the cores had an elevated water table depth of 20 cm. Similarly, the water table depth was maintained by topping up the dip wells when the level had gone below the 20 cm mark on the depth management pipe (Figure 3.5C). The water table depth was checked every two days to make sure they had not fallen below the required depths.

3.2.7.2 Soil temperature monitoring

Throughout the experiment, the soil temperature was recorded using LI-COR 8150-203 Soil Temperature Probes that were inserted into each core to a depth of 10 cm. The temperature probes were permanently inserted into the cores; therefore, they produced continuous readings

of the soil temperature throughout the experiment. The temperature readings were recorded at the same time as the fluxes on the IRGA.

3.2.8 Experiment timeline

The experiment was conducted from the 9th of November 2018 until the 10th of May 2019. During the first week, baseline GHG measurements were recorded before the water tables were raised. GHG fluxes were measured in response to the water table alone for four weeks. FOM was added on the 16th of December 2018 and GHG flux measurements were recorded until the end of the experiment.

3.2.9 Methane (CH_4) consumption scaling

The scaling up of CH_4 consumption was achieved using the average CH_4 flux in Table 3.2. The calculation was made by first converting the average flux from hourly to yearly fluxes. The value was then scaled up from m^2 to the estimated area of cultivated peatland of 250 000 km^2. This value was then converted from µg to g.

3.2.10 Global warming potential (GWP) and carbon dioxide (CO_2) equivalent (CO_2eq) of nitrous oxide (N_2O) and methane (CH_4)

The CO_2 equivalent (CO_2eq) GHG fluxes were calculated using the GWP 100-year timeline which was adopted by the Intergovernmental Panel on Climate Change (IPCC). The GWP for CH_4 used in the calculations was 28 and for N_2O it was 273 as reported by the IPCC (IPCC, 2021). The calculations were calculated using the equations below ((1) CO_2, (2) CH_4) and (3) N_2O) adapted from Naser et al. (2005):

(CO_2eq) GHG flux CO_2 (g CO_2 m^{-2} hr^{-1}) = CO_2 (g C m^{-2} hr^{-1}) x (1 g CO_2) x (44 g CO_2/12 g C)

(1)

(CO_2eq) GHG flux CH_4 (g CO_2 m^{-2} hr^{-1}) = CH_4 (g C m^{-2} hr^{-1}) x (28 g CO_2/1g CH_4) x (16 g CH_4/12 g C)

(2)

(CO_2eq) GHG flux N_2O (g CO_2 m^{-2} hr^{-1}) = N_2O (g N m^{-2} hr^{-1}) x (273 g CO_2/1 g N_2O) x (44 g CO_2/28 g N)

(3)

3.2.11 Statistical analysis

All figures that accompany these results were produced using R. Statistical calculations were made using RStudio Version 1.2.5033 with R version 3.6.3 (2020-02-29). Significance was accepted at $p \leq 0.05$. Normality tests were performed on the data using the *ggpubr* package and the moments package. These packages showed that the data was positively skewed. Therefore, transformation was necessary. The data was log-transformed, after transformation, the data was tested for normality again and it fitted the normality test.

3.2.11.1 Carbon Dioxide (CO_2) fluxes from the LI-COR LI8100

As the data in this study was measured repeatedly on the same cores over time, a Repeated Measures ANOVA (Analysis of Variance) approach was most appropriate. A mixed-effects model was chosen because the same cores were measured continuously throughout the experiment therefore they would have been a problem with pseudoreplication. Pseudoreplication is a problem because repeated measures on the same core will mean there is no independence of errors which is an important assumption of standard statistical analysis (Crawley, 2013).

3.2.11.2 Carbon Dioxide (CO_2) fluxes from the Picarro G2508

For comparative purposes, the data from the Picarro G2508 was also subjected to the same approach as the IRGA data. A similar two-way ANOVA method was used to calculate the statistical significance of the observed CO_2 fluxes from the G2508 GHG analyser.

3.2.11.3 Carbon dioxide (CO_2) Cumulative fluxes

In addition to flux calculations being analysed using averages, the flux cumulative totals were also calculated, the calculations were completed in R using the cumsum function. With the cumsum function, the fluxes were added cumulatively from the beginning to the end of the study.

3.2.11.4 Methane (CH_4) fluxes from the Picarro G2508

CH_4 fluxes were analysed using a similar two-way ANOVA as that used for CO_2 fluxes from both the IRGA and the G2508.

3.2.11.5 Nitrous Oxide (N_2O) fluxes from the Picarro G2508

The N_2O fluxes were calculated using a similar two-way ANOVA method used for the fluxes of CO_2 and CH_4.

3.3 Results

3.3.1 Soil temperature

The weekly soil temperature averages are shown in Figure 3.8 for each treatment. The mean temperatures remained above freezing during the experiment. No statistical analyses were conducted on the soil temperatures.

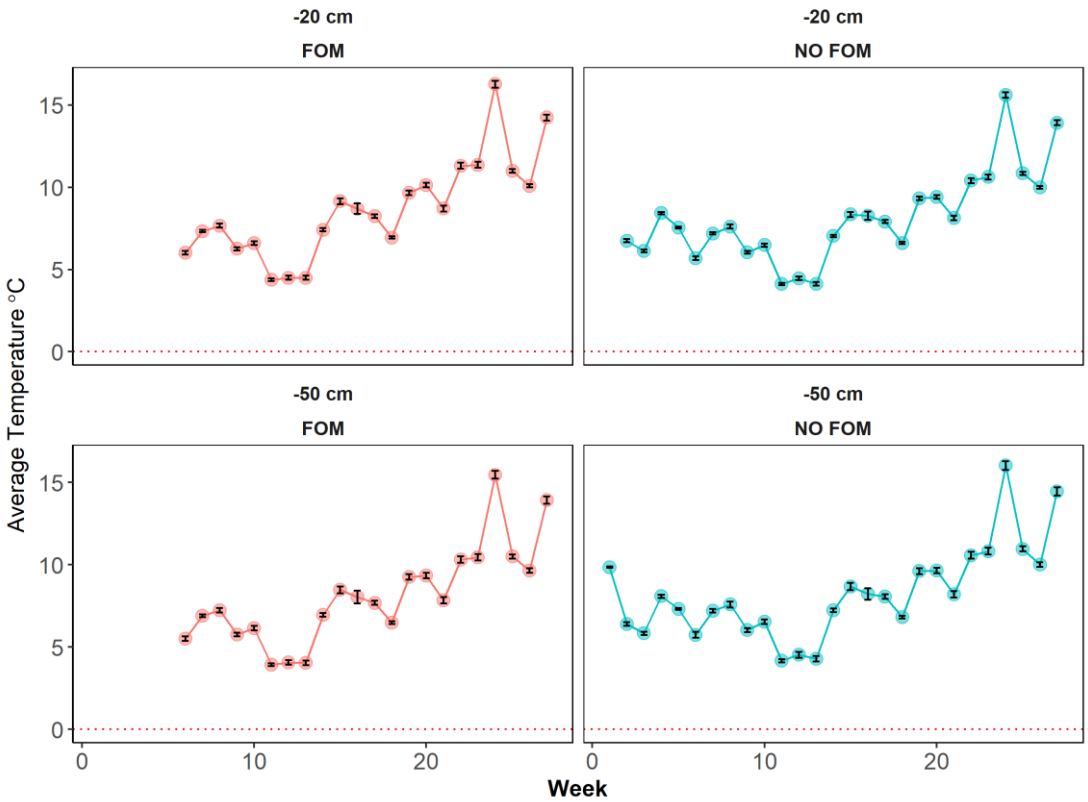


Figure 3.8 Average weekly soil temperatures in the cores throughout the experiment from week 0 in November 2018 to week 27 in May 2019. All values represent means ± SEM (n = 4).

3.3.2 Greenhouse gas (GHG) fluxes

Table 3.2 Average fluxes of carbon dioxide (CO_2) collected using the LI-COR LI8100 Infra-Red Gas Analyser (IRGA) for the duration of the experiment over 27 weeks and average methane (CH_4) and nitrous oxide (N_2O) fluxes measured by the Picarro G2508. All values represent means ± SEM (n = 4).

Water table (cm)	Treatment	Mean CO_2 (g CO_2 m^{-2} hr^{-1})	CO_2 SE	Mean CH_4 (µg CH_4 m^{-2} hr^{-1})	CH_4 SE	Mean N_2O (µg N_2O-N m^{-2} hr^{-1})	N_2O SE
20 cm	FOM	0.104	0.06	-0.97	0.18	40.16	1.88
50 cm	FOM	0.084	0.06	1.59	0.32	39.04	1.61
20 cm	NO FOM	0.138	0.06	-3.17	0.30	34.08	1.63
50 cm	NO FOM	0.098	0.05	0.33	0.33	39.10	1.39

3.3.2.1 Carbon dioxide (CO_2) fluxes

3.3.2.1.1 Average carbon dioxide (CO_2) fluxes

The average CO_2 fluxes which were collected using the IRGA, Table 3.2, show that there were significant differences in the fluxes of the cores either with or without FOM ($p < 0.001$) in both water table treatments ($p < 0.001$). The CO_2 fluxes in the 20 cm water table were significantly lower than in the 50 cm water table ($p < 0.001$). Overall, the cores with FOM had higher CO_2 flux averages ($p < 0.001$).

The CO_2 fluxes responded positively to increasing soil temperature changes as shown in Figure 3.9. As the soil temperatures increased, so did the associated CO_2 fluxes. Figure 3.10 breaks down the observed CO_2 fluxes on a week-by-week basis. The fluxes were much lower at the beginning of the experiment when the temperatures were lower in November, but they increased with higher temperatures from about week 20 to week 27. In Figure 3.10, the effect of FOM is similarly evident. A linear regression test on the effect of temperature on the CO2 fluxes showed that there is a significant relationship ($p < 0.001$), there was a 0.011 unit increase of CO2 with every unit of temperature increase.

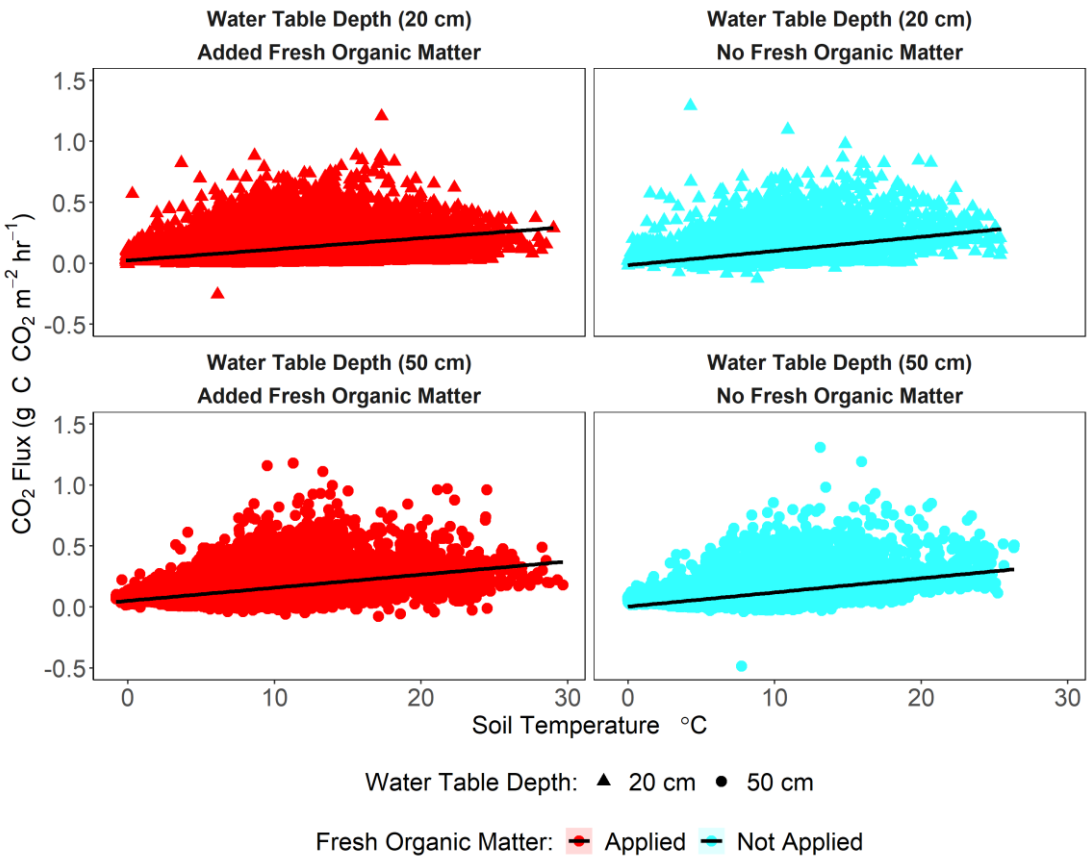


Figure 3.9 Carbon dioxide (CO_2) fluxes during the lowland peat study increased with higher soil temperature. The CO_2 fluxes went up as the soil temperature increased (n = 4).

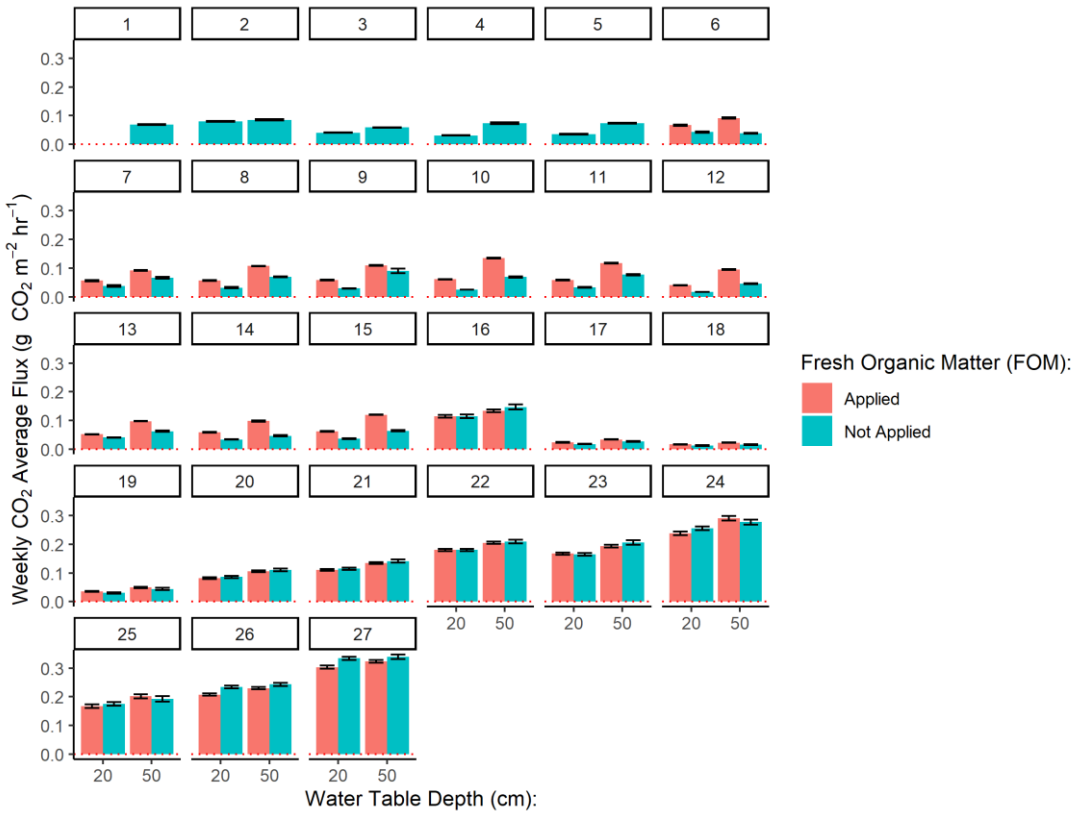


Figure 3.10 Lowland peatland study average carbon dioxide (CO_2) fluxes per week over the 27-week duration of the experiment. In week 1, all the cores had a 50 cm water table for the baseline values. The water table was altered in half of the cores from week 2. In week 6, fresh organic matter (FOM), in the form of barley (Hordeum vulgare L.) straw, was added to half of the 20 cm water table cores and half of the 50 cm water table cores. Values represent means ± SEM (n = 4) except for week 1 (n = 16) and week 2 to 5 (n = 8).

3.3.2.1.2 Cumulative carbon dioxide (CO_2) totals

The cumulative total CO_2 fluxes (Figure 3.11) from the cores in the 20 cm water table cores with added FOM was 15.5 ± 0.85 s.e. g CO_2 m^{-2}, while the fluxes in the cores without FOM was 15.8 ± 1 s.e. g CO_2 m^{-2}. Statistical analysis comparing the FOM versus no FOM treatments in the 20 cm water table showed that the cumulative CO_2 flux differences were not statistically significant (p = 0.52). Conversely, the differences between the FOM and no FOM in the 50 cm water table treatment were statistically significant (p < 0.001). The FOM treatment had a cumulative total of 20.5 ± 0.9 s.e. g CO_2 m^{-2} and the treatment without FOM had a cumulative total of 11.5 ± 1.6 s.e. g CO_2 m^{-2}. The effect of the water table was significant, p < 0.001

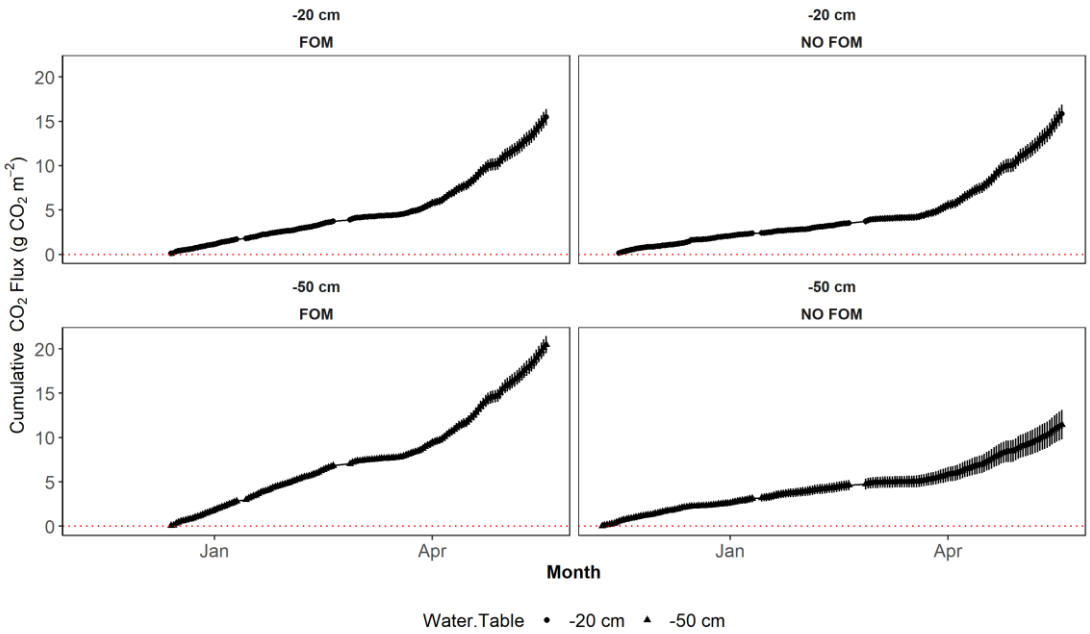


Figure 3.11 Daily cumulative carbon dioxide (CO_2) fluxes from the LI-COR IRGA for the 27-week duration of the study. The cumulative values were calculated from daily means of the cores ± SEM (n = 4).

3.3.2.2 Methane (CH_4) fluxes

3.3.2.2.1 Average methane (CH_4) fluxes

All the CH_4 fluxes were recorded using the Picarro G2508 gas analyser as the LI-COR LI8100 IRGA can only record CO_2 fluxes. The results summarised in Table 3.2, show that there is a significant effect of both water table manipulation and the addition of FOM. The Repeated Measures ANOVA on the effect of the water table was significant $p < 0.001$ and the effect of FOM, $p < 0.001$. The interaction of elevated water table and FOM was significant, $p < 0.001$. The addition of FOM led to the cores being sinks of CH_4 in both the 20 cm and 50 cm water table treatments. However, the sink effect induced by FOM was more pronounced in the 50 cm cores. The cores without FOM were sources of CH_4, but the 20 cm cores produced significantly more CH_4 than the 50 cm cores. Figure 3.12 shows that unlike CO_2 emissions that increased with rising temperature, the CH_4 fluxes showed a downward trend to increased soil temperature. A linear regression test confirmed this, $p < 0.001$, there was a -0.00000045 unit decreased with every unit of temperature increase. In week 3, there was an observed spike of fluxes in the cores without FOM, but this spike shows no obvious correlation to any other measurement. Figure 3.13 summarises the average fluxes on a week-by-week basis. Before the water table was manipulated, the cores were consuming atmospheric CH_4 but after the alteration of the water table, there was a spike in CH_4 production which lasted for 2 weeks before the cores returned to being consumers of CH_4.

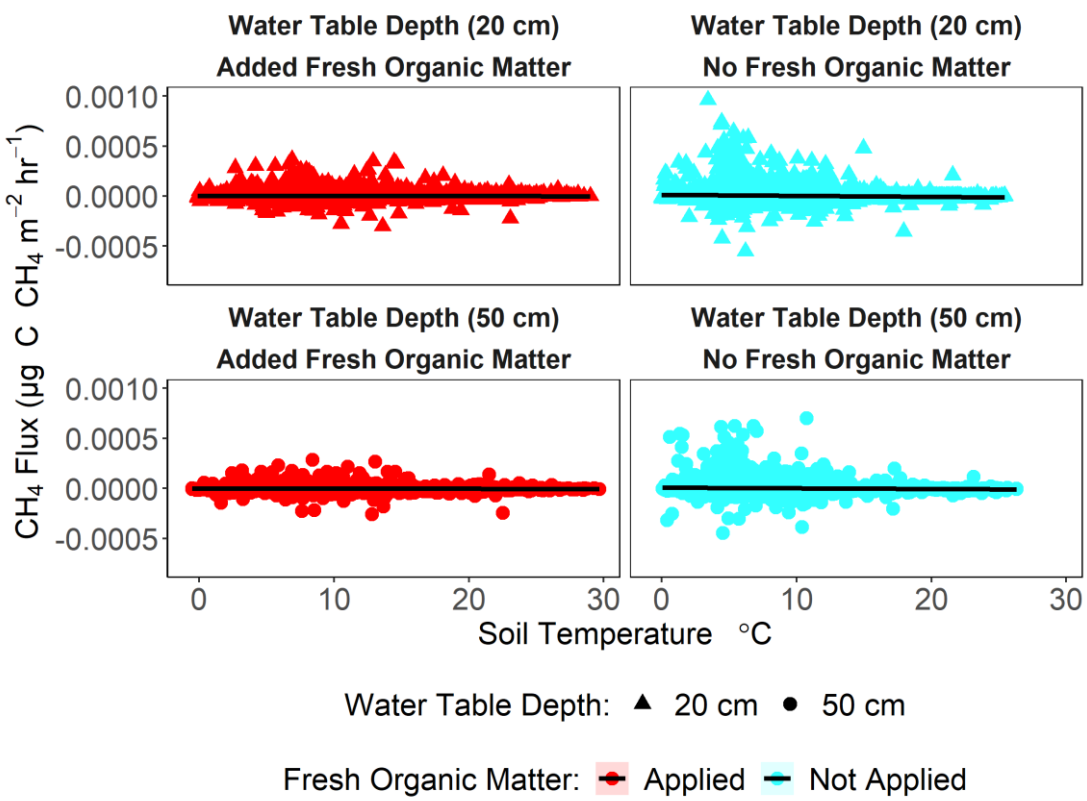


Figure 3.12 During the lowland peat study to assess the effectiveness of fresh organic matter (FOM) and water table depth, the methane (CH_4) fluxes did not show a discernible response to the soil temperature (n = 4).

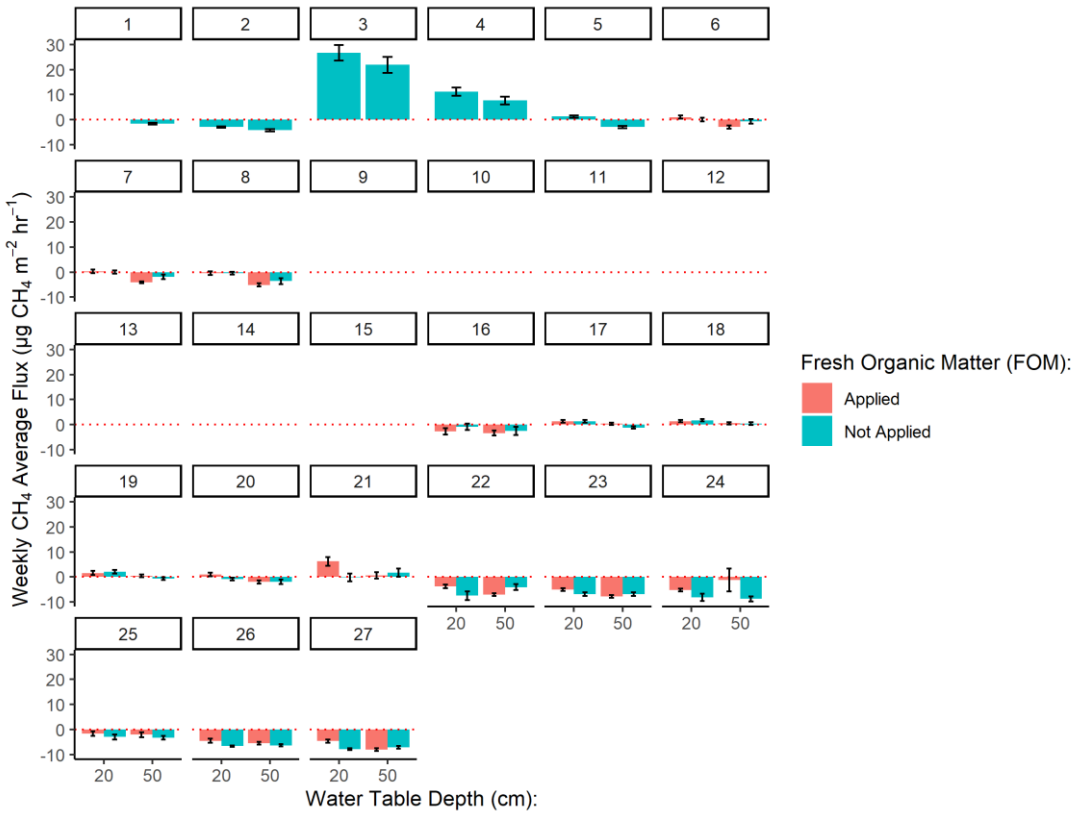


Figure 3.13 Weekly average methane (CH_4) fluxes from a lowland peatland agricultural soil in response to different water table depths (20 and 50 cm) and the presence or absence of barley (Hordeum vulgare L.) crop residue over 3 months. Values represent means ± SEM (n = 4) except for week 1 (n = 16) and week 2 to 5 (n = 8).

3.3.2.2.2 Methane (CH_4) cumulative totals

The FOM treatment in the 20 cm water table cores (Figure 3.14), consumed -106 ± 191 s.e. µg CH_4 m^{-2} compared to -16 ± 280 s.e. µg CH_4 m^{-2} from the cores without FOM. This effect was statistically significant, $p < 0.001$. The 50 cm cores with FOM treatment consumed -332 ± 200 s.e. µg CH_4 m^{-2}, but the 50 cm cores without FOM released a cumulative total of 91 ± 298 s.e. µg CH_4 m^{-2}. Overall, the effect of the water table was statistically significant, $p < 0.001$.

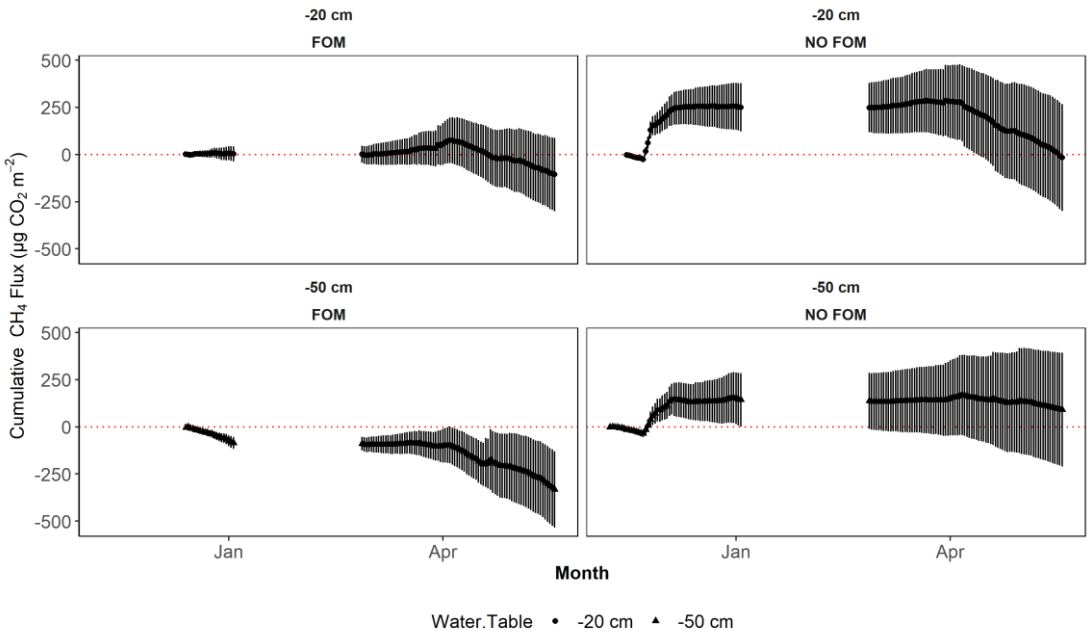


Figure 3.14 The methane (CH_4) cumulative fluxes for the 27-week duration of the experiment. The cumulative values were calculated from the daily means of the cores ± SEM (n = 4).

3.3.2.3 Nitrous Oxide (N_2O) fluxes

3.3.2.3.1 Average nitrous oxide (N_2O) fluxes

The data collected on N_2O fluxes in Table 3.2 shows that fluxes were significantly different between the water tables ($p < 0.001$). Within each water table treatment, the effect of FOM was significant ($p < 0.001$). However, the differences in the fluxes of the FOM treatments was not statistically significant between the water tables ($p = 0.17$) Within the 20 cm water depth cores, the difference between the cores treated with FOM against those without FOM was not as pronounced as the difference observed in the cores with 50 cm water table. Like the fluxes of CH_4, the N_2O fluxes do not seem to follow a trend or respond to soil temperature fluctuation (Figure 3.15 and Figure 3.16).

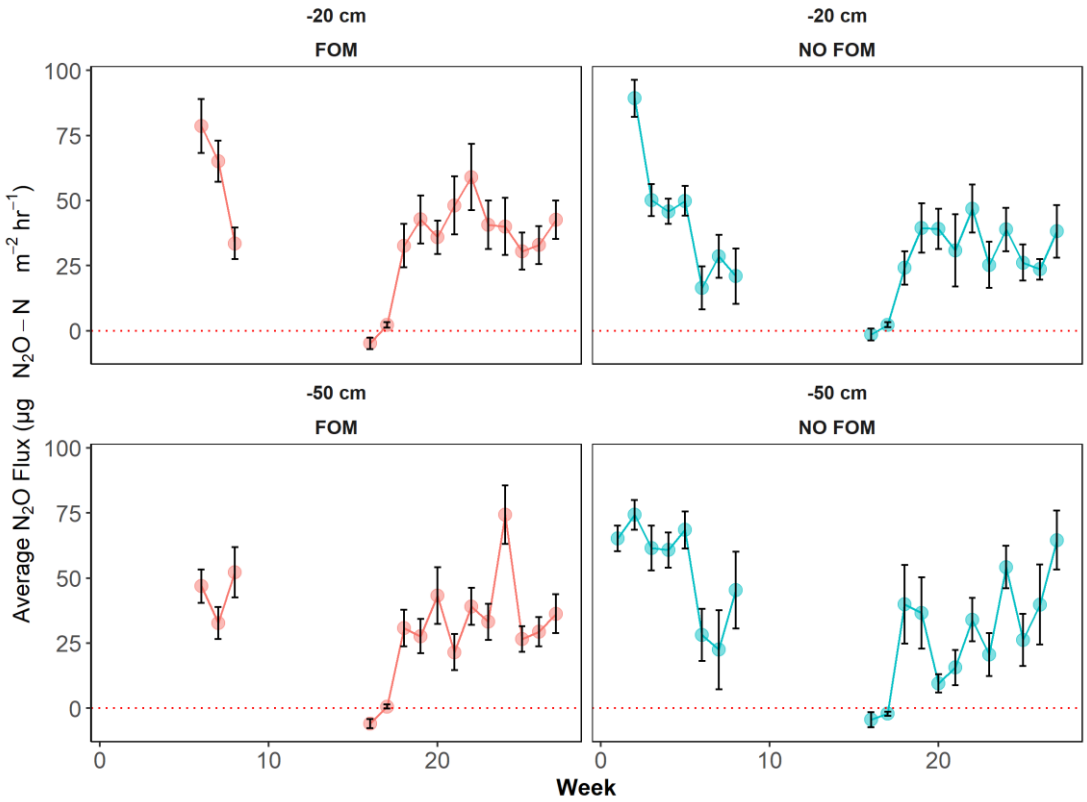


Figure 3.15 The weekly averages of the nitrous oxide (N_2O) fluxes observed on the Picarro G2508 gas analyser. All values represent the means of the cores ± SEM (n = 4).

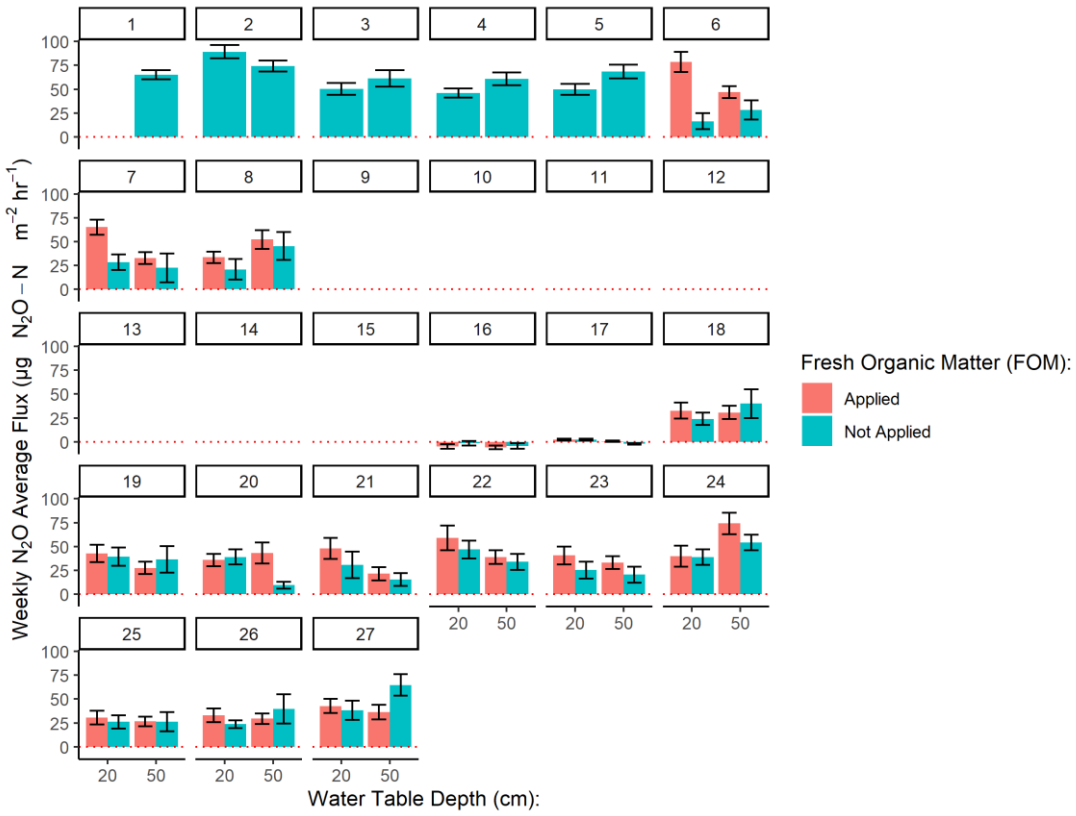


Figure 3.16 The weekly averages of the nitrous oxide (N_2O) fluxes throughout the experiment. In week 1, all the cores had a 50 cm water table for the baseline values. The water table was altered in half of the cores from week 2. In week 6, fresh organic matter (FOM), in the form of barley (Hordeum vulgare L.) straw, was added to half of the 20 cm water table cores and half of the 50 cm water table cores. Values represent means ± SEM (n = 4) except for week 1 (n = 16) and week 2 to 5 (n = 8).

3.3.2.3.2 Cumulative nitrous oxide (N_2O) totals

The statistical analyses of the N_2O cumulative fluxes (Figure 3.17) show that the effect of FOM both in the 20 cm and 50 cm cores was significant, $p < 0.001$. The 20 cm FOM treatment produced 4133 ± 1964 s.e. µg N_2O-N m^{-2}, whilst the treatment without FOM produced a cumulative total of 4562 ± 2252 s.e. µg N_2O-N m^{-2}. In the 50 cm water table cores, the FOM treatment produced 3476 ± 1633 s.e. µg N_2O-N m^{-2}, whilst the treatment without FOM produced 4750 ± 2120 s.e. µg N_2O-N m^{-2}. The effect of the water table was likewise statistically significant $p < 0.001$.

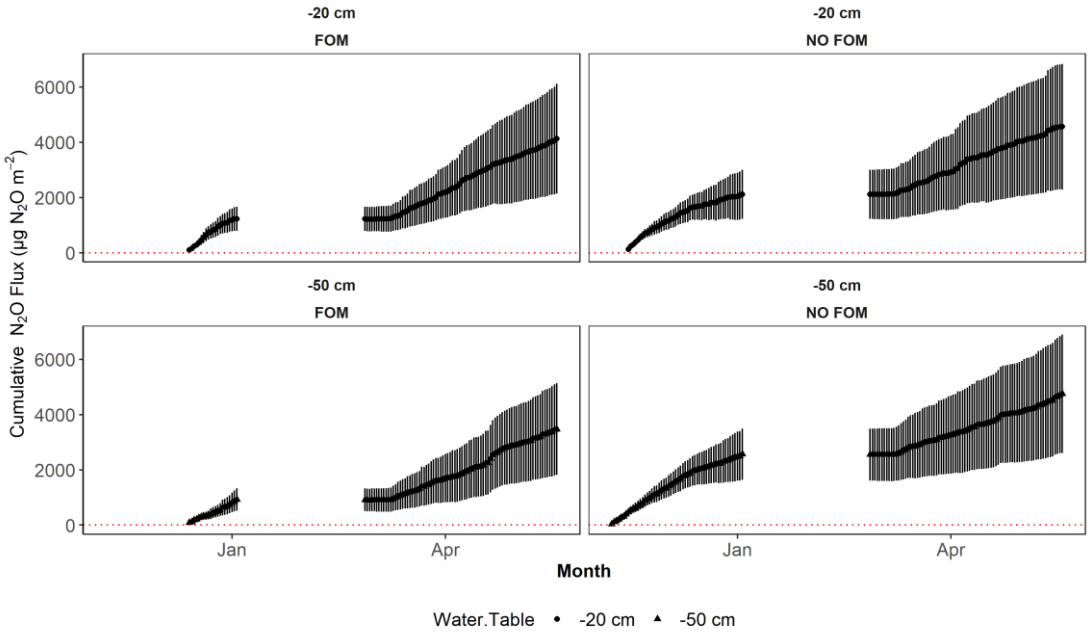


Figure 3.17 The nitrous oxide (N_2O) cumulative fluxes for the 27-week duration of the experiment. The cumulative N_2O fluxes increased during the study. The missing data is due to the failure of the Picarro G2508 for 6 weeks of the study. The cumulative values were calculated from daily means of the cores ± SEM (n = 4).

3.3.3 Carbon (C) losses in the soil cores

3.3.3.1 Average carbon (C) loss

Atmospheric loss of carbon from the cores was either as CO_2 or CH_4. Cumulative CO_2 represented the largest loss of carbon from the system, over 99.9 % of total carbon. This loss was more pronounced in the cores that had a water table of 50 cm. Over the 27-week duration of the experiment, the 20 cm cores without FOM produced an average of 103 g C m^{-2}. The 20 cm cores with FOM produced a higher average amount of carbon of 129 g C m^{-2} over the same 27 weeks, which is a difference of over 25 g C m^{-2}. In the 20 cm water table treatment, the amount of C lost as CO_2 was equivalent to 16% of the C added as FOM.

In the low water table cores of 50 cm, the cores with added FOM produced an average of 171 g C m^{-2}, while the cores without added FOM produced an average of 121 g C m^{-2}. The difference between the 20 cm and 50 cm cores with added FOM was an average of 42 g C m^{-2}, while the difference between the 20 cm and 50 cm cores without added FOM was an average of 17 g C m^{-2}. In the 50 cm water table, an average of 50 g C m^{-2} was lost as CO_2 which is equivalent to 31.6% of the added FOM carbon. Statistical analyses on the means used to calculate the carbon loss, Table 3.2, show that these differences were statistically significant, $p < 0.001$.

The total amount of carbon lost in the form of CH_4 was insignificant compared to CO_2; there was C loss only in the cores that did not have added FOM. The 20 cm (NO-FOM) cores lost on average 0.002 g C m^{-2} over 27 weeks, which was about 0.00019% and the 50 cm (NO-FOM) cores lost an average of 0.0004 g C m^{-2}, which is 0.0012% of the total carbon lost into the atmosphere. The cores that had added FOM under both water table treatments had negative fluxes, meaning the soil was consuming C as CH_4. In the 20 cm cores, this amounted to an average of 0.0012 g C m^{-2} gain over 27 weeks, and in the 50 cm cores, the gain was marginally higher at an average of 0.0039 mg C m^{-2}. The difference between both water table cores with added FOM was 0.0019 g C m^{-2} and the difference between both water table cores without FOM was 0.0004 mg C m^{-2}.

3.3.4 Global warming potential (GWP) and carbon dioxide equivalent (CO_2eq) of nitrous oxide (N_2O) and methane (CH_4)

Table 3.3 Carbon dioxide (CO_2) equivalent (CO_2eq) for the fluxes of CO_2, methane (CH_4), and nitrous oxide (N_2O). The cumulative values were calculated from daily means of the cores ± SEM (n = 4).

Water Table (cm)	FOM	Cumulative CO_2 g CO_2 m^{-2}	CO_2eq (CO_2) g CO_2eq m^{-2}	Cumulative CH_4 g CH_4 m^{-2}	CO_2eq (CH_4) g CO_2eq m^{-2}	Cumulative N_2O g N_2O-N m^{-2}	CO_2eq (N_2O) g CO_2eq m^{-2}
20	Yes	15.5	56.7	-0.11	-0.004	4.1	1.8
20	No	15.8	58.0	-0.02	-0.001	4.6	2.0
50	Yes	20.5	75.1	-0.33	-0.012	3.5	1.5
50	No	11.5	42.0	0.09	0.003	4.7	2.0

3.4 Discussion

3.4.1 Effects of barley (*Hordeum vulgare* L.) residue and water table on greenhouse gas (GHG) fluxes

The addition of FOM to the cultivated peat soil played a key role in all the observed GHG fluxes during the 27-week study. All the gases responded to the addition of FOM in a non-uniform way to each other. Below the response of all the GHGs measured is discussed in detail.

3.4.1.1 Carbon dioxide (CO_2) fluxes

An increase in the flux of CO_2 was observed in cores that had FOM added compared to the cores that did not have any FOM added, especially in the cores that were under the 50 cm water table treatment. An increase in the amount produced when FOM is added was not unexpected as previous studies such as Taft et al. (2018) showed comparable results where the application of FOM led to a marked increase in soil respiration. Furthermore, Bader et al. (2018), Nishigaki et al. (2021) and Wang et al. 2015) have shown that the addition of FOM can potentially lead to a PE. Adding FOM was expected to lead to increased CO_2 fluxes as the FOM is readily available to soil microbes. However, the water table played a vital role in mitigating the resulting fluxes from the added FOM. The cores with the 20 cm water table produced less CO_2 than the cores in the 50 cm water table treatment. A closer look at the cumulative CO_2 fluxes shows that the cores in the 20 cm water table treatment with FOM added produced less CO_2 compared to the cores that had no FOM added. This result emphasises the importance of water table depth in the mitigation of GHG emissions from soil.

Whilst the addition of FOM led to the increased emissions of CO_2, these emissions could be negligible given the potential benefits of FOM. After 27 weeks of experimental work, the losses of C were calculated. From the mass balance analysis, the majority of added FOM remained in the soil for both water table treatments. Whilst the mass balance showed that FOM addition was beneficial in this study, its PE should not be neglected as this could lead to the decomposition of existing recalcitrant C thus leading to CO_2 loss. It is not possible to say with certainty that the CO_2 produced after the addition of FOM was due to the accelerated decomposition of peat or the FOM decomposition. However, as the addition of water slowed down the decomposition rate, the use of FOM could be beneficial in the long term as it will increase the amount of SOC in the soil which will prolong the usability of the soil for horticultural purposes. Nevertheless, given the length of this study, it is uncertain whether these

benefits will persist longer. Studies in mineral soils have shown varied results for the addition of FOM. Garcia-Franco et al. (2015), demonstrated that adding crop residue by tilling it into the soil increased the organic carbon (OC) concentration by 30% in the tilled layer (0-15 cm). However, these benefits were not observed by Piotrowska & Wilczewski (2012) who concluded in their long-term study, that the treatment of catch crop residue did not influence OC. How FOM is applied is just as important as FOM itself. Taft et al. (2018) demonstrated that not tilling in the FOM can lead to more flux than when the FOM is tilled into the soil. Therefore, it is important to ensure that any FOM, whether imported or from a previous cover crop is ploughed into the soil, otherwise, its benefits are lost.

3.4.1.2 Methane (CH_4) fluxes

The addition of FOM led to the consumption of CH_4 in both 20 cm and 50 cm water table treatments. The consumption was more pronounced in the cores that were under the 50 cm water table depth (Figure 3.14). However, when the amount of CH_4 being consumed is scaled up to a global scale, the benefits are minuscule compared to the damage caused by drained peatland cultivation. On average, the 20 cm water table depth consumed 0.97 µg CH_4 m^{-2} hr^{-1} while the 50 cm water table cores consumed 3.17 µg CH_4 m^{-2} hr^{-1}. Using the estimated amount of peatland under drained agriculture of 250 000 km^2 as reported by Evans et al. (2021) and the average fluxes in this study, Table 3.2, the scaled-up amount of CH_4 consumed by the 20 cm water table would be an average of 2124 g CH_4 m^{-2} yr^{-1} and for the 50 cm water table this would be 6942 g CH_4 m^{-2} yr^{-1}. This is important as it shows that while there is potential to sequester C through the consumption of CH_4, this is not enough to offset the large amount of C being released by the soil as CO_2 is estimated to be 7600 kt CO_2e yr^{-1} (Evans et al., 2017).

The CH_4 fluxes responded to the water table manipulation. The cores that had a water table of 20 cm from the surface produced more CH_4 than the cores with a water table of 50 cm from the surface. An increase in CH_4 was expected as the anoxic conditions that favour methanotrophs were brought closer to the surface and would also increase an abundance of methanogens (Turetsky et al., 2014). However, the water table alone is not the only influence of CH_4 flux, the presence of vegetation plays a key role in the fluxes of CH_4 (Evans et al., 2021; Cooper et al., 2014; Henneberg et al., 2016). Plant aerenchyma can aid the transport of CH_4 from deeper anaerobic zones of soil and they can provide a substrate for methanogenesis. Henneberg et al. (2016) found that CH_4 emission rates are lower in soil without vegetation than in soil that was planted with *Juncus effusus* L., and Cooper et al. (2014) reported similar

findings with hare's-tail cottongrass (*Eriophorum vaginatum* L.). Nevertheless, the soil in this study did not have any plants thus it is not an effect that this study would have observed. So, our CH_4 fluxes may be lower than they could have been if plants were growing in the cores.

Although there was more CH_4 produced in the 20 cm cores (see Table 3.2), the 50 cm cores also remained a source of CH_4. Whilst this is a small amount, lower than that produced by natural wetlands, it does show that peatlands using a water table of 50 cm contribute to the global CH_4 emissions.

3.4.1.3 Nitrous oxide (N_2O) fluxes

The addition of FOM to the soils led to a reduction of the N_2O fluxes in both the 20 cm and 50 cm cores. However, in the 20 cm water table cores, the difference of the average and cumulative N_2O fluxes was not as pronounced as for the 50 cm water table treatment. The statistical analysis confirmed that the addition of FOM did not have any significant effect on the fluxes of N_2O in the 20 cm water table cores. However, in the 50 cm water table treatment, the addition of FOM was statistically significant. The 50 cm cores with FOM produced a cumulative total of 3476 ± 1633 s.e. µg N_2O-N m^{-2} while the no FOM treatment cores produced 4750 ± 2120 s.e. µg N_2O-N m^{-2}. The 20 cm water table with FOM treatment in this study produced significantly more N_2O at 4133 ± 1964 s.e. µg N_2O-N m^{-2} compared to 4133 ± 1964 s.e. µg N_2O-N m^{-2} produced by 50 cm water table cores. Wen et al. (2019) showed comparable results where their higher water table treatments with FOM produced four times more N_2O flux. The magnitude of the fluxes they observed could be due to the high-quality N rich FOM used, nevertheless, a water table close to the surface can increase the amount of N_2O flux. The increased N_2O fluxes were attributed to increased denitrification. Under the elevated water table, it is expected that this was due to anaerobic denitrification. In this study, it is expected that similar processes to those observed in Wen et al. (2019) were responsible for the increased N_2O fluxes in the 20 cm water table cores.

Interestingly, the N_2O flux in the 50 cm water table cores with FOM was not just lower than the similar treatment in the 20 cm water table but it was significantly lower than the treatment without FOM in the 50 cm water table cores. The reason reduced N_2O flux was observed in the dry and aerobic soils could be owing to the heterotrophic denitrifying bacteria not going through the denitrification process because of the presence of O_2. Most denitrifiers undertake denitrification only when O_2 is unavailable because nitrate is a less efficient electron acceptor

than O_2 (Robertson & Groffman, 2015; Morley & Baggs, 2010; Russow et al., 2009). However, whilst it has been shown that there is a regulation of the consumption of N_2O in dry and aerobic soil by soil moisture content, it is evident that denitrification can still occur in aerobic soil hence our observation of N_2O fluxes in the low water table cores albeit smaller than the elevated cores (Wu et al., 2013).

3.4.2 Benefits of fresh organic matter (FOM) addition

This study was short at only 27 weeks. However, it does point to the importance of FOM to the global fight against climate change. In the limited time the study was conducted, the majority of the added FOM remained in the soil. The benefits of FOM are further amplified using elevated water tables, especially with regards to the fluxes of CO_2. However, as this study used dried barley straw as FOM, it is not clear if all kinds of FOM or fresh FOM will have the same effect. Future studies should be conducted using other types of crop residue to find out if there are better crops and which are worse. Whilst the majority of added FOM remained in the soil, it cannot be stated conclusively whether the FOM had a PE or not.

3.4.3 Use of larger and longer cores

In line with studies such as Matysek et al. (2019), Musarika et al. (2017) and Wen et al. (2019), this study was conducted using cores of 60 cm length. However, the cores were wider than those used in the aforementioned studies, with a diameter of 20 cm which allowed for more of the soil area to be sampled. A height of more than 60 cm would have been ideal as there could be more variation of GHG fluxes due to increased depth as deeper cores include a larger anoxic area and other variables, such as undecomposed wood, which is common at deeper depths on the cultivated peatlands where the soil samples were collected. However, coring to a depth greater than 60 cm provides challenges, especially in relation to soil compaction due to the friction of the PVC pipes which makes the cores unreliable representatives of the field conditions. Furthermore, at greater depths, chances of encountering bogwood and large rocks are high, and this often leads to broken cores. To maintain the reliability of the results, this study used 60 cm cores. A future study could develop methods that can allow for coring to greater depths. Nevertheless, the depth of the cores was more than sufficient as it allowed for a considerable proportion of the peat to remain undisturbed when the FOM was added to the top 30 cm of the core. Furthermore, regardless of core depth, the same depth of oxic layer will remain..

3.4.4 More replicates versus high-frequency data trade-off

This experiment was conducted using only 16 cores as that is the maximum the LI-COR 16 port Multiplexer and LI-COR autochamber setup can take. Whilst this limits the number of replicates, it is a better setup with higher resolution than manual measurements taken less frequently. Having only 16 cores is a justifiable trade-off as the data is of the highest temporal resolution possible allowing for accurate estimations of the fluxes. Understanding exactly when the water table depths begin to have an effect could better inform which water table is best to use. In a future experiment, ideally using a second set of LI-COR autochambers setup, it would be ideal to have a variety of water table depths e.g., 10 cm, 20 cm, 30 cm. Having these water table depths would enable a greater understanding of the effects of these varying water table depths and how each depth responds to different treatments.

3.4.5 Conclusions

The addition of FOM to cultivated peatlands has produced a double-pronged result. The addition of barley straw as a source of FOM has potential benefits when it comes to the emissions of CH_4 and N_2O where consumption was observed. The cores with FOM in the 20 cm water table had higher N_2O fluxes compared to the 50 cm. However, these fluxes are not sufficient to disregard the benefits provided by the addition of FOM. The addition of FOM in 50 cm has benefits through the reduction of N_2O fluxes but this is still not enough given the associated CO_2 fluxes caused by the oxidation of peat when the anaerobic conditions are removed.

While the addition of FOM showed an increase in the fluxes of CO_2, there was no evidence of a PE as substantial amounts of the added FOM remained in the soil. At the very least, adding FOM is beneficial to cultivated peatlands and has the potential to increase the C stock in the soil to provide some damage limitation. However, this study has shown it will not be possible for FOM addition to offset the enormous amount of CO_2 emissions from drainage-based peat cultivation.

Chapter 4. Long-term impact of fresh organic matter addition on the emissions of carbon dioxide from a horticultural peat

Abstract

As a result of the current population growth trend, more land is going to be required for food production to meet food demands. Consequently, there will be a growing need for productive arable land. This puts peatlands under immense pressure as they are productive soils for arable agriculture. However, cultivated peatlands are a source of anthropogenic CO_2. Restoring peatlands to their natural state can turn them from sources of carbon (C) to sinks but that will render them unsuitable for food production; doing so will also have an economic impact. For example, the total agricultural production output of the Fens is worth £1.23 billion per year. Consequently, taking peatlands completely out of productive use is unlikely to be feasible in practice. Therefore, there is a need to develop mitigation measures that can allow for the reduction of CO_2 fluxes while at the same time maintaining food production. One mitigation option is the addition of fresh organic matter (FOM) and the use of an elevated water table. The study was conducted using intact peat cores which were divided into two water table treatments, 20 and 50 cm from the surface of the soil. Half of the cores had added FOM in the form of barley (*Hordeum vulgare* L.) straw. The results show that both FOM and an elevated water table can lead to a reduction in CO_2. The effect of FOM was still noticeable in the long term, though not as pronounced as it was at initial application. Regular application of FOM could be beneficial, but the use of an elevated water table is the best approach as it leads to significant reductions in CO_2 fluxes. However, due to complications associated with wide-scale water table deployment (e.g., cost) and the potential effect on agricultural output, its adoption could prove impossible on active farms.

4.1 Introduction

The global population is projected to grow to an estimated 10 billion people in the next 30 years, up from 7.9 billion currently (Hickey et al., 2019). As a result of this trend in population growth, more land for food production is undoubtedly going to be required unless major improvements can be made to current agricultural practices e.g., the adoption of sustainable intensification where more crops can be produced with less land and resources. If the status quo is maintained, anthropogenic greenhouse gas (GHG) emissions from agriculture are projected to continue increasing as global food demand grows. From 1961 to 2005, the world's population grew by 111% up from 3.08 to 6.51 billion. This growth was combined with increased crop production of 162% from 1.8 to 4.8 billion tonnes per year (Burney et al., 2010). Figure 4.1 shows the growth trend of population, crop area, crop production, and the use of fertilisers.

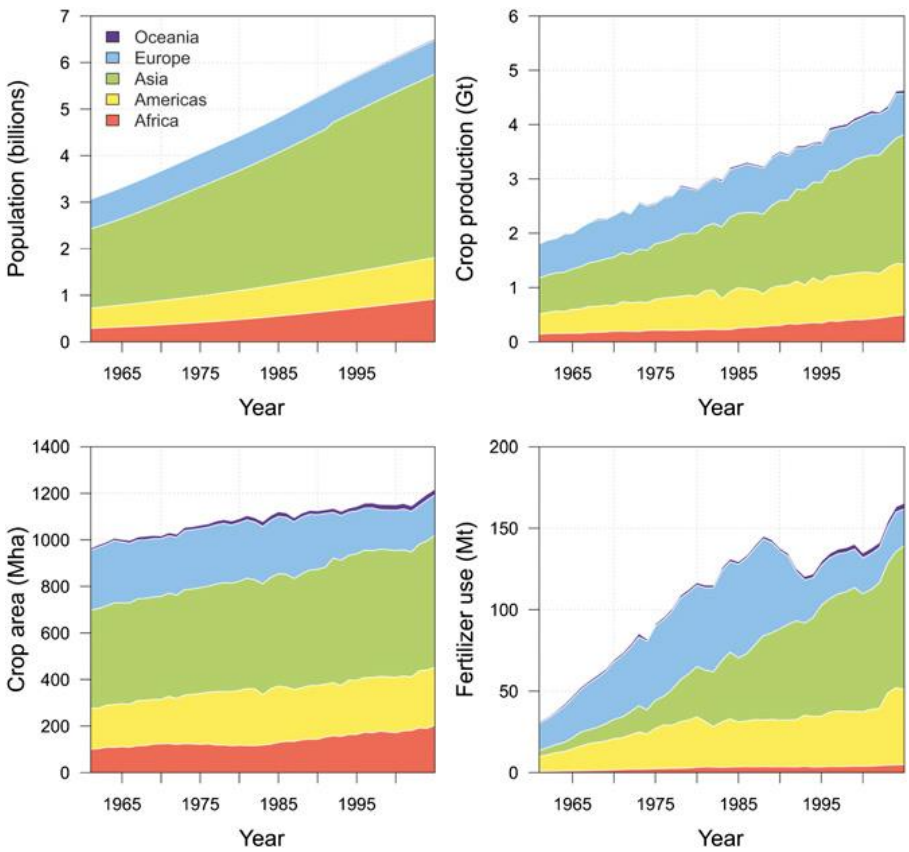


Figure 4.1 Global population growth (top left), crop area (bottom left), crop production (top right), and fertiliser use (bottom right) between 1961 and 2005 (Burney et al., 2010). As the population has grown so has the amount of land that is required to meet the increased food demand.

In 2006, global agricultural production contributed between 1.4 to 1.7 gigatonnes of carbon (GtC) emissions into the atmosphere, equivalent to 10-12% of total anthropogenic GHG emissions (Burney et al., 2010). In Europe, the EEA reported in 2016 that agriculture was the second largest contributor of GHG emissions (European Environment Agency, 2016; 2021).

Cultivated peatlands produce a considerable amount of CO_2 in the UK. Parish et al. (2008) and Natural England (2010) reported that cultivated peatlands produce over six times more CO_2 than peatlands that have been restored to a near-natural condition, which is 26 t CO_2e ha^{-1} yr^{-1} compared to only 4 t CO_2e ha^{-1} yr^{-1} from restored peatland. However, this amount has since increased, with Evans et al. (2017) estimating that cultivated peatland produces 39 t CO_2e ha^{-1} yr^{-1}.

Restoring peatlands to their natural state can turn them from sources of CO_2 to sinks of CO_2. Unfortunately, that will take them out of food production, leading to a negative economic outcome and possible food shortages and/or a need for increased food imports. It is likely that this could, in turn lead to increased emissions from the importation of food from other countries (Webb et al., 2013). Peatlands in the UK are important for food production, especially of fresh vegetables, thus taking them out of agricultural production is currently not an option. The Fens, for example, have a total agricultural production, including livestock worth £1.23 billion yr^{-1} (NFU, 2019). Farmers in The Fens grow a variety of crops such as lettuce (*Lactuca sativa* L.), celery (*Apium graveolens* L.), potatoes (*Solanum tuberosum* L.), onions (*Allium cepa* L.), garlic (*Allium sativum* L.), oilseed rape (*Brassica napus* L.), peas (*Pisum sativum* L.), sugar beet (*Beta vulgaris* L.), maize (*Zea mays* L.), wheat (*Triticum aestivum* L.), and barley (*Hordeum vulgare* L.) (NFU, 2008; Wood et al., 2019). The total annual crop output in The Fens is worth £700 million with vegetables making up the largest portion of this output at £357 million. Table 4.1 summarises the crop outputs and their worth in The Fens.

Table 4.1 Value of crops grown in the East Anglian Fens and the percentage they contribute to agricultural output (NFU, 2019).

Crop	**Percentage of crops in England (%)**	**Value in million pounds (£m)**
Potatoes	20	112
Sugar beet	20	30
Vegetables	32.8	357
Plants and flowers	21.4	232
Fruit	31	19
Total crop output	21.5	750

Due to their commercial importance for food production, lowland peatlands are likely to remain in agricultural use for the near future. However, the UK government has highlighted the need for policies and interventions to protect these valuable peat soils. In a report on how the UK can contribute to tackling global warming, the Committee on Climate Change (2019) emphasised the need to improve peatland management and restoration where possible. In addition to a large amount of GHGs emitted from soil, the UK government has concluded that over £1 billion a year is lost due to soil degradation alone (Parliamentary Office of Science and Technology, 2015; DEFRA, 2021). This figure, however, includes all types of soil, not just peatlands. This amount is certainly going to increase as more pressure is put onto the already fragile peatlands due to increased food demand that is associated with population growth and progressive urban expansion.

Numerous mitigation measures have been discussed to preserve the UK's cultivated peatlands, for example, i) longer crop rotations, ii) use of no-tillage or reduced tillage methods, iii) growing winter cover crops e.g., wheat, barley, clover (*Trifolium spp.* L.), and vetch (*Vicia spp.* L.) or other native legumes instead of leaving it fallow, then ploughing in the crop residue as green manure or fresh organic matter (FOM), and iv) raising water tables to reduce peat

oxidation (Locke & Bryson, 1997; Musarika et al., 2017; Triplett Jr & Dick, 2008; Ostle et al., 2009; Parliamentary Office of Science and Technology, 2015).

Bringing the water table closer to the soil surface can lead to a significant decrease in the fluxes of CO_2 as anoxic conditions are created. This enables the reduction of peatland degradation by decreasing oxidation of the soil, therefore changing the microbial community towards methanogens (Moore & Dalva, 1993). Water tables that are more than 20 cm below the surface have been shown to have little to no impact on the fluxes of CH_4 (Matysek et al., 2019; Musarika et al., 2017; Taft et al., 2017). However, with a water table of less than 20 cm, it is expected that the CH_4 levels will increase due to methanogenesis as the anoxic environment is brought up closer to the soil surface. Increasing the water table depth has been shown to either increase or decrease crop yield, depending on the type of crop. Renger et al. (2002) showed that an elevated water table in a grassland led to a 10% crop yield loss. In horticultural crop studies, Matysek et al. (2019) who studied celery observed a loss of crop yield, in elevated water table cores; they reported that celery shoots both wet and dry weight were on average 19% lower under a 30 cm water table compared to the 50 cm water table. However, increased crop yield has been observed in radish (*Raphanus sativus* L.) grown at 30 cm water table depth, and ryegrass (*Lolium perenne* L.) grown at an elevated water table (Musarika et al., 2017; Berglund & Berglund, 2011). Regardless of the reported minor losses in crop yields, raising water table depths is the only viable long-term mitigation measure if crop production is to be maintained on peatlands. It is a compromise that could prolong crop production on drained peatlands if other feasible solutions are not readily available.

The addition of crop residues (e.g., from cover crops) has been reported to improve and maintain the soil quality and further prolong its suitability for agriculture by effectively reducing the decomposition of old peat (Bader et al., 2018). However, the crop residue or FOM is an essential source of energy and nutrients for soil microbes. Therefore, its addition to the soil can potentially increase the rate of SOM mineralisation, especially in peatlands where a considerable proportion of the existing SOM is recalcitrant (Bader et al., 2018; Fontaine et al., 2003). In the previous chapter (Chapter 2), the effects of fresh organic matter (FOM) were visible on GHG fluxes. Even though the application of FOM led to significantly increased emissions of CO_2, the majority of the applied FOM remained, therefore enhancing the amount of C in the soil. In the 20 cm water table (WT) treatment, 83.8% of the FOM remained in the soil and in the 50 cm WT cores, 68.4% of the added FOM remained in the

soil. The addition of FOM led to decreased N_2O emissions in the business-as-usual setup of 50 cm water table and it led to minimal CH_4 consumption in both 50 cm water table depth and 20 cm water table depth. Nevertheless, the study only looked at the immediate effects of added FOM not the longer-term impacts on SOM dynamics. The earlier study focused on only the first six months after FOM addition. Fewer studies have explored the effects of FOM in the long term, there is a particular gap in understanding regarding whether the soil remains undisturbed after the application of FOM (Bader et al., 2018; Wang et al., 2015; Wen et al., 2019).

This study aimed to assess how CO_2 from cultivated peatlands will respond in the long term to added FOM of a commercially important crop. Furthermore, it assesses the additional impact of an elevated water table depth as a potential option to reduce CO_2 fluxes. The hypotheses are that i) after the initial application of FOM, the fluxes will stabilise as the FOM would either have been fully incorporated into the soil or have been mineralised. Therefore, in this study, the effect of FOM on GHG fluxes is not expected; ii) raising the water table depth has long term benefits; therefore, it is expected that there will be significant differences in the long-term CO_2 fluxes between the low water table and the high-water table treatments. Therefore, confirming that use of elevated water table has long term benefits.

4.2 Materials and Methods

The experiment used the same cores as Chapter 3 which were collected from Rosedene Farm. However, modifications were made to the above ground setup of the cores to allow the growth of crops.

4.2.1 Above ground LI-COR LI8100 setup

In this study, the original plan was to grow crops in the cores, but due to the COVID-19 pandemic lockdowns, this did not prove possible. However, prior to the lockdowns, modifications were made to the cores to allow crops to grow unimpeded by creating more headroom. The LI-COR autochambers normally have about 10 cm clearance from the ground. To increase ground clearance, individual mesocosms were connected using a coupler to Perspex pipes with a diameter of 200 mm and a height of 300 mm (Figure 4.2A). 300 mm was chosen as the ideal height as this creates a reduced greenhouse effect within the cores and removes the need to have a fan to keep the air circulating, which in turn can affect the gas fluxes being observed. The LI-COR collars were fitted onto the Perspex pipes, then the LI-

COR autochambers were placed on the top (Figure 4.2B). The autochambers were elevated from the ground using 60 mm PVC pipes that acted like stilts and were anchored to the ground to give solid support (Figure 4.2C&D). The LI-COR multiplexer only allows up to 16 autochambers to be connected. Therefore, the number of replicates in this experiment was limited by the number of autochambers.

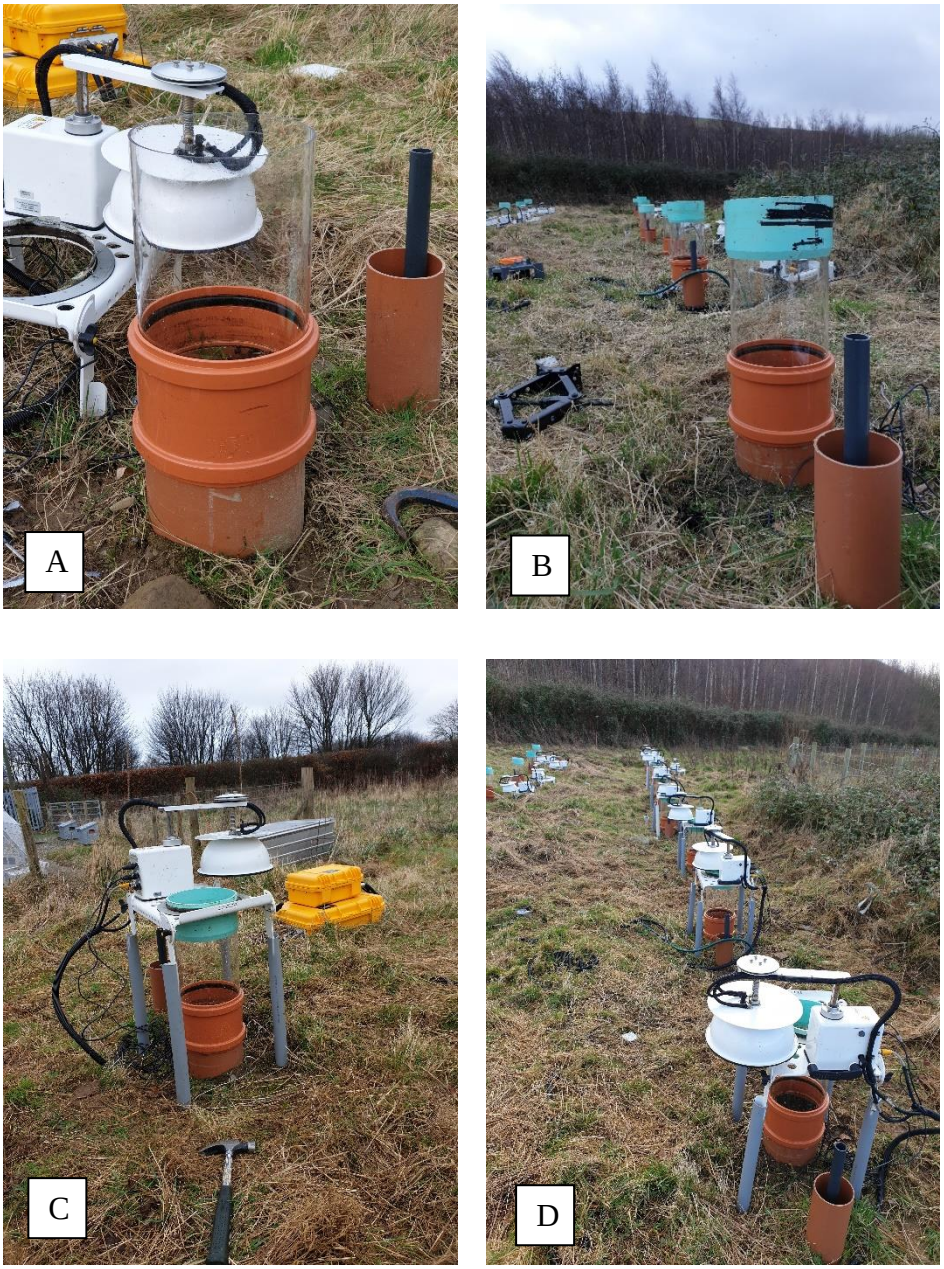


Figure 4.2 The above-ground setup of the mesocosms showing A) a Perspex pipe coupled using a 200 mm coupler to a core that is buried in the ground. To the right, a depth management pipe can be seen in the 110 mm dip well. B) The LI-COR collars (blue) were fitted onto the Perspex pipes before the LI-COR autochambers (Long Term Chamber – Model 8100-104) were connected. (C&D) The LI-COR autochambers were elevated above ground using 60 mm PVC pipes as stilts which were anchored to the ground.

4.2.2 LI-COR LI8100A offsets

The LI-COR LI8100A Infra-Red Gas Analyser (IRGA) requires that offsets for the collars be defined in the software to account for any volume displacement discrepancies due to varying ground clearance. According to the manual, the chamber offsets are normally measured by the distance between the soil surface and the upper edge of the chamber base plate. The first step is to measure the distance from the soil surface to the top of the soil collar and then subtract the distance between the upper edge of the chamber base plate and the top of the soil collar to obtain the offset value (LI-COR, 2007). However, as the setup was modified to allow the use of the Perspex pipes, calculating the offsets was not necessary. All the offsets were set at 1 cm. The Perspex pipes were taken as an extension of the chamber; therefore, the volume of the chamber was altered manually in the software to reflect the increase in the volume.

4.2.3 Water table management

The water table was supposed to be measured and maintained every week throughout the study. However, due to the extended lockdown caused by the COVID-19 pandemic, it was not possible to visit the experiment to verify the water table depths. Only towards the end of the study were the water table depths measured. Nevertheless, it is not expected that the cores would have completely dried out as precipitation events would have let water into the cores. The design of the water table management allows for excess water to overflow out of the cores to maintain the set depth of 20 cm and 50 cm.

4.2.4 Fresh organic matter (FOM)

No additional FOM was added to the cores, this study analysed the long-term effect of FOM that was added from the previous experiment in Chapter 4. In Chapter 4, 10 grams of barley straw was added to the cores.

4.2.5 Gas analysers

The study was set up to use two GHG analysers. A Los Gatos UGGA was connected in series with a LI-COR IRGA LI8100A. The UGGA was intended to measure the fluxes of CH_4 whilst the IRGA collected the CO_2 fluxes. Unfortunately, a fault occurred with the UGGA that meant it was not possible to collect CH_4 data during the experiment. Due to the COVID-19 lockdown, it was not possible to attend to the equipment while it was running, therefore this fault was not discovered until the end of the study. The LI-COR IRGA LI8100A was connected to a 16 port

LI-COR LI-8150 multiplexer. The LI-COR IRGA measured CO_2 flux from soils continuously at 1 Hz.

4.2.6 LI-COR LI8100A autochambers

The hemispherical shape of the LI-COR Long Term Chamber –Model 8100-104 (Figure 4.3) together with the position of the inlet and outlets allows the mixing of the air in chambers (LI-COR, 2016). Because the system does not use a fan, there are no artificial chamber pressure perturbations (Hanson et al., 1993). Furthermore, the LI-8100A chambers feature a radially symmetrical pressure vent that maintains ambient pressure inside the chamber thus eliminating sensitivity to wind direction (LI-COR, 2016).

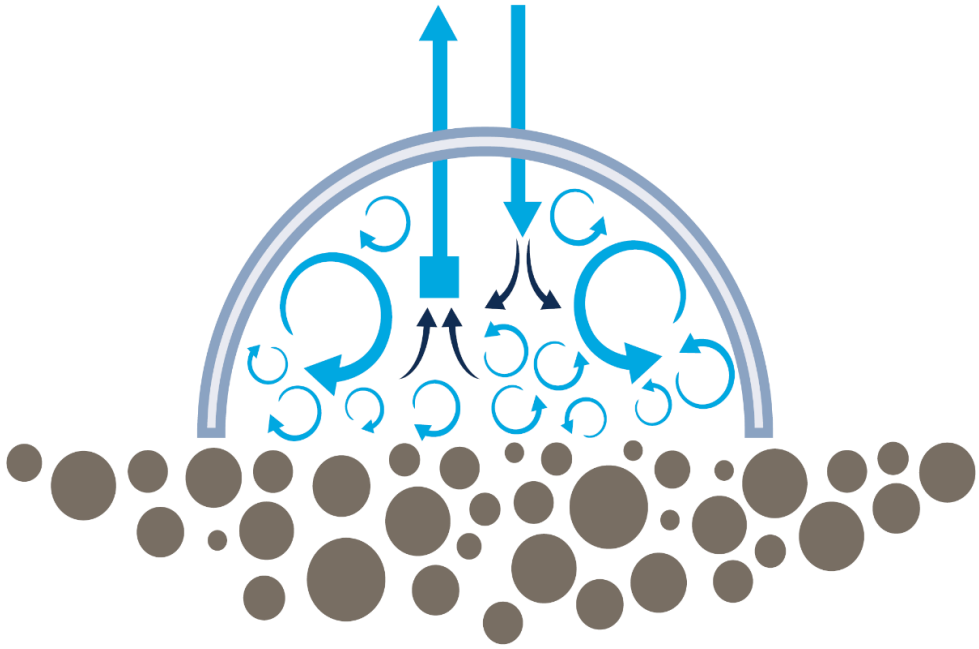


Figure 4.3 The hemispherical LI-COR Long Term Chamber LI-8100-104 chamber shows the position of the inlet and outlet. The position of the inlet and outlet allows the mixing of air without the need for a fan which would lead to artificial chamber pressure perturbations. Image source: (LI-COR, 2016).

4.2.7 Experiment timeline

The experiment was conducted from the 19th of March 2020 until the 11th of September 2020. Due to the COVID-19 pandemic and associated lockdowns during this period, data was collected automatically while the facility was under lockdown.

4.2.8 Statistical analysis

All figures that accompany results in this study were produced using R/RStudio. Statistical calculations were also done using R/RStudio. R Studio Version 1.4.1717 with R version 4.1.0 (2021-05-18). Statistical significance was accepted at $p \leq 0.05$.

The data in this study was measured repeatedly on the same cores during the experiment so a Repeated Measures ANOVA (Analysis of Variance) approach was used. A mixed-effects model was chosen because the same cores were measured continuously throughout the experiment therefore they would have been a problem with pseudoreplication. Pseudoreplication is a problem because repeated measures on the same core will mean there is no independence of errors which is an important assumption of standard statistical analysis (Crawley, 2013). The lmerTest package was used to evaluate the linear model of the data to calculate statistical significance.

4.3 Results

4.3.1 Rainfall during the experiment

During the experiment, a total of 504.2 mm of rain was recorded at Henfaes Farm. Table 4.2 summarises the total amount of rainfall and the monthly totals. August was the wettest month with a total of 200.8 mm of rainfall recorded and May was the driest month with only 9.6 mm of rainfall recorded.

Table 4.2 Summary of the rainfall recorded at Henfaes Farm during the experiment. The table shows the summary for each month.

Month	Total Rainfall (mm)
Apr	14.2
May	9.6
Jun	115
Jul	132.2
Aug	200.8
Sep	32.4
Total	504.2

4.3.2 Cumulative carbon dioxide (CO_2) fluxes summary

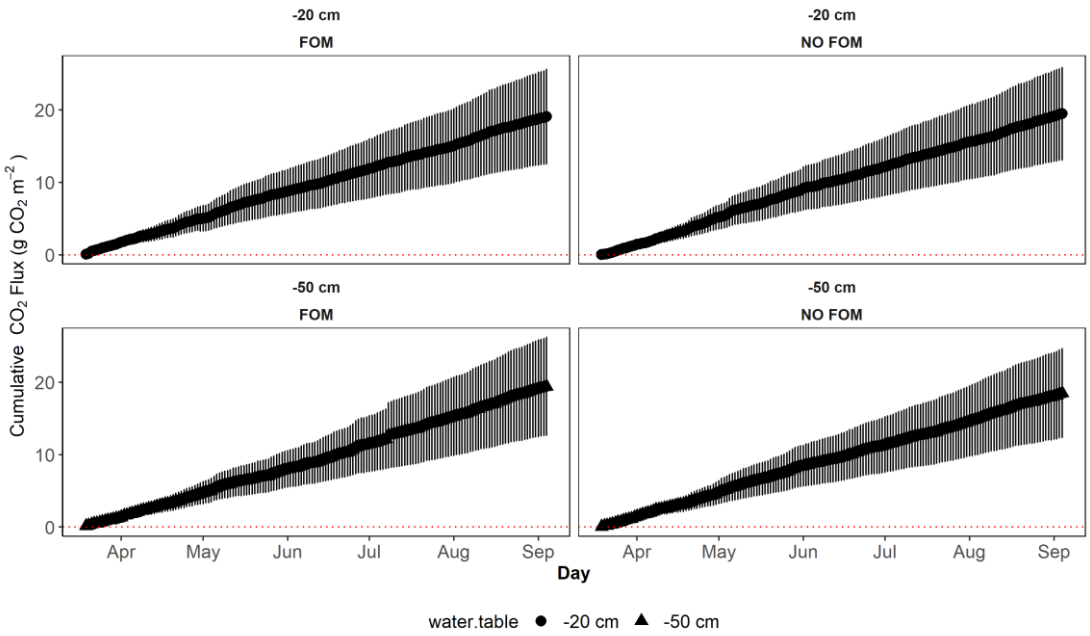


Figure 4.4 Daily cumulative carbon dioxide (CO_2) fluxes for the duration of the study. Values represent means ± SEM (n = 4).

Figure 4.5 summarises the total daily cumulative amounts of all the treatment, FOM and water table. The trends of the daily fluxes from the beginning to the end of the experiment are shown in Figure 4.4.

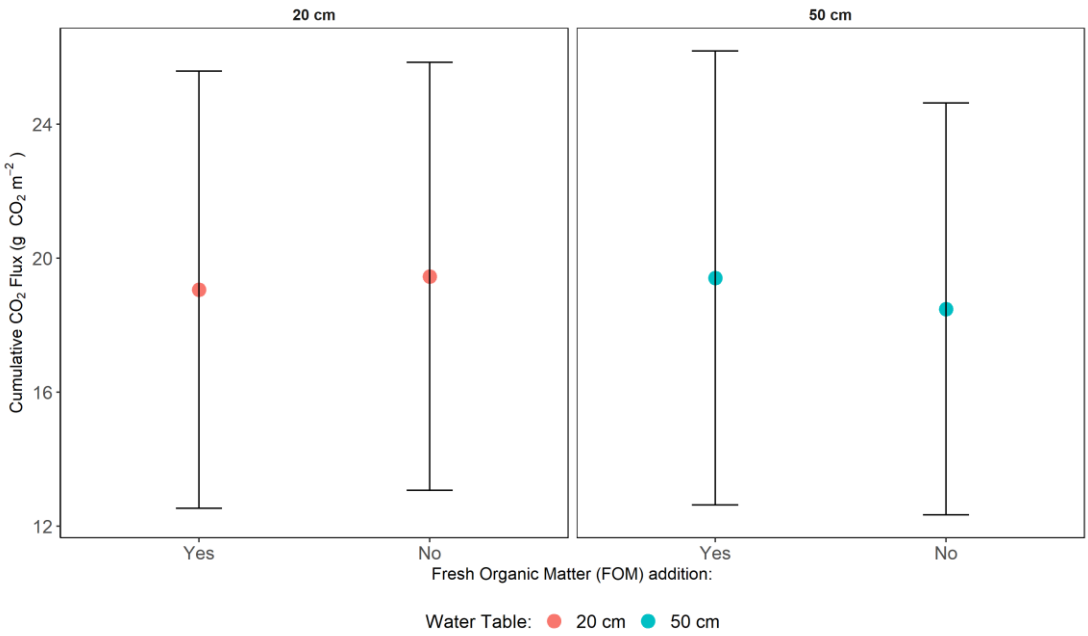


Figure 4.5 Total daily cumulative fluxes of carbon dioxide (CO_2) for the duration of the experiment. Values are average cumulative totals of all cores ± SEM (n = 4).

The CO_2 fluxes of FOM addition versus none were significantly different ($p < 0.001$). Cores with added FOM in the 20 cm water table had lower CO_2 than cores without FOM. The FOM treatment in the 50 cm water table produced more CO_2 than the same treatment in the 20 cm water table. The difference between the treatments without FOM treatment in the 20 cm and 50 cm water table was significant ($p < 0.001$). Overall, the 50 cm treatment produced less CO_2 compared to the 20 cm water table ($p < 0.001$), due to the unexpectedly low fluxes in the treatment without FOM.

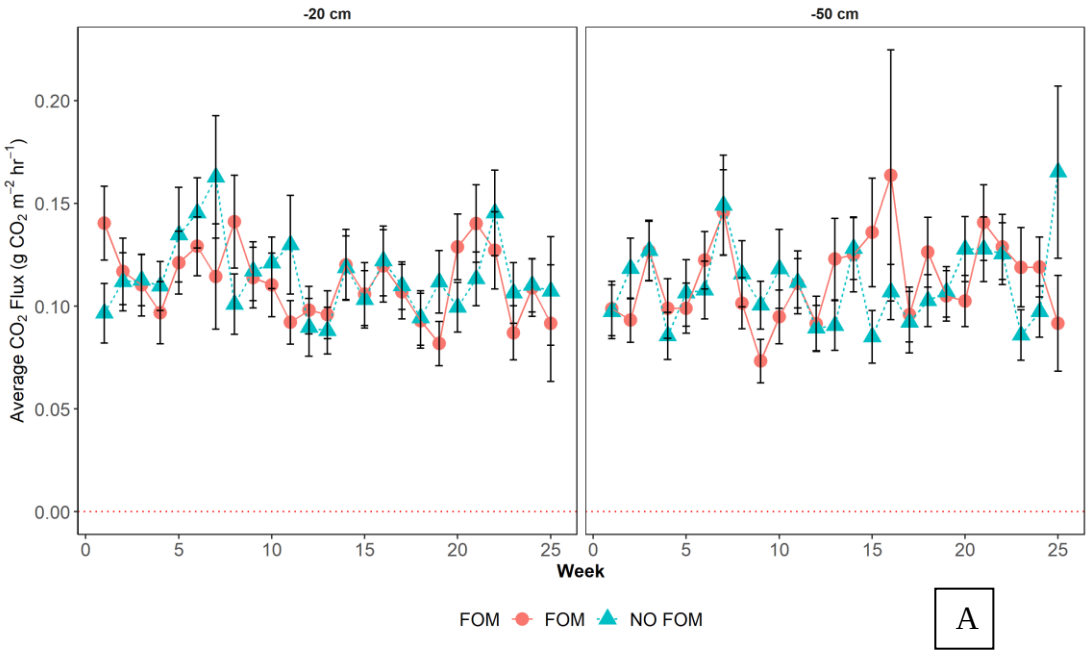
4.3.3 Average carbon dioxide (CO_2) fluxes

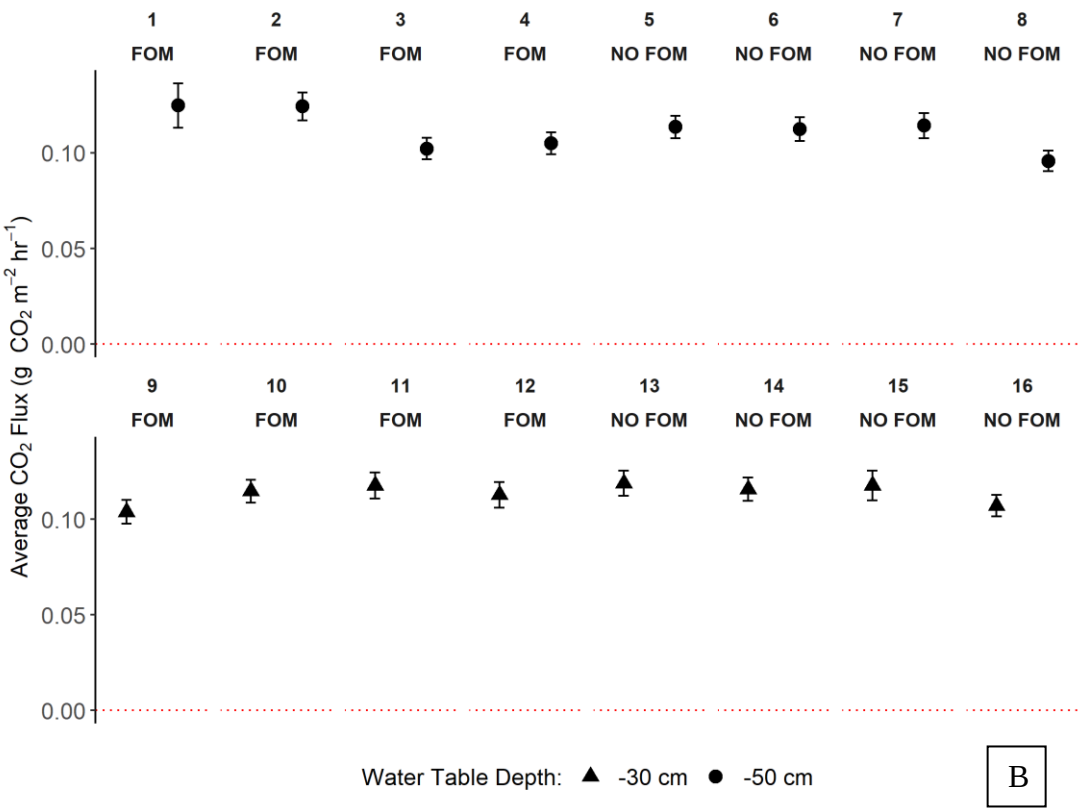
In addition to calculating the cumulative fluxes, averages of the fluxes were also calculated. Table 4.3 summarises the average CO_2 fluxes.

Table 4.3 Average carbon dioxide (CO_2) fluxes for the duration of the study. Values represent means ± SEM (n = 4).

Water table Depth	**Fresh organic matter (FOM)**	**Mean CO_2 g CO_2 m^{-2} hr^{-1}**	**CO_2 SEM**
20 cm	+	0.112	0.0032
20 cm	-	0.115	0.0033
50 cm	+	0.114	0.0040
50 cm	-	0.109	0.0030

Figure 4.6 compares the weekly averages of the cores from all the treatments. It shows the CO_2 fluxes of the cores with added FOM and those without in each water table plotted on top of each other. The averages are not too dissimilar from each other.


-20 cm
-50 cm
Average CO_2 Flux (g CO_2 m^{-2} hr^{-1})
0.20
0.15
0.10
0.05
0.00
0
5
10
15
20
25
Week
FOM
FOM
NO FOM
A


1 FOM
2 FOM
3 FOM
4 FOM
5 NO FOM
6 NO FOM
7 NO FOM
8 NO FOM
9 FOM
10 FOM
11 FOM
12 FOM
13 NO FOM
14 NO FOM
15 NO FOM
16 NO FOM
Average CO_2 Flux (g CO_2 m^{-2} hr^{-1})
0.10
0.05
0.00
Water Table Depth:
-30 cm
-50 cm
B

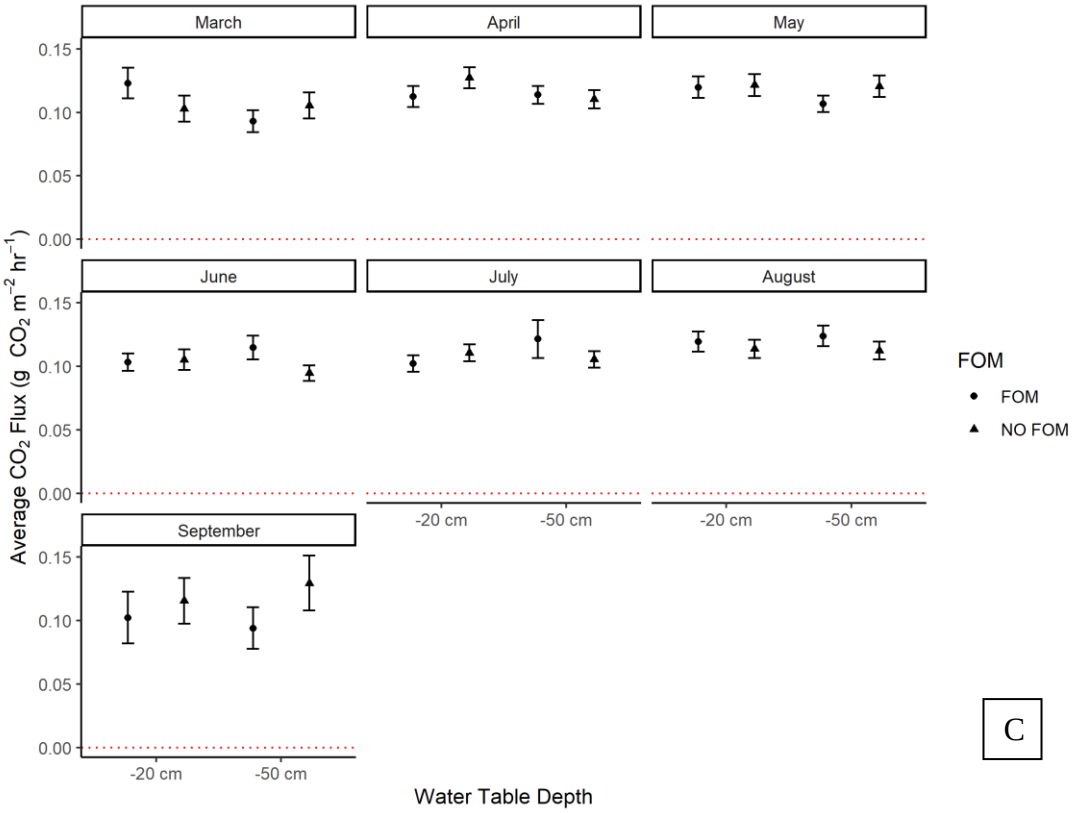


Figure 4.6 (A). Compares the weekly average carbon dioxide (CO_2) fluxes in the 20 cm and 50 cm cores. The fresh organic matter (FOM) treatments are plotted on top of each other. (B). Shows the average CO_2 fluxes from each core for the duration of the study. Cores 1-8 had a water table of 50 cm while for cores 9-16 it was 20 cm. Cores 1-4 and cores 9-12 were previously treated with FOM. (C). Monthly averages of the CO_2 fluxes during the study from March to September 2020. Values represent means ± SEM (n = 4).

4.4 Discussion

4.4.1 Carbon dioxide (CO_2) fluxes

When the water table is brought closer to the surface, the expected result is a reduction in the fluxes of CO_2. This was indeed the observation in this study. Even though the difference was small, the cores that were treated with FOM in the 20 cm water table produced less CO_2 compared to the 50 cm cores treated with FOM. This result can be ascribed to the water table suppressing CO_2 production due to the prevailing anoxic conditions with greater waterlogging. This result confirms that the water table set up was working as intended even though it was not possible to regularly attend to and maintain the dip wells that control the water table. This result is in line with findings from other studies that have shown that water table depth will influence the fluxes of CO_2 (Kritzler et al., 2016; Matysek et al., 2019; Musarika et al., 2017; Swails et al., 2019). However, in the 20 cm cores without FOM treatment, the cumulative CO_2 flux was higher compared to the 50 cm cores without FOM. This observation could either be due to the failure in the water table as it was not possible to check the water table as often as planned meaning that the water table was primarily influenced by precipitation events or due to the combination of FOM and elevated water table enabling greater CO_2 consumption or another effect that needs further investigation.

Water table depth is undoubtedly the best option to suppress the fluxes of CO_2. Lafleur et al. (2005) add that for a water table to have a significant effect, a water table depth greater than 30 cm below the surface needs to be achieved. This is also supported by conclusions drawn by Astiani et al. (2016) and Evans et al. (2021) that a water level close to the soil surface is needed to reduce soil CO_2 emissions. However, having water table depths that are closer to the surface could significantly impact horticultural production. Most farm equipment such as tractors and crop harvesters are very heavy and therefore traffic may not be possible with a water table close to the surface. However, specialised machinery exists with tracks that exert low ground pressure allowing them to operate in areas with water table depths closer to the surface (Wichmann, 2017). Another compounding factor to consider when water table depths are brought closer to the surface is the potential of the prevalence of plant fungal diseases such as water moulds (Oomycetes) e.g., *Aphanomyces*, *Pythium*, and *Phytophthora* that will significantly impact crop yield (Katan, 2000); *Phytophthora*, caused potato blight from 1845 leading to the Great Famine in Ireland which caused starvation, death, and mass migration (Geber & Murphy, 2012). Therefore, the threat of plant disease should not be overlooked. Plant

diseases could be controlled using chemicals e.g., fungicides but this could bring further problems which will need further investigation. Alternatively, the use of biological approaches such as antagonistic fungi could be a better option (Jorjani et al., 2012).

4.4.2 Methane (CH_4) fluxes

If the LGR UGGA had worked as intended, the expectation was to see an increase in CH_4 in the cores with the 20 cm water table depth. As the aerobic conditions that led to significantly reduced CO_2 emissions from the soil are replaced by anaerobic conditions, this environment will favour methanogens resulting in increased CH_4 fluxes (Rey-Sanchez et al., 2019; Reumer et al., 2017; Wen et al., 2018).

4.4.3 Long term effects of fresh organic matter (FOM)

This study shows that in the long term, FOM has a moderate influence on fluxes of CO_2. 22 months after the initial application of FOM, CO_2 fluxes were still being influenced by the added FOM. In the short term, the initial application of FOM increased CO_2 fluxes (Chapter 2), but the majority of added FOM remained in the soil. In the 20 cm cores, only 16% of the added FOM carbon was lost as CO_2 whereas in the 50 cm cores, 31.6% of the added FOM carbon was lost as CO_2. It is reassuring to observe that the FOM does not have a long-term effect on the fluxes and if the water table is maintained as high as is possible, then this could lead to an increase in the SOC of cultivated peatlands. This finding is supported by the findings by Linkosalmi et al. (2015) who concluded in their study that organic soils do not support priming to the same extent as mineral soils when fresh organic matter is added. However, it is important to note that due to the limitations of this current study i.e., no use of a 14C or 13C isotopic partitioning method, it is difficult to conclude if any of the FOM was incorporated into the soil or was lost through mineralisation, especially in the cores that had a water table further down from the surface of the soil.

4.4.4 Study limitations

This study was initially set up to study the long-term impact of added FOM on GHG fluxes and its future contribution when plants are included. In the study, the intention was to i) plant crops in the cores ii) analyse the loss of carbon as dissolved organic carbon (DOC) and iii) add fertiliser at the proportions used at Rosedene Farm where the soil samples were collected. Modifications were made to the cores set up and LI-COR autochambers to increase the ground

clearance to allow the inclusion of plants. Nevertheless, due to the COVID-19 pandemic lockdown, the scope of this study was significantly reduced. This meant that it was not possible to include plants as the study site was inaccessible during the growing season. Further, it was not possible to check that the equipment was functioning that the water tables were correct or whether weeds or algae were growing in the cores. During the experiment, a Los Gatos Research UGGA was set up in parallel to provide a reading on the fluxes on CH_4 but unfortunately, all the data collected by the UGGA is unusable as it did not show any fluxes. It is possible that either the UGGA was faulty, or a gas line was not connected properly or became loose after setup. However, as it was not possible to attend to the site until towards the end of the study, these faults were discovered too late to rectify them. The LI-COR IRGA LI8100A worked as expected to provide CO_2 readings during the experiment, which became the focus of this study. Though the study produced significant results, it is important to emphasise that the differences in fluxes are quite small and could very well be down to noise.

4.4.5 Conclusions

This study has demonstrated that the effect of water table depth is long term, and due to its benefits in suppressing CO_2 is the best approach to preserving cultivated peatlands. But this approach comes with practical, cultural, and economic challenges that could make it unfeasible for crop production. Whilst studies such as Musarika et al. (2017) and Berglund & Berglund (2011) have shown that a water table depth closer to the surface will not be detrimental to crop production, other studies such as Matysek et al. (2019) and Renger et al. (2002) have shown that having the water too close to the surface will impact crop production. Increased water supply will only be beneficial to shallow rooting crops that will not be affected by the excess water and to crops such as radish that thrive with increased water intake, which will lead to less irrigation (Lambers et al., 2013).

Water tables close to the surface have been successful in mesocosm based experiments similar to the ones in this study, however, deploying them full-scale at farm level poses numerous challenges. First, farmers will have to invest in new machinery that works in the shallower water table depths. Whilst the machinery is expensive, this is not an unsurmountable problem. The biggest challenge is how these shallow water table depths will be delivered at a farm scale. Dawson et al. (2010) alluded to the difficulty in establishing water tables due to the topography of the area. Currently, the water table is managed for all the fields simultaneously by the Water Board and there are four different water tables managed in the East Anglia Fens due to the

topographical variations (Dawson et al., 2010). To achieve the desired water table depths, this ideally must be done on a per-field or per farm basis to allow careful control of the water tables and to ensure that the water is not draining away. Furthermore, there may be a problem with other infrastructure such as roads and power lines which may become affected by the shallow water table.

The use of FOM seems straightforward, even at a usual set up of a water table depth of 50 cm from the surface of the soil, a considerable proportion of the added FOM remains in the soil. Ideally, a water table depth of 20 cm from the soil surface would have greater benefits but due to the complications of managing water tables at the farm level, a 50 cm compromise would still be a better option than leaving all the soil exposed to oxidation.

Chapter 5. Effectiveness of $CaSO_4$, $FeCl_2$ and/or $FeSO_4$ at suppressing greenhouse gas emissions and dissolved organic carbon loss from cultivated peatlands

5.1 Abstract

Drained and cultivated peatlands represent one of the biggest agricultural sources of CO_2 released into the atmosphere. Raising the water table in these cultivated peatlands offers one potential solution to reduce CO_2 losses, however, it may also hinder farming activities, increase plant disease prevalence, and turn the peatland from a sink into a source of CH_4. The hypothesis was that the addition of terminal electron acceptors to soil (e.g., Fe^{3+} and SO_4^{2-}) could offer a practical way to minimise CH_4 emissions under a high-water table management regime. In this mesocosm-scale study, greenhouse gas (GHG) emissions were quantified in response to three soil amendments ($FeCl_2$, $FeSO_4$, and $CaSO_4$) and three different water table depths (0, 20, and 50 cm). Overall, the results showed that terminal electron acceptor addition did little to counter GHG emissions. In contrast, the addition of $FeSO_4$ and $FeCl_2$ led to an increase in CO_2 loss. $CaSO_4$ was moderately effective at reducing GHG losses, but not enough to justify its use as an amendment. Nevertheless, this study did not assess the effects under all field conditions such as the presence of crops, fluctuating water tables, cooler temperatures, and extended time in the soil. The treatments might have more effect in a warm, wet, and vegetated long-term experiment. Overall, the results showed that water table manipulation was highly effective at lowering CO_2 and that in the short-term it did not stimulate CH_4 losses. Furthermore, CH_4 consumption was apparent at both 0 cm and 20 cm and was stimulated in the presence of crop residue. The amendments proved successful at reducing the amount of dissolved organic carbon (DOC) lost from the soil as the DOC from all the treatments was significantly lower than the control treatment. Based on the findings presented here, longer-term studies are required to ascertain when CH_4 production will commence in peatlands with elevated water tables. However, in the short term, it is evident that raising water tables represents an effective strategy to reduce GHG emissions in cultivated peatlands.

5.2 Introduction

Drained agricultural peatlands are amongst the biggest sources of greenhouse gas (GHG) as they lose carbon (C) in the form of carbon dioxide (CO_2). Waterlogged or flooded peatlands, on the other hand, are an important sink of C. CO_2 absorbed by peatlands in the northern hemisphere is estimated to be 0.10 ± 0.02 Pg C y^{-1} (Hugelius et al., 2020). Nevertheless, these peatlands are an important source of atmospheric methane (CH_4) (Hugelius et al., 2020). Northern hemisphere peatlands release 31-65 Tg CH_4 yr^{-1} (Mcguire et al., 2009). Even though CH_4 has a lower atmospheric concentration (about 2 ppm) than CO_2 (over 400ppm), its radiative forcing is 28 times that of CO_2 over a hundred years (IPCC, 2014; Lindsey, 2020). However, unlike CO_2, the residence time of CH_4 in the atmosphere before it is oxidised is only 7-10 years (Ruddiman & Thomson, 2001). CO_2 can persist in the atmosphere for longer, although the exact residence time is hotly contested (Cawley, 2011; Harde, 2017; Köhler et al., 2018). About 65% to 80% of CO_2 released into the atmosphere is dissolved into the oceans over 20-200 years depending on numerous factors, such as temperature. Photosynthetic uptake consumes a small proportion of atmospheric CO_2. However, the rest of it is removed by processes such as chemical weathering and rock formation, which can take hundreds of thousands of years (Archer et al., 2009). Despite the differences between Global Warming Potential (GWP) and residence time in the atmosphere between CO_2 and CH_4, CH_4 is the second most important GHG in terms of contribution to climate change and therefore it needs to be reduced alongside CO_2. Nevertheless, challenges arise from the fact that one of the mitigation measures for CO_2, water table close to the surface, can potentially lead to increased CH_4 emissions from the soil.

Raising the water table in drained peatlands is an effective method to reduce the net loss of CO_2. However, if the water table is too close to the surface, there may be a considerable increase in CH_4 emissions over time as a community of methanogens become more prolific. Wicken Fen is an area of wetland in Cambridgeshire in the East Anglian Fens, which has been cited multiple times as an example of the benefits of rewetting peatland, with increases in biodiversity and reductions in CO_2 emissions (Boreham, 2018; McCartney & Hera, 2004). However, Wicken Fen produces considerable amounts of CH_4 with an annual average of 3.8 to 4.0 g CH_4 m^{-2} yr^{-1} (Kaduk et al., 2015). Figure 5.1 shows the (expected) relationships between different gases and water table depth. Having the water table closer to the surface creates an anoxic environment which produces favourable conditions for methanogens (as

opposed to methanotrophs under oxic conditions), leading to increased CH_4 release from the soil (Moore & Dalva, 1993; Yang et al., 2014). Water tables of 30 cm or below will not be as big a problem compared to water tables closer to the surface that could have more CH_4 release, specifically 0 to 30 cm (Renger et al., 2002). Water table depths of less than 10 cm will be more conducive for methanogenesis hence will lead to the soil becoming a CH_4 net source (Evans et al., 2021). If shallower water tables are to be chosen, the resulting CH_4 fluxes need to be addressed. The effect between water table depth and N_2O has been shown by studies such as Renger et al. (2002) and Couwenberg et al. (2010) as generally having a linear relationship. However, it is important to state that this relationship is uncertain and can be influenced by temperature (Yang et al., 2013).

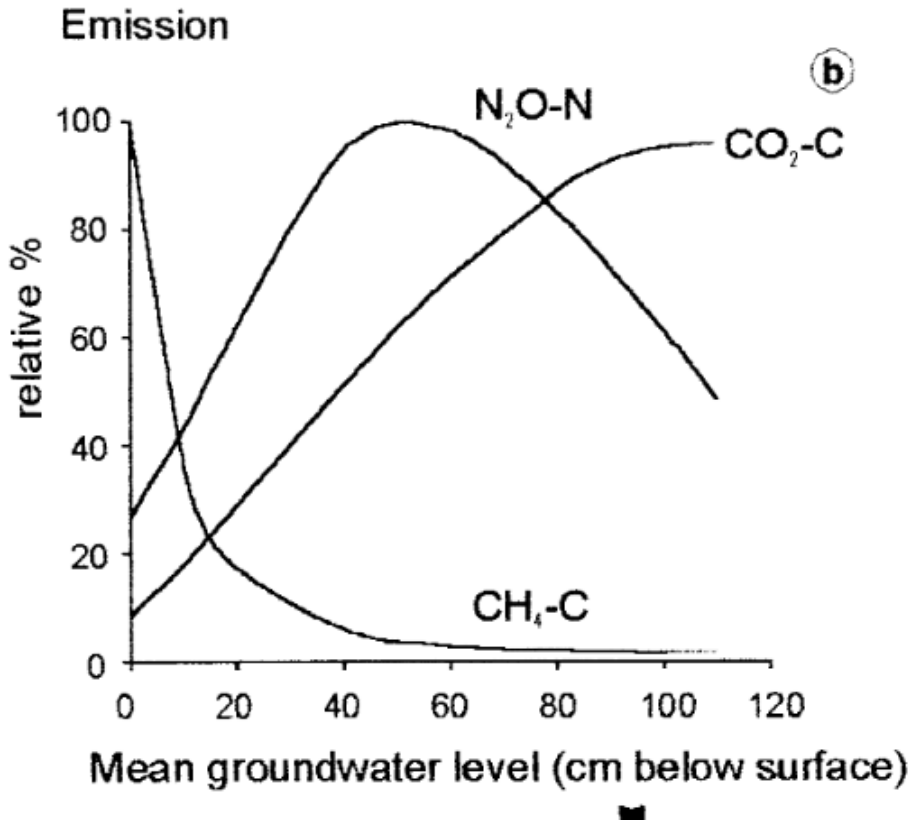


Figure 5.1 Net fluxes of nitrous oxide (N_2O), carbon dioxide (CO_2) and methane (CH_4) relative to water table depth in peatlands. More CH_4 is observed when the water table is 0-30 cm below the surface (Renger et al., 2002).

Adding fresh organic matter (FOM) can be effective at reducing CH_4 emissions (see Chapter 3). However, the addition of FOM is associated with CO_2 emissions due to the addition of labile C that is readily broken down by soil microbes. Studies, such as Fontaine et al. (2003) and Wang et al. (2015) have shown that adding labile C can lead to the breakdown of the previously stable soil organic matter (SOM) as the new C stimulates microbial activity and exoenzyme production, a priming effect (PE). Nevertheless, the quality of the added C is also important. Higher quality and more labile FOM amendments are preferred by soil microbial communities which can cause a prolonged PE on existing SOM (Wang et al., 2015).

Another mitigation measure with the potential to suppress CH_4 in cultivated peatlands is sulphate (SO_4^{2-}). There is a precedent in marine sediments which are anoxic but are not prolific CH_4 producers. In marine sediment, SO_4^{2-} reduction is the most important anaerobic degradation pathway for organic matter. SO_4^{2-} reducing microbes outcompete the methanogenic microbes for labile substrate, therefore leading to a decline in CH_4 emissions (Pester et al., 2012). Calcium sulphate ($CaSO_4$) applied on rice grown on wetlands has been shown to lower the emissions of CH_4 (Lindau et al., 1994). A study by Han & Tokunaga (2014) on mineral soil showed that the addition of $CaSO_4$ can lead to increased CO_2 consumption; the consumption was due to the suppression of microbial oxidation of soil organic carbon (SOC) by increased calcium (Ca^{2+}) and SO_4^{2-} from the dissolution of $CaSO_4$ minerals. In natural wetlands SO_4^{2-} can lower CH_4 emissions by as much as 40% (Gauci et al., 2005; Dise & Verry, 2001). The mechanisms involved should be the same in cultivated peatlands, thus the application of SO_4^{2-} could help with the lowering of CH_4 emissions that will result from an elevated water table that will bring anoxic conditions closer to the surface. Iron (II) chloride ($FeCl_2$) has also been shown to suppress the production of CH_4. In aerobic soils, reduced Fe rapidly oxidises leading to the precipitation of ferrihydrite. This can bind labile SOM and exoenzymes reducing SOM turnover. The addition of $FeCl_2$ in soils that had slurry applied to them resulted in a 99% decrease in CH_4 fluxes (Brennan et al., 2015). In rice paddy studies, the addition of $FeCl_2$ was reported to have decreased the production of CH_4 by over 50% (Hu et al., 2020).

It is important to note that the addition of these amendments could have a negative impact on crop production. Therefore, even if these amendments are effective against GHG fluxes in cultivated peatlands, their impact on crop production needs to be assessed and well understood before they can be implemented on farms. Gypsiferous soils are rich in $CaSO_4$ but are infertile

even when fertilised as $CaSO_4$ affects nutrient solubility. However, a study by Elrashidi et al. (2010) demonstrated that the addition of peat to gypsiferous soil improved nutrient solubility and therefore improved soil fertility for crop production. Based on the findings of Elrashidi et al. (2010), the expectation is that these drained peatlands will not have nutrient solubility issues when $CaSO_4$ has been added. $CaSO_4$ is commonly used to raise the pH of highly acidic soils (Rocha et al., 2014). Unlike bogs (raised and blanket) where acidic conditions prevail because they get most of the water from rain (ombrotrophic), the peatlands in The Fens have a high pH as they get most of their water from mineral-rich groundwater (minerotrophic) (Griffiths et al., 2019; Pedrotti et al., 2014).

This study assesses the effectiveness of $CaSO_4$, $FeCl_2$ and $FeSO_4$ in suppressing CO_2 and CH_4 from cultivated peat soils and their potential as amendments for GHG mitigation. This will be explored at contrasting water table depths, reflecting different drainage and rewetting conditions. The addition of $CaSO_4$, $FeCl_2$, and $FeSO_4$ to peat soils will undoubtedly influence the pH status of the soil. Effects on the pH level of the soil will be assessed throughout the study to understand how the pH responds both at the initial application of treatment and during the incubation period. Furthermore, this study assesses how the effects of $CaSO_4$ and $FeSO_4$ will impact the pH and DOC status of the soil.

The hypotheses for this study are that: i) a water table closer to the surface, between 0 cm and 20 cm, will suppress CO_2 emissions but lead to increased CH_4 emissions compared to the 50 cm water table due to the anoxic conditions being brought closer to the surface, ii) amendment with $CaSO_4$ and $FeSO_4$ will lead to a suppression of both CH_4 and CO_2 fluxes, iii) $FeCl_2$ will increase CO_2 fluxes but will lead to the consumption of CH_4, and iv) the addition of all amendments will suppress DOC leaching through the effects of pH and ionic strength on the pore water thus leading to reduced organic matter solubility. The results of this study will provide evidence of contrasting amendments that may help reduce the overall GHG burden associated with horticultural peatlands, helping to pave the way for more sustainable agricultural production methods. This will be important as the agricultural sector aims to become carbon neutral by 2050.

5.3 Materials and methods

5.3.1 Study site

5.3.1.1 Location of the sampling site and experiment location

This study consisted of two experiments. The experimental work was conducted at Bangor University (Grid Reference: SH 57770 71959, UTM: 30U 424429, 5897963). Both experiments used PVC cores containing peat soil that was collected from Rosedene Farm.

5.3.2 The first experiment (experiment 3a)

The first experiment (experiment 3a) analysed in this chapter used intact cores that preserved the field soil structure and characteristics. The cores were collected on the 16th of July 2021 and transported to Bangor University for the mesocosm study. To collect the intact cores, the PVC pipes were driven into the soil covering the whole pipe before being excavated. Each core was sealed at the bottom using a 160 mm PVC coupler and cap to ensure no water or soil loss occurred from the bottom of the core. At the time of soil sample collection, the field had not been cultivated for that year's growing season nor had it been cultivated the previous season. Therefore, at the time of sample collection, the field was covered in weeds (Figure 5.2), such as nettles (*Urtica dioica* L.), thistle (*Cirsium vulgare* (Savi.) Ten.), burdock (*Arctium minus* (Hill.) Bernh.), common ryegrass (*Lolium perenne* L.), wormwood (*Artemisia vulgaris* L.), prickly sow-thistle (*Sonchus asper* (L.) Hill.), dock leaf (*Rumex obtusifolius* L.), and cow parsley (*Anthriscus sylvestris* (L.) Hoffm.). At the time of sampling, the water level in the drainage dykes was more than 2 m below the surface of the soil.

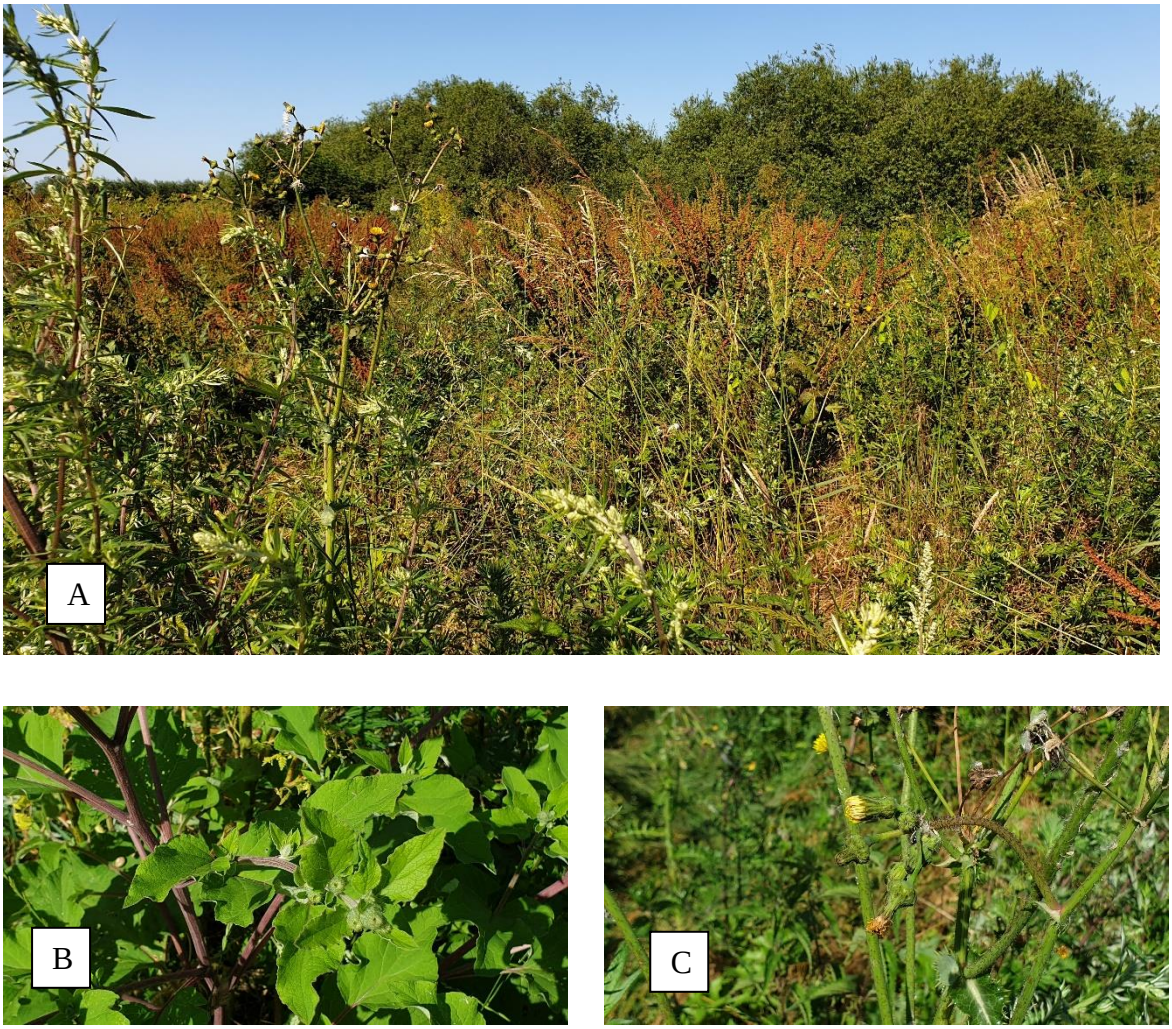


Figure 5.2 The field where the soil samples were collected for the experimental work was completely covered in weeds at the time of soil sample collection (A). The weeds included species such as burdock (Arctium minus (Hill.) Bernh.) (B) and prickly sow-thistle (Sonchus asper (L.) Hill.) (C).

5.3.2.1 Experiment design

5.3.2.2 Treatments and the quantities of treatments

The experiment was split into two water table treatment regimes. 16 of the cores had a water table depth of 20 cm below the soil surface and the other 16 had a water table depth of 50 cm below the soil surface. Each water table treatment had four treatments of i) calcium sulphate ($CaSO_4$), ii) iron chloride ($FeCl_2$), (iii) iron sulphate ($FeSO_4$) and, iv) untreated control (see Figure 5.3). This study did not include any vegetation cover and the soil was left fallow

throughout the study. Any weeds or algae that grew in the cores during the experiment were manually removed without disturbing the soil.

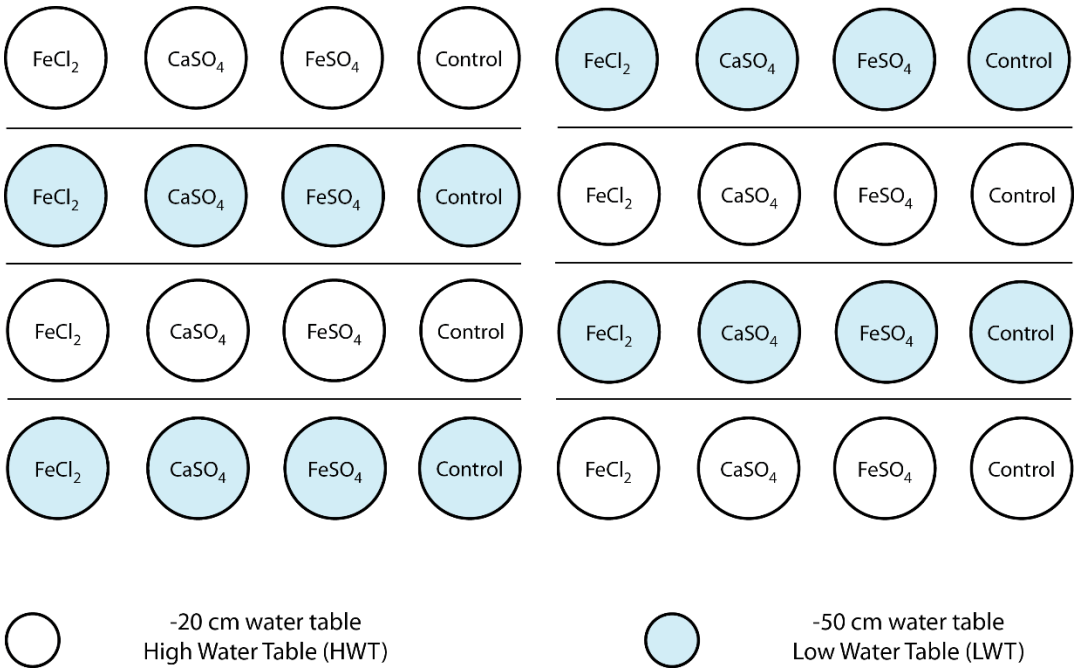


Figure 5.3 The layout of the experiment, 16 cores in both 20 cm water depth and 50 cm. Each water table treatment has four replicates of the amendment treatment of iron (II) chloride ($FeCl_2$), iron (II) sulphate ($FeSO_4$), calcium sulphate ($CaSO_4$), and the control which had no amendment.

Studies that have used iron sulphate as an amendment such as Fresno et al. (2020) have added it as a percentage of the soil volume. In this study, the amendments were added on a per hectare basis. Lindau et al. (1994) applied different quantities of $CaSO_4$ in their study, 0, 1000 and 2000 kg ha^{-1}. However, it was considered that these quantities might not be enough to trigger the desired effect, so an equivalent of 4000 kg ha^{-1} was applied in this study. The per hectare equivalent in the cores was 8 g per core. Lower quantities are possible in a mesocosm as the applied amendments are normally contained in the cores and therefore will not diffuse away from the mesocosms. However, as some of the cores, i.e., the cores in the 50 cm water table drained away excess water gained through precipitation, it was necessary to have a higher amount of amendment. The $CaSO_4$ (calcium sulphate 2-hydrate) was applied in powdered form while the $FeCl_2$ (iron (ii) chloride 4-hydrate) and $FeSO_4$ (iron sulphate 7-hydrate) were applied

in granular form. The amendments were applied to the top 20 cm of all the cores above the water table boundary.

Figure 5.4 The amendments added to the cores on the 16th of September 2021 were A) calcium sulphate ($CaSO_4$) in powdered form, B) iron chloride ($FeCl_2$) in granular form and, C) iron sulphate ($FeSO_4$) in granular form. 8 grams of each amendment was added.

5.3.2.3 Specifications of the cores

This experiment was made up of 32 PVC soil cores measuring 600 mm in height and 160 mm in diameter. The 600 mm core height was chosen as that provided a significant amount of soil below ground to replicate field conditions as closely as possible. The surface area of a single core was 0.000002 ha or 0.02 m^2. The individual cores were placed in 90 litre plastic bins (Figure 5.5) which were used to maintain the water table of the cores to their relevant water table depth (Figure 5.5 and Figure 5.6). The water provided insulation to the cores, so the soil temperature was not immediately affected by atmospheric temperature fluctuations. The 20 cm cores had a hole drilled 20 cm from the top, in this hole a 6.4 mm Bev-A-Line tubing was inserted, and this tube passed through a hole in the plastic bin to allow excess water to drain out of the cores. Furthermore, two holes were drilled 5 mm below the hole at 20 cm to allow the cores to fill up with water from the bins. The 50 cm cores each had a hole drilled at 50 cm from the top. Compression fittings were fitted into the holes, then a 6.4 mm Bev-A-Line tubing was fitted. The tubing then exited the water bins through a hole at the same height. The water table in the 50 cm cores was maintained by watering from the top. Any excess water was washed out through the Bev-A-Line tubing at 50 cm, therefore, maintaining the required water table depth.

Figure 5.5 (Left) The 50 cm water table cores showing the 6.4 mm Bev-A-Line tubing connected using a compression fitting. (Right) The 50 cm and 20 cm water table cores in the 90 litre bins with the water table management Bev-A-Line tubing protruding from the buckets.

5.3.2.4 Rhizon samplers for pore water analysis

Individual Rhizon samplers were inserted into each core, Figure 5.6. The samplers were inserted 10 cm from the top of the core. The Rhizon samplers were used to extract soil pore water samples for pH and DOC analysis. Rhizon samplers are non-destructive tools that can be used in-situ to extract soil solution without changing the composition or structure of the soil during extraction (Amoakwah et al., 2013; Seeberg-Elverfeldt et al., 2005).

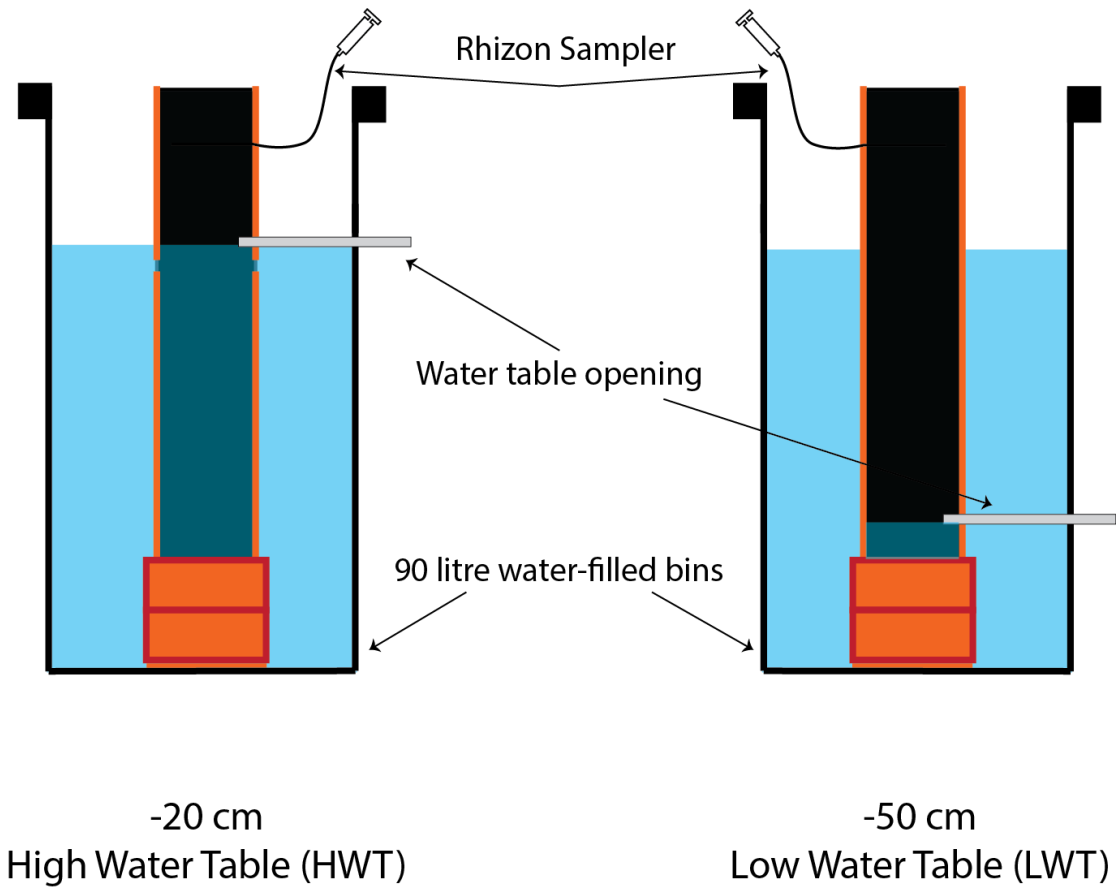


Figure 5.6 Set up of intact peat soil mesocosms, in 90 litre water-filled bins. Left shows the 20 cm water table core with a Bev-A-Line tubing at 20 cm from the top of the core to allow water to drain out to maintain the required 20 cm water table and 5 mm from the tubing are two holes that allowed the cores to be filled with water from the bins. The 50 cm core did not have an opening into the water bins, but they had Bev-A-Line tubing at 50 cm from the top of the core to draw out excess water thus maintaining the water table. Both cores were placed into bins that were filled to the same water level to provide insulation to the cores. Rhizon samplers were inserted 10 cm from the top of the core.

5.3.2.5 pH and electrical conductivity (EC) analysis

pH in the soil pore water that was collected using the Rhizon samplers was analysed at the end of the experiment. The soil pore water was stored in a freezer for preservation. The pH analysis was conducted using a Metrohm 888 Titrando. At the same time as pH analysis, the soil pore water samples had their electrical conductivity (EC) determined using a Jenway Conductivity Meter 4320.

5.3.2.6 Dissolved organic carbon (DOC)

After the pH and EC analysis, the water samples had their dissolved organic carbon (DOC) content analysed using the Analytical Sciences Limited Thermalox. The analyser measured non-purgeable organic carbon (NPOC). The soil pore water samples were first acidified with 45µl 0.5M hydrochloric acid (HCl) and purged with oxygen (O_2) for 90 seconds. After acidification, total carbon (TC) analysis was conducted by injecting aliquots of the samples into a reactor containing platinum oxide at 680 °C with a flow of pure O_2 gas. This combusts and oxidises all carbon species into CO_2 gas which is then detected by a nondispersive infrared (NDIR) CO_2 detector.

5.3.2.7 Soil characteristics

This study used soil samples identical to Wen et al. (2019) from an identical field at Rosedene Farm. The soil is classified as Earthy Sapric Fen Soils under the UK soil taxonomy or, under the US soil taxonomy, they are a Typic Haplosaprist with a humification score of H9 on the von Post scale (Wen et al., 2019). The characteristics of the soil are summarised below (Table 5.1).

Table 5.1 Soil Characteristics for the lowland organic cultivated soil.

	Bulk density ($g\ cm^{-3}$)	**Organic matter (%)**	**Total C (%)**	**Total N (%)**	**C: N**	**pH (H_2O)**	**EC (µS/cm)**
0-10 cm	0.32 ± 0.03a	78.5 ± 0.5b	50.7 ± 0.4b	2.71 ± 0.04a	18.7 ± 0.2b	6.45 ± 0.10	394 ± 120
10-30 cm	0.31 ± 0.01a	78.7 ± 0.7b	50.5 ± 0.3b	2.71 ± 0.03a	18.6 ± 0.1b	6.29 ± 0.08	558 ± 119
30-50 cm	0.22 ± 0.02b	84.1 ± 1.2a	54.8 ± 0.7a	2.45 ± 0.07b	22.4 ± 0.9a	6.09 ± 0.11	377 ± 30

Abbreviation: EC, electrical conductivity.

All values mean ± standard errors (n = 4). Different letters indicate statistically significant differences, after a one-way (Analysis of Variance) ANOVA test with Tukey's pot-hoc test at $p \leq .05$).

5.3.2.8 Experiment timeline

The experiment was conducted for 10 weeks from the 17th of August 2021 to the 21st of October 2021. From the 17th of August to the 2nd of September, baseline GHG flux readings were observed without the alteration of the water table or added amendments. Then from the 2nd of September until the end of the study all the water tables were altered. The amendments of $CaSO_4$, $FeCl_2$ and $FeSO_4$ were added on the 14th of September.

5.3.2.9 Rate of data sampling and gas analyser frequency

The GHG flux measurements were recorded twice a week on a Tuesday and a Thursday. A closed chamber covered the cores for 2 minutes to measure the flux on a Benchtop Los Gator Research (LGR) Greenhouse Gas Analyser (Model: 908-0010-0002). There was a gap of one-minute in-between measurements to allow the fluxes to return to ambient level values. The fluxes were read by the LGR analyser at a frequency of 1 Hz. Vacuum tubes were inserted once a week on a Friday to collect soil pore water for pH and DOC analysis in the laboratory.

5.3.2.10 Rainfall measurement

The amount of rainfall that fell during the experiment was measured using a rain gauge from the time when the amendments of $FeCl_2$, $CaSO_4$, and $FeSO_4$ were added on the 16th of September 2021. The measurements were taken for the days between the flux readings.

5.3.3 The second experiment (experiment 3b)

While experiment 3a used intact cores, this experiment used soil that was filled into the cores and bulked together. The difference is that the intact cores are representative of the soil profile while the bulked-up soil only contains mixed soil from the top 40-50 cm. The soil was collected on the 10th of November 2021; it was collected from an adjacent field to the one used in the experiment 3a. At the time of soil collection, no crops were growing in the field. The previous crop grown in the field was radish (*Raphanus sativus* L.), which was harvested during the summer prior to the soil being collected. There were almost no weeds in the field, only seedlings of stinging nettles were identifiable. The cores used in this experiment had the same specifications as the previous experiment.

5.3.3.1 Experiment design

A total of 16 cores were collected for this experiment. Unlike experiment 3a, this experiment only had one water table treatment of 0 cm (fully flooded/wet). Similarly, this study did not include any plants. In the fourth week of the experiment, potato (*Solanum tuberosum* L.) residue was added as a substrate to the cores to trigger methanogenesis. The experiment was initially not planned to have the addition of any substrate. However, as the GHG fluxes from the cores responded in a comparable way to experiment 3a, it was clear that just having a 0 cm water table was not enough to trigger the methanogens to produce CH_4 as there was not enough labile carbon. Potato was chosen as a labile substrate because it is one of the main crops grown in The Fens. Furthermore, it has a high soluble carbohydrate content which is ideal for methanogenesis (Parawira et al., 2008). All the cores were applied with the residue because there were not enough cores to have a no-potato residue control.

5.3.4 Water table management

The water table for this experiment was managed in a comparable way to experiment 3a. However, in these cores, a hole was drilled 10 mm from the top of the cores. The buckets were then filled with water as close to the top of the core as possible. The hole which was 10 mm from the top of the core was submerged under the water to allow the water to fill into the core, therefore flooding it to allow the water table to maintain at 0 cm.

5.3.5 Experiment timeline

The experiment was conducted from the 23rd of November 2021 to the 4th of February 2021. In the first week of the experiment, baseline GHG fluxes were measured before any amendment or water table alteration. The water table was raised to 0 cm on the 29th of November 2021. On the 13th of December 2021, potato residue was added to stimulate CH_4 fluxes.

5.3.6 Rate of data sampling and gas analyser frequency

The GHG measurements in this experiment were measured twice a week, on a Monday and Wednesday. A closed chamber was put on each of the cores for 2 minutes to measure the flux on the LGR analyser. A gap of at least one minute was allowed between the cores to allow the chamber to purge and return the fluxes to ambient levels. Unlike during the first experiment discussed in this chapter, this experiment did not use Rhizon samplers to collect soil pore water.

5.3.7 Methane (CH_4) and carbon dioxide (CO_2) analysis

The GHG fluxes were recorded using the same Benchtop Los Gatos Research (LGR) Greenhouse Gas Analyser (Model: 908-0010-0002) with a closed chamber system used in the first experiment. The experiments in this study focused on the GHG emissions of CH_4, and CO_2. There was no addition of nitrogen (N) rich fertiliser, therefore, no elevated N_2O pulses were expected, so N_2O fluxes were not measured. Effectively, the addition of N based fertiliser can lead to N_2O fluxes as the N is mineralised in oxic peat soil (Minkkinen et al., 2020). In terrestrial ecosystems, N_2O production is often constrained by inorganic N availability (Dijkstra et al., 2012).

5.3.7.1 Static closed chamber

The custom-made chamber was made from the same brown PVC used for the cores; it had a height of 475 mm and a diameter of 160 mm. The static chamber was connected to the LGR analyser by an 11 m outlet gas line and an 11 m return gas. There was a vent fitted at the top of the chamber to allow pressure release from the chamber when it was inserted into a core for flux measurements. The vent was designed and positioned in a way that eliminated the need for a fan, and it was not affected by external air movement or turbulence.

5.3.8 Soil and air temperature

During the experiment, soil and air temperatures were recorded at the time of sampling each core. The air temperature was recorded using a Criacr WA10-F Digital Hygrometer Thermometer. The WA10-F reads up to 1 decimal point (DP) and it has a temperature range of -40 °C to 60 °C with an accuracy of ± 1°C. The soil temperature was recorded using a JAOK Soil Tester which can read temperatures ranging from -9 °C to 50 °C. The JAOK tester uses a thermocouple to read the temperature. However, it is not stated in its specifications what type of thermocouple it is. The soil temperature was measured to a depth of 20 cm.

5.3.9 Statistical analysis and data graphs

All statistical analyses and graphs in this study were produced in RStudio version 1.4.1717 and R version 4.1.1 (2021-08-10). Since the fluxes data was collected from the cores repeatedly, the statistical calculation was conducted using a Repeated Measures ANOVA (Analysis of Variance) technique. A mixed-effects model was chosen because the same cores were measured continuously throughout the experiment therefore they would have been a problem

with pseudoreplication. Pseudoreplication is a problem because repeated measures on the same core will mean there is no independence of errors which is an important assumption of standard statistical analysis (Crawley, 2013). The lmerTest package was used to evaluate the linear model of the data to calculate statistical significance. No transformation was done on the data in this experiment. Before the data were analysed in R/RStudio, it was processed and collated in Microsoft Excel then saved in flat formats suitable for working in R/RStudio.

5.4 Results

5.4.1 Results for the first experiment

5.4.1.1 Rainfall during the experiment

2 mm of rain was measured the night before the amendments were added. Table 5.2 summarises the rainfall from the 16th of September 2021 until the end of the experiment on the 14th of October 2021.

Table 5.2 Total rainfall for the periods in between flux readings after the application of iron (II) chloride ($FeCl_2$), iron (II) sulphate ($FeSO_4$) and calcium sulphate ($CaSO_4$).

Observation day	Amount of rainfall (mm)
10	2
11	7
12	0.5
13	45
14	0
15	55
16	5
17	0.9
18	2
Total rainfall	**117.4**

5.4.1.2 Carbon dioxide (CO_2) fluxes

The average carbon dioxide fluxes are shown for each observation day throughout the experiment in Figure 5.7. All the cores were producing CO_2 but the fluxes in the 50 cm water table cores were higher than in the 20 cm water table cores. The addition of the amendments led to a spike in CO_2 emissions in the $FeCl_2$ and $FeSO_4$ treatments, but no such spike was observed in the $CaSO_4$ treatments.

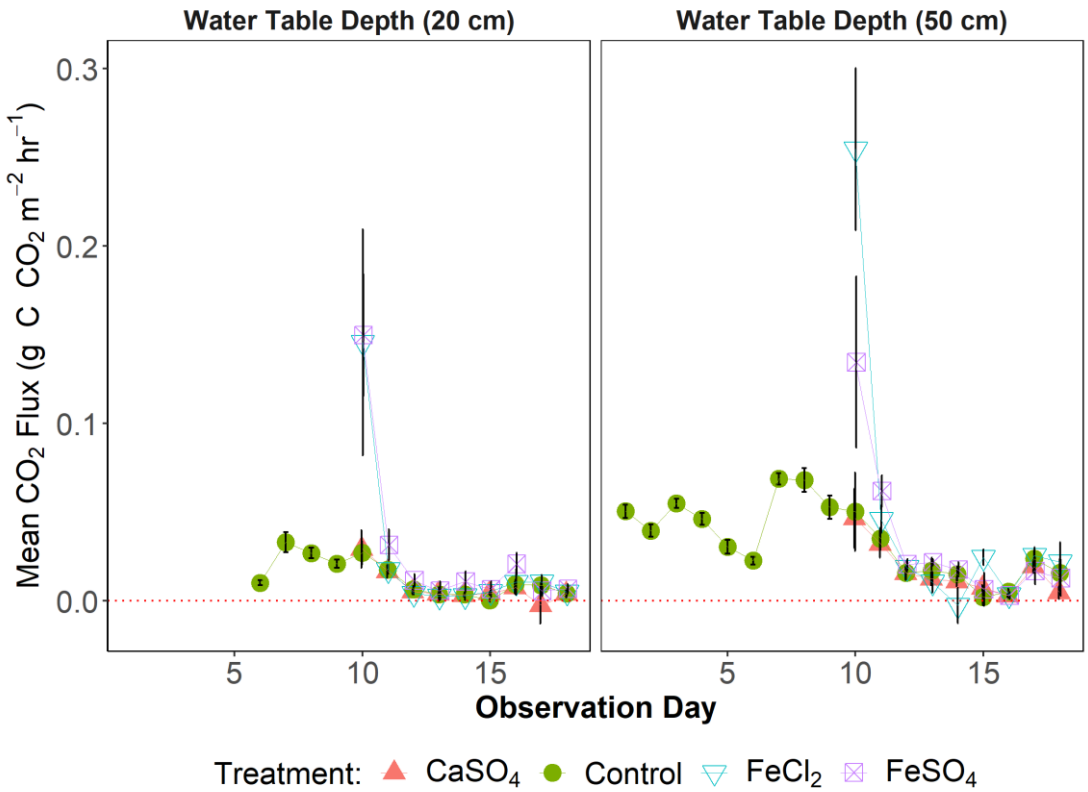


Figure 5.7 Daily CO_2 fluxes from the lowland peat study for the duration of the experiment assessing the effect of iron (II) chloride ($FeCl_2$), iron (II) sulphate ($FeSO_4$) and calcium sulphate ($CaSO_4$). Values represent means ± SEM (n = 4) .

5.4.1.2.1 Cumulative carbon dioxide (CO_2) fluxes

The cumulative sums for all the GHG fluxes during the experiment are summarised in Table 5.3. The fluxes of CO_2 resulting from the application $CaSO_4$ (Table 5.3) were lower than in the control treatment but this was not statistically significant ($p > 0.05$). While the fluxes resulting from the application of $FeCl_2$ and $FeSO_4$ were higher than the control treatments and this effect was statistically significant ($FeCl_2$, $p < 0.001$; $FeSO_4$, $p < 0.001$). The CO_2 fluxes in the 20 cm water table were lower than in the 50 cm cores for the control ($p < 0.001$). The fluxes resulting from the application of $FeCl_2$ were significantly higher in the 50 cm water table than in the 20 cm water table ($p < 0.001$). However, the fluxes resulting from the application of $FeSO_4$ and $CaSO_4$ did not differ significantly in the different water tables ($FeSO_4$, ($p > 0.005$); $CaSO_4$, ($p > 0.05$).

Table 5.3 Total cumulative carbon dioxide (CO_2) fluxes from observation day ten when the amendments were added. Values represent means ± SEM (n = 4).

Water Table (cm)	**Treatment**	**Cumulative CO_2 (g C -CO_2 m^{-2})**	**CO_2 SEM**	**Cumulative CH_4 (mg C-CH_4 m^{-2})**	**CH_4 SEM**
20	$CaSO_4$	0.070	0.033	-0.024	0.013
20	Control	0.081	0.023	-0.020	0.009
20	$FeCl_2$	0.204	0.085	-0.014	0.006
20	$FeSO_4$	0.249	0.073	-0.034	0.024
50	$CaSO_4$	0.151	0.044	-0.016	0.010
50	Control	0.180	0.061	-0.021	0.011
50	$FeCl_2$	0.403	0.094	-0.020	0.009
50	$FeSO_4$	0.295	0.094	-0.018	0.018

5.4.1.2.2 Fluxes after adding amendments vs long term effect

Shortly after adding the amendments, there was a spike in the fluxes of CO_2 (Figure 5.7). The fluxes from the $FeSO_4$ 50 cm water table treatment reached 0.134 ± 0.048 s.e. g CO_2 m^{-2} hr^{-1} and 0.150 ± 0.034 s.e. g CO_2 m^{-2} hr^{-1} in the 20 cm water table. The fluxes of $FeCl_2$ reached 0.255 ± 0.046 s.e. g CO_2 m^{-2} hr^{-1} in the 50 cm water table and 0.146 ± 0.064 s.e. g CO_2 m^{-2} hr^{-1} in the 20 cm water table. However, this was short-lived (7 days), and the amendments did not appear to have any long-term impact on the fluxes. After the initial spike, the average fluxes in the $FeSO_4$ (50 cm water table) were 0.014 ± 0.002 s.e. g CO_2 m^{-2} hr^{-1} and in the 20 cm water table were 0.010 ± 0.002 s.e. g CO_2 m^{-2} hr^{-1}. The fluxes of $FeCl_2$ were 0.015 ± 0.003 s.e. g CO_2 m^{-2} hr^{-1} in the 50 cm water table and 0.006 ± 0.001 s.e. g CO_2 m^{-2} hr^{-1} in the 20 cm water table.

5.4.1.3 Methane (CH_4) fluxes

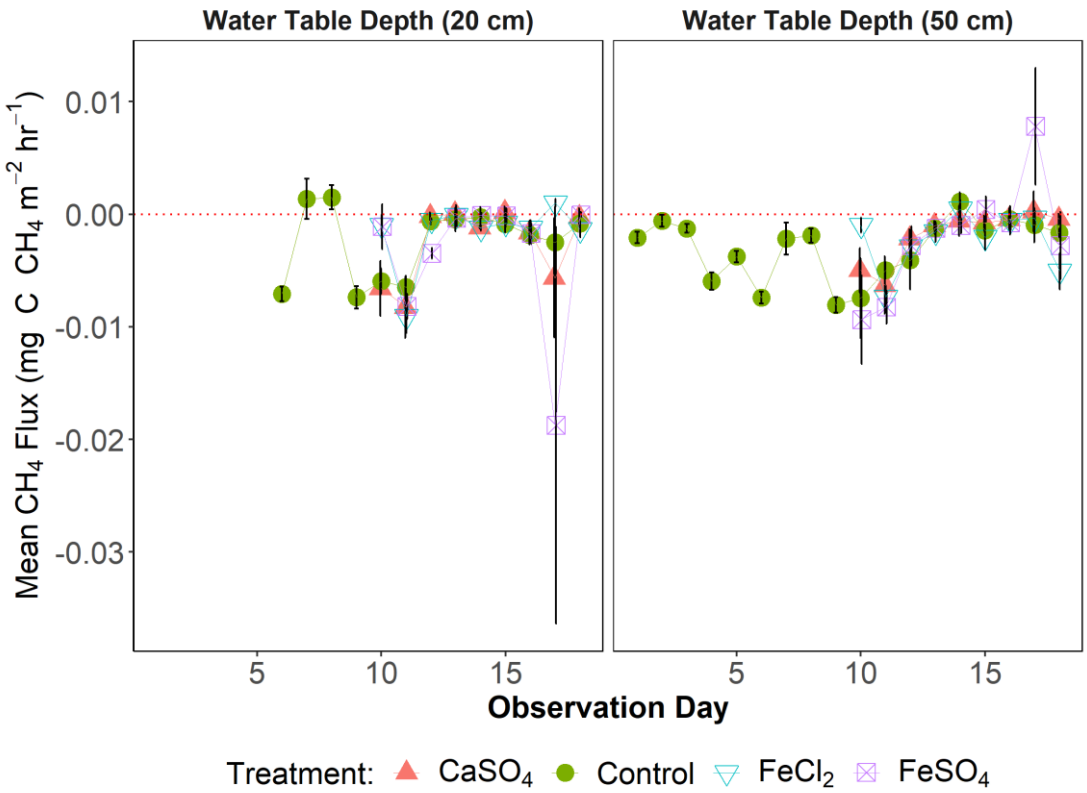


Figure 5.8 Daily methane (CH_4) fluxes for the duration of the study. Values represent means ± SEM (n = 4).

5.4.1.3.1 Cumulative methane (CH_4) fluxes

During this study, there was consumption of CH_4 in both the treated and control cores. The amendments were expected to affect the fluxes of CH_4 but as the cores were consuming CH_4, it was difficult to assess this. The cumulative CH_4 fluxes are summarised in Table 5.3. The effect of all the amendments and the control did not have any statistically significant differences ($CaSO_4$, $p = 0.064$; $FeCl_2$, $p = 0.315$; $FeSO_4$, $p = 0.60$). Similarly, the effect of water table depth did not produce fluxes that were statistically distinct from each other ($p = 0.185$). The only difference in CH_4 fluxes that was statistically significant was the fluxes in cores treated with $CaSO_4$ between the 20 cm and 50 cm water table.

5.4.1.4 pH

The pH status of the sampled pore water was influenced by the addition of the amendments as opposed to the water table depth. Except for the control cores where the water table influenced changes in pH, all the cores with amendments added had a lower pH compared to the control cores. This effect was the same in both the 20 cm and 50 cm water table depth (Figure 5.9). In the 20 cm water table, the effect of the added amendments was significantly different from the control cores ($p < 0.05$). However, in the 50 cm water table, the difference in pH for the cores treated with $FeCl_2$, $FeSO_4$ and $CaSO_4$ was not different statistically from the control cores ($p > 0.005$). The pH for the control cores in the 50 cm water table depth was lower than that in the 20 cm water table depth throughout the experiment and this difference was statistically significant ($p < 0.05$). Owing to the reduced pH for the control cores in the 50 cm water table treatment, the differences in pH for the $FeCl_2$, $CaSO_4$ and $FeSO_4$ between the water table treatments were not statistically significant ($p > 0.05$).

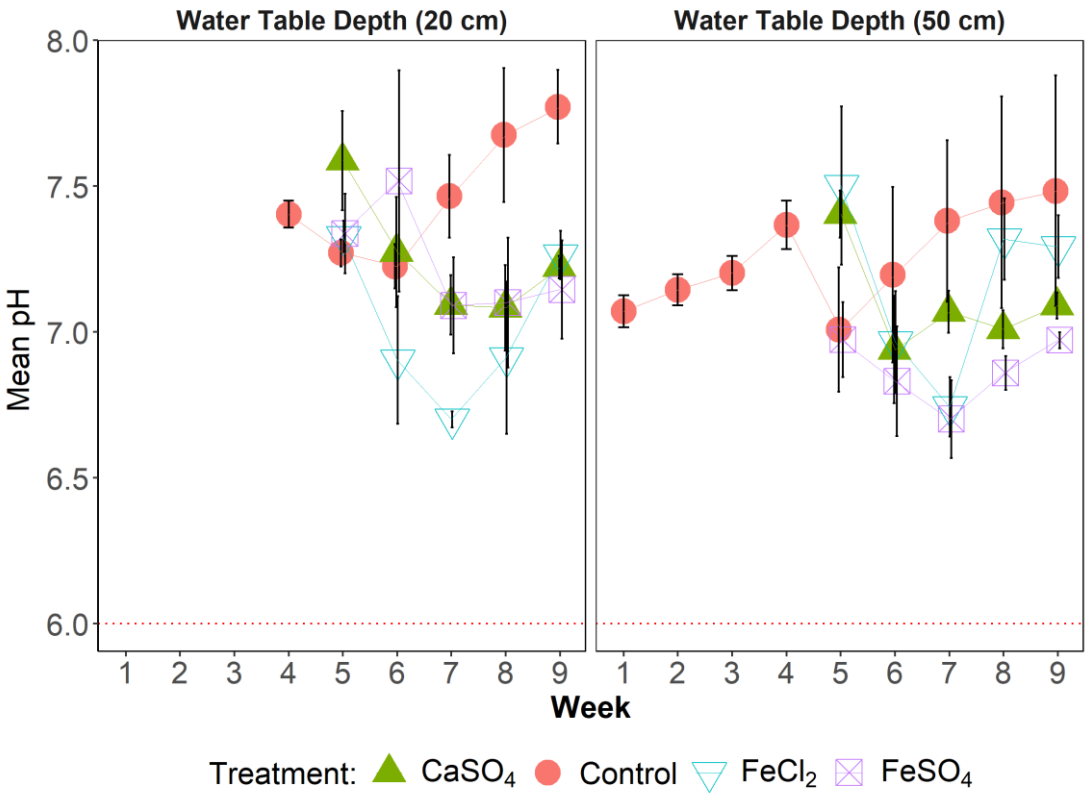


Figure 5.9 Average weekly pH for all the cores during the study into the effects of the amendments of iron (II) chloride ($FeCl_2$), iron (II) sulphate ($FeSO_4$) and calcium sulphate ($CaSO_4$). Values represent means ± SEM (n = 4).

5.4.1.5 Electrical conductivity (EC)

The application of the $FeCl_2$ led to the electrical conductivity (EC) of the sampled soil pore water spiking, Figure 5.10. This spike was more pronounced in the 20 cm water table treatments than the 50 cm. The EC reduced to lower levels after the initial spike. The amendments of $FeSO_4$ and $CaSO_4$ did not have a large spike. However, the EC in the cores with the amendments was higher than that of the control cores.

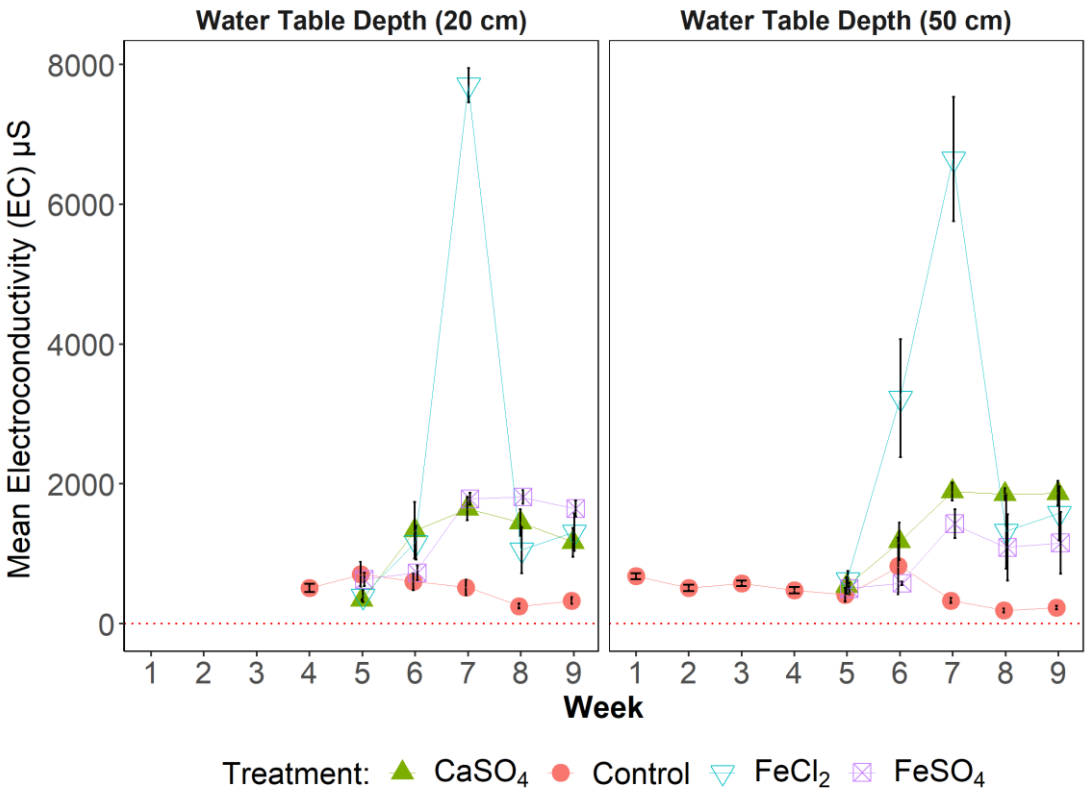


Figure 5.10 Average weekly electrical conductivity (EC) averages for the treatments of iron (II) chloride ($FeCl_2$), iron (II) sulphate ($FeSO_4$), calcium sulphate ($CaSO_4$), and no treatment control, in the lowland peatland experiment. Values represent means ± SEM (n = 4).

5.4.1.6 Dissolved organic carbon (DOC)

The amount of DOC measured as non-purgeable organic carbon (NPOC) in the experiment was higher in the control cores than in those with amendments of $FeSO_4$, $FeCl_2$, and $CaSO_4$. Initially, there was a spike in the DOC when the amendments were added to the soil in week 5 but after that, the DOC remained lower than the control. Statistical analysis shows that the difference in the DOC of the amendments was significantly different from the control cores ($CaSO_4$, $p < 0.001$; $FeCl_2$, $p = 0.02$; $FeSO_4$, $p = 0.004$). However, the amount of DOC was not statistically different between the 20 cm and 50 cm water table depths ($CaSO_4$, $p = 0.08$; $FeCl_2$, $p = 0.5$; $FeSO_4$, $p = 0.06$; control, $p = 0.1$).

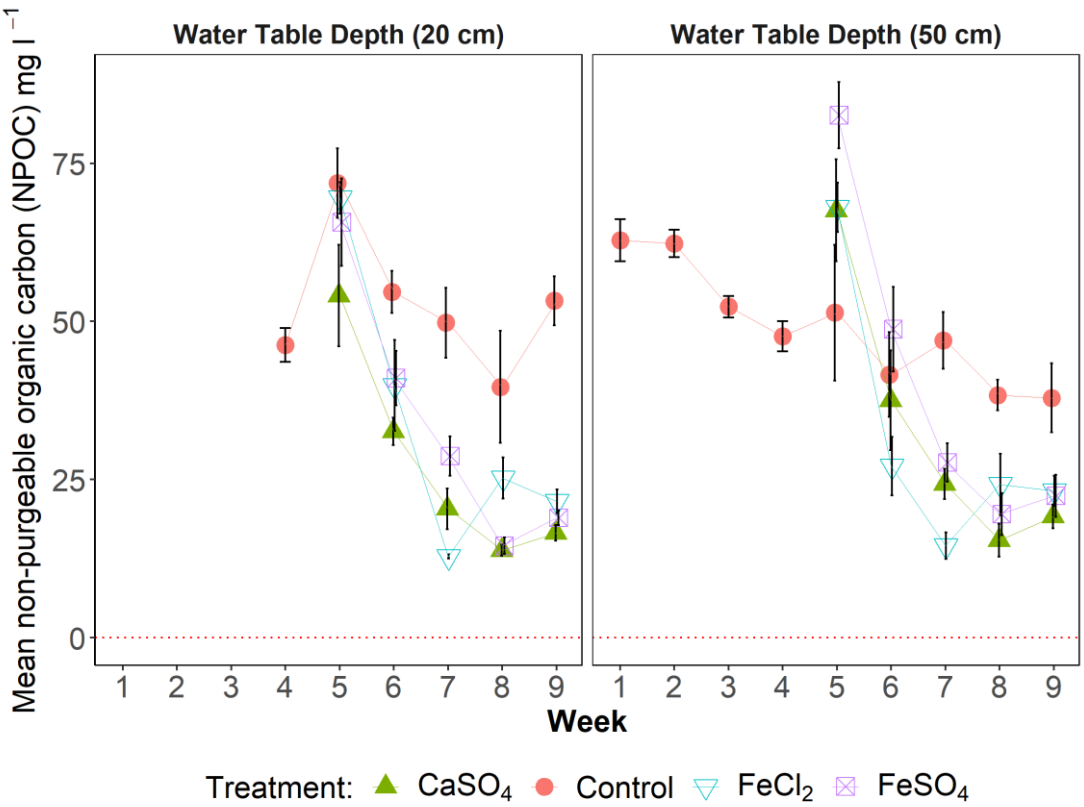


Figure 5.11 Average non-purgeable organic carbon (NPOC) for all the treatments in the lowland peatland experiment to assess the effectiveness of iron (II) chloride ($FeCl_2$), iron (II) sulphate ($FeSO_4$) and calcium sulphate ($CaSO_4$) at reducing the amount of carbon (C) loss from cultivated peatlands under differing water table treatments of 50 cm and 20 cm from the surface. Values represent means ± SEM (n = 4).

The amount of DOC in the soil pore water showed a relationship to the pH status of the soil Figure 5.12. The amount of DOC in the soil was reduced in the lower soil pore water pH.

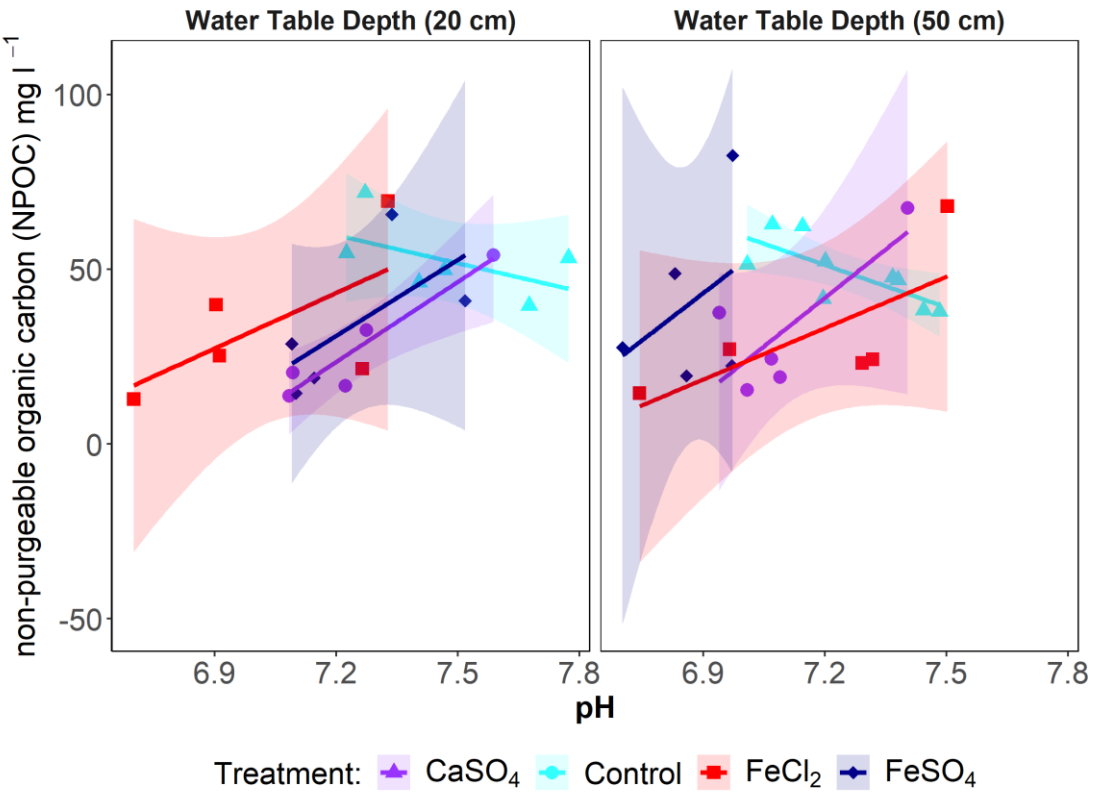


Figure 5.12 The relationship between dissolved organic carbon (DOC) and the pH status of the soil. The concentration of DOC was lower in the low pH environment created by the amendments of iron (II) chloride ($FeCl_2$), iron (II) sulphate ($FeSO_4$) and calcium sulphate ($CaSO_4$). The shading represents the standard error of means.

5.4.2 Results for the second experiment

5.4.2.1 Soil temperature during the follow-up experiment (experiment 3b)

The average soil temperature during the experiment remained above freezing, Figure 5.13. On observation day nine, measurements were conducted at 6 am to find out if there was overnight freezing in the cores which might be influencing CH_4 and CO_2 production. However, all the cores were above freezing albeit colder than as expected during the day.

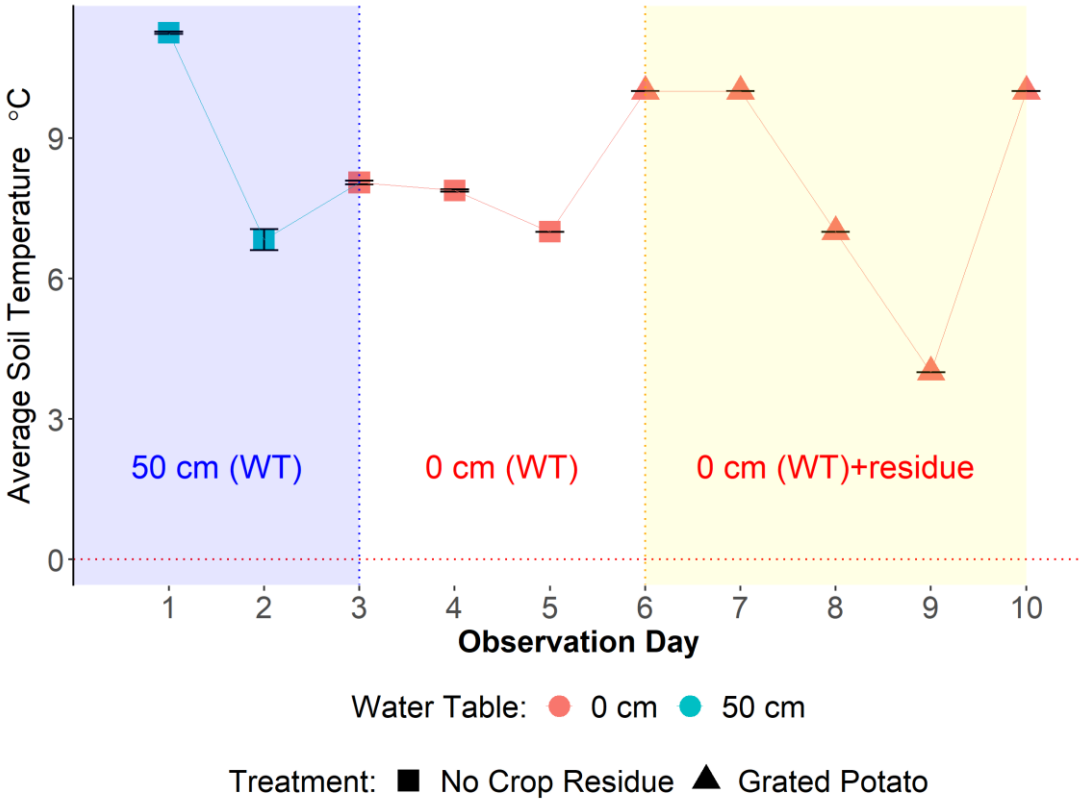


Figure 5.13 Weekly soil temperature (°C) averages for the peatland soil mesocosm experiment. Recorded temperatures were above freezing for all the cores throughout the experiment. Values represent means ± SEM (n = 16).

5.4.2.2 Follow up experiment carbon dioxide (CO_2) fluxes

All the cores produced CO_2 fluxes when the water table was at 50 cm, here shown from observation day 1-2 Figure 5.14. When the water table in the cores was brought up to the surface (0 cm), CO_2 fluxes dropped to nearly zero. On observation day 6, potato residue was added to all the cores, measurements taken soon afterwards showed CO_2 was still being consumed but from observation days 7 to 10, the cores were producing CO_2.

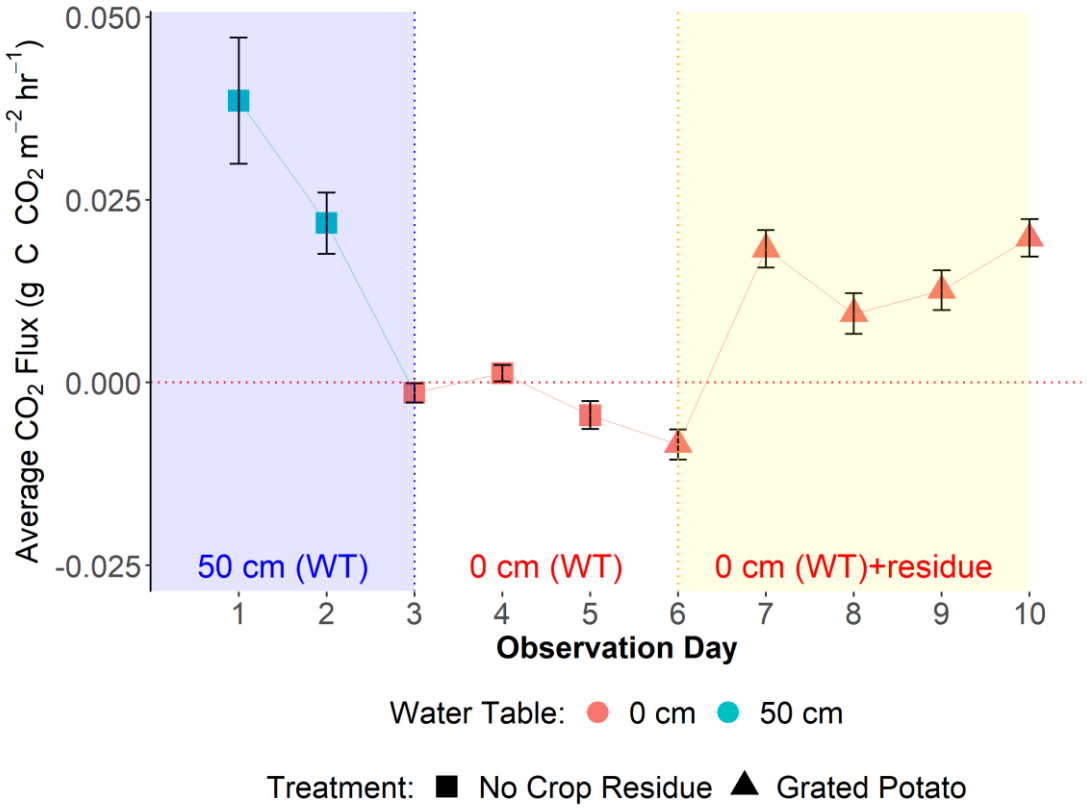


Figure 5.14 Carbon dioxide (CO_2) average flux for each observation day from cultivated peatland soil mesocosms for the follow-up experiment. The water table was altered from the third reading onwards and crop residue potato (Solanum tuberosum L.) was added from the sixth reading. Before the water table was raised, the cores produced CO_2 but became consumers at 0 cm. Crop residue triggered CO_2 fluxes. Values represent means ± SEM (n = 16).

5.4.2.2.1 Follow up experiment cumulative carbon dioxide (CO_2) fluxes

After the water table was raised, the CO_2 fluxes were close to zero. The cores started to consume CO_2, but the addition of crop residue led to the cores releasing CO_2 into the atmosphere, the cumulative total of CO_2 produced is shown in Table 5.4.

Table 5.4 Cumulative carbon dioxide (CO_2) and methane (CH_4) for the cores at 0 cm water table depth after the addition of potato residue (Solanum tuberosum L.), n = 16

Cumulative CO_2 (g C-CO_2 m^{-2})	**CO_2 SEM**	**Cumulative CH_4 (mg C-CH_4 m^{-2})**	**CH_4 SEM**
0.052	0.013	0.053	0.065

5.4.2.3 Second experiment methane (CH_4) fluxes

The cores in the experiment were consuming CH_4 from the beginning of the experiment Figure 5.15. When the water table was raised closer to the surface on observation day three, there was a minor CH_4 flux from the cores. However, during the subsequent observation days 4 and 5, the cores were consuming CH_4. After potato residue was added from observation days 6 to 9, the cores continued to consume CH_4; the cores consumed more CH_4 than when they were at 0 cm and 50 cm without the added residue. Nevertheless, after six weeks of incubation between observation days 9 and 10, the cores finally started producing CH_4 fluxes.

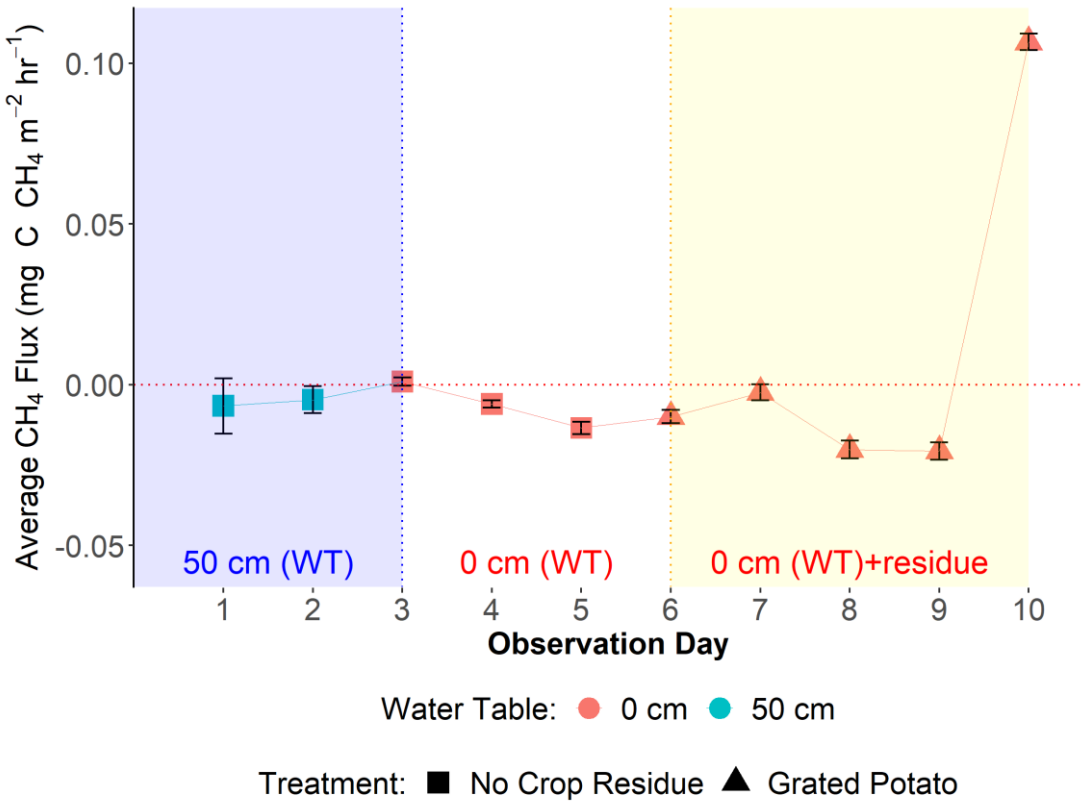


Figure 5.15 Methane (CH_4) flux averages from cultivated peatland soil mesocosms for the follow-up experiment. The water table was altered from the third reading onwards and potato residue (Solanum tuberosum L.) was added from the sixth reading. All the cores were consuming CH_4 except for initial fluxes when the water table was raised and on observation day after a long incubation period. Values represent means ± SEM (n = 4).

5.4.2.3.1 Follow-up experiment cumulative methane (CH_4) fluxes

The consumption of CH_4 was observed from the beginning of the experiment with a 50 cm water table, then when the cores were flooded and continued for weeks after the addition of potato residue (Figure 5.16). On observation day 10, which was 6 weeks after the application of potato residue, there was finally some CH_4 production. However, this varied from core to core as shown by the error bars in Figure 5.16.

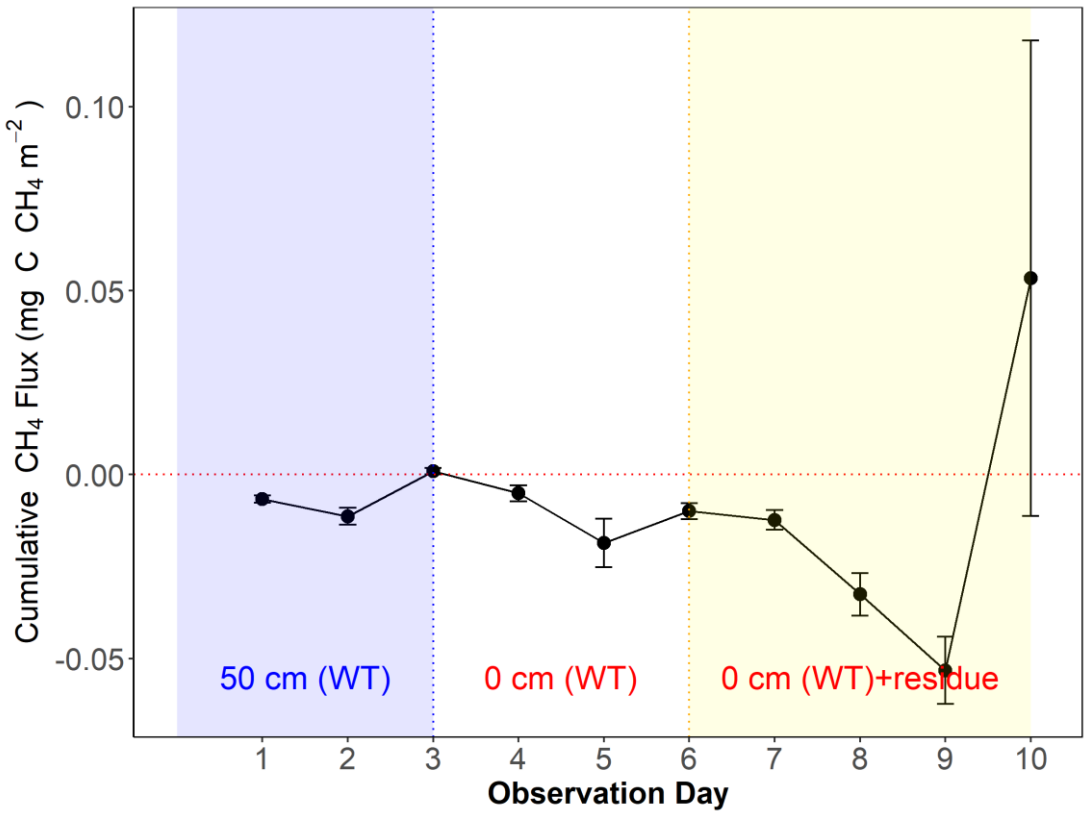


Figure 5.16 Cumulative methane (CH_4) fluxes from the follow-up experiment on lowland cultivated peat from the observation days after potato (Solanum tuberosum L.) was added as a crop residue to trigger CH_4. The addition of potato residue led to increased consumption of CH_4. Incubating the cores for longer between observations 9 and 10 allowed for methanogenesis to occur. Values represent means ± SEM (n = 16).

5.5 Discussion

The results in this study indicate that the use of $FeCl_2$, $FeSO_4$ and $CaSO_4$ as amendments to mitigate GHG flux will have an impact on the fluxes. However, no clear mitigation benefits were demonstrated by the addition of these amendments under the conditions in which the experiment was undertaken (i.e., short incubation period and relatively low water table). All the amendments except $CaSO_4$ triggered more CO_2 flux than in the cores without any treatment. Under the 20 cm water table treatment, the $FeCl_2$ and $FeSO_4$ treated cores produced close to four times more CO_2 than the control cores (Table 5.3). The ferrous iron (Fe^{2+}) was expected to increase CO_2 flux. Through the Fenton reaction, Fe^{2+} together with H_2O_2 is a strong oxidising agent through the formation of hydroxyl radicals (OH) (Winterbourn, 1995; Van Bodegom et al., 2005). These OH radicals play a critical role in the decomposition of lignin via phenol oxidases under oxygen-rich conditions. Van Bodegom et al. (2005) showed that the presence

of Fe^{2+} leads to an increased C mineralisation rate. Fe^{2+} stimulates phenoloxidase activity, therefore leading to the increased decomposition of organic matter (OM). In this study, a spike in CO_2 fluxes was observed soon after the application of the amendments, which was most certainly caused by the oxidation of Fe^{2+} in the $FeSO_4$ and $FeCl_2$. The spike in CO_2 fluxes was more pronounced in the 50 cm water table where most of the soil was under oxic conditions. However, after the initial observations, the level of CO_2 flux in the treatments of $FeSO_4$ and $FeCl_2$ dropped down to the same levels as in the other treatments of $CaSO_4$ and the control cores in both water table treatments. This suggests that this reaction is short-lived, most likely due to all the Fe^{2+} being used up. To counteract this undesired effect, the use of ferric iron (Fe^{3+}) instead of ferrous iron (Fe^{2+}) could reduce C mineralisation because of the ability of Fe^{3+} to coagulate with labile organic matter, suppressing its degradation by microbes and potentially lower CO_2 emissions from the soil (Herndon et al., 2017; Van Bodegom et al., 2005)

The fluxes from the $CaSO_4$ treatment did not differ from the control in the 20 cm cores. The effect of the $FeCl_2$ and $FeSO_4$ amendments were not as pronounced in the 50 cm water table cores with fluxes close to one and a half times more than the baseline. The $CaSO_4$ treatment produced the least amount of CO_2 which was even lower than the control. These findings suggest that only the addition of $CaSO_4$ will have some marginal benefits which are only evident in the 50 cm water table. From this short-term study, no potential benefit in reducing CO_2 and CH_4 flux can be derived from adding the amendments.

Even though Fe^{2+} was expected to increase CO_2 flux, its use was justified due to its potential to suppress the fluxes of CH_4. $FeCl_2$ was expected to suppress CH_4 production. However, in the drained and cultivated peatlands studied, this was not the case as there were no differences in the fluxes of CH_4 owing to the application of $FeCl_2$. Furthermore, $CaSO_4$ and $FeSO_4$ were anticipated to lead to a suppression of both CH_4. However, this was not the case; only $CaSO_4$ in the 20 cm water table had a significant impact suppressing the fluxes of CH_4. The effect of $FeSO_4$ was small and not statistically significant. The soils did not behave as anticipated i.e., there were no CH_4 emissions even in the controls, as the peatlands studied are currently heavily drained for agricultural production.

5.5.1 Methane (CH_4) fluxes response to the water table

After the water table was raised, thus increasing the anoxic environment, and bringing it closer to the surface, the anoxic environment was expected to trigger increased methanogenic activity.

The increase in methanogenic activity would have led to increased CH_4 flux from the soil. Water tables of less than 20 cm are expected to have higher CH_4 flux, this has been backed up by studies such as Evans et al. (2021) and Renger et al. (2002). However, during the first experiment, the CH_4 fluxes from the 20 cm water table were similar to the fluxes in the 50 cm water table. 20 cm was chosen as a reasonable compromise water table due to its effectiveness in reducing CO_2 fluxes, whilst potentially still allowing some farming activity to continue, and it was anticipated that it would provide a balance between CH_4 and CO_2 fluxes.

As the CH_4 fluxes did not increase as projected with the 20 cm water, a decision was to conduct a follow-up experiment with a water table closer to the surface (0 cm). Evans et al. (2021) reported that a water table depth of less than 10 cm is more effective at increasing methanogenesis. However, after 3 weeks of 0 cm water table in the cores, the expected CH_4 fluxes still had not occurred. Instead of being sources, the cores were sequestering both CH_4 and CO_2. Studies have shown that at times a substrate is required for methanogenesis (Henneberg et al., 2016). It was suspected that the methanogens might not have enough substrate for the energy they require to produce CH_4. A FOM substrate in the form of grated potato was added to stimulate methanogenic activity. Initial data was not promising. However, after a long incubation period, there was some CH_4 production.

There are a few factors to consider in assessing the CH_4 fluxes. First, the lack of CH_4 fluxes might be due to the CH_4 being produced deeper in the soil with limited diffusion to the surface. The absence of plants may further confound the issue. Plant aerenchyma aid the movement of CH_4 from deeper anoxic zones of soil in addition to providing a substrate for methanogenesis. CH_4 emission rates tend to be lower in soil without vegetation than in soil with vegetation (Cooper et al., 2014; Henneberg et al., 2016). It is possible that CH_4 production occurred at depth, but it was consumed in the aerated layer of the soil. Deep rooting below the 20 cm aerated part of the soil would have bypassed the CH_4 oxidisers, leading to increased CH_4 flux from the cores. Secondly, the observed CH_4 consumption could be due to the presence of Fe^{3+} reducing microbes in the soil which have been shown to consume CH_4 in the process of Fe^{3+} reduction (Lovley & Phillips, 1987). The Fe^{3+} reducing microbes can outcompete methanogens thus leading to the observed CH_4 consumption (Achtnich et al., 1995). However, this is less likely to be the case with these organic soils because they are not iron rich. Thirdly, it could be that a greater amount of time is required for the methanogenic community to establish. In their study, Keane et al. (2021) reported a delay of 84 to 88 days after raising the water table.

The finding in this experiment shows that both time and substrate are key for methanogens to become dominant in cultivated peatlands. Of course, the effect of a water table <10 cm also plays a significant role. However, water tables <10 cm are not practical for agricultural productivity, therefore further study is necessary to see if substrate can have the same effect with a more practical water table e.g., 30 cm.

5.5.2 Effects of the amendments on pH

The addition of the amendments did not have detrimental effects on the pH of the soil. Only $FeCl_2$ seems to have a noticeable effect on pH, which was lowest in the 20 cm water table. $FeCl_2$ is commonly used in contaminated soils to stabilise arsenic waste in soil by reducing the pH of the waste (Lin et al., 2017). However, there is a risk of making the soil acidic; this is particularly pertinent in the cultivated peatlands in The Fens. These peatlands are minerotrophic thus they are normally not acidic which is ideal for agriculture but making them acidic will not be beneficial and could lead to the need for liming to bring the pH back to neutral. Whilst Fenland soils are generally not acidic, some of the soils (acid sulphate soils) can become extremely acidic at depth due to the build-up of sulphides in the soil which can get re-oxidised when the soil is drained therefore releasing sulphuric acid. As crops cannot root into the acidic layers, it limits crop production (Cook et al., 2000; Lindgren et al., 2022)

Nevertheless, the quantity of $FeCl_2$ added in this study was low so it did not have an adverse effect on the pH of the soil. $FeSO_4$ had a slight effect on the pH, but this was not substantially different to the control and the $CaSO_4$ amendment. $CaSO_4$ is an important amendment as it is used for soil liming Elrashidi et al. (2010); Rocha et al. (2014). As these soils were already close to a neutral pH, $CaSO_4$ did not have any effect. Using these pH effects as a proxy, the amendments could potentially be used without major impacts on the nutrient availability in the soil.

5.5.3 Effects of the amendments on electrical conductivity (EC)

There is evidence that the added amendments of $FeSO_4$ and $CaSO_4$ precipitated in the soil rather than staying in solution, as with $FeCl_2$. The $FeCl_2$ was further down the soil profile soon after application, as shown by the large spike in EC after application in week 7 (Figure 5.10). The reduction of SO_4 was most likely happening as $CaSO_4$ and $FeSO_4$ slowly precipitated down the profile.

5.5.4 Environmental risks of adding amendments

There are wider environmental risks associated with the addition of amendments. Whilst using Fe^{3+} could have produced the desired results compared to Fe2+, its environmental risk likely outweighs its impact on fluxes. Due to its insolubility, ochre (Fe (III)) causes problems for farmers as it can clog up drains and ditches (Aven, 2020). Furthermore, ochre can impact the amount of oxygen and light in rivers, thus leading to brown lifeless rivers (Amisah & Cowx, 2000). On the other hand, sulphate (SO_4^{2-}) ions can have environmental issues when combined with minerals such as clay, aluminium, or calcium. The minerals can potentially block soil pores which could have a detrimental effect on irrigation (Brahmacharimayum et al., 2019). Furthermore, if sulphate is washed away into important water sources for animals, it could damage animal health if consumed in high quantities. Negative effects of this include diarrhoea and dehydration (Backer, 2000; Brahmacharimayum et al., 2019). Additionally, there are concerns that SO_4^{2-} could lead to the formation of hydrogen sulphide (H_2S) which can produce unpleasant odours and have health and ecological damage (Long et al., 2016) Furthermore, this could contribute to the acid sulphate problem prevalent in soils with high sulphide build-up (Sohlenius & Öborn, 2004).

5.5.5 Effects of amendments on dissolved organic carbon (DOC)

This study has shown that the amendments did not lead to long term increases in the amount of DOC in the soil. The amount of DOC in the soil pore water was highest in the first week of amendment application but in the subsequent weeks, the DOC was lowest in the treatments of $FeCl_2$, $FeSO_4$ and $CaSO_4$. The amount of DOC in the soil is linked to the soil pH levels, Figure 5.12. The cores with the amendments had lower pH compared to the control cores. This finding is supported by research conducted by Clark et al. (2011) who reported that reduced DOC was related to changes in solubility driven by changes in pH. Soil studies in the Peak District, an upland area in England, UK, show that both alkaline and acid manipulation led to responses in the level of DOC; the alkaline treatments led to large increases of DOC while the acid treatments led to reduced DOC (Evans et al., 2012). Even though pH plays a key role, the role of the ionic strength cannot be ruled out as both ionic strength and pH co-vary, which is most likely with the calcium compounds (Clark et al., 2011). Overall, it appears that the amendments lowered the rate of DOC loss in the soil pore water. This could have long term benefits but as this study had temporal limitations it is not possible to conclude how long these benefits will

last. Nevertheless, the negative impact of the amendments, outweigh their small and possibly short-term impact.

5.5.6 Effects of amendment on N_2O

Whilst effects of the amendments on N_2O was not evaluated; through the binding of available N, the amendments could play a significant role in mitigating N_2O emissions from cultivated peat soil. Future studies could explore this further, however, the risks these amendments pose to the environment outweigh the benefits.

5.5.7 Impact of the quantities of amendments used

While the same weight of amendments was added, the redox potential or electron availability (E_H) of the amendments is different. E_H is a measure of the ease with which a molecule will accept electrons, which means molecules with a positive E_H are readily reduced. The ease of reduction increases with higher positive numbers (Mansfeldt, 2004). $FeCl_2$ has an E_H of +2 whilst $FeSO_4$ and $CaSO_4$ both have an E_H of +6 thus making them better sources of electrons. Due to the difference in E_H, a comparison of these amendments using the same weight is imbalanced as their capability will not be equal. A future study needs to consider this and add the amendment quantities based on their E_H instead of using the same quantity weight-wise.

5.5.8 Conclusions

The application of amendments proved to be ineffective against the fluxes of CO_2, except for $CaSO_4$ when the water table was closer to the surface at 20 cm. However, the difference in flux from the cores without any treatment was negligible thus ruling out $CaSO_4$ as an effective amendment. As $FeCl_2$ and $FeSO_4$ led to more CO_2 fluxes than the controls, this makes them unsuitable candidates even though they led to some CH_4 consumption. This study was not able to evaluate the potential benefits of the amendments because under the 'business as usual' water table depth and an elevated 20 cm water table depth, the soil did not produce enough CH_4. The follow-up experiment provided the right conditions to produce CH_4 i.e., flooded and added crop residue. However, these conditions are not compatible with active peatland horticulture and have risks such as plant disease.

What this study proves once again is that raising the water table is the best solution to reduce the flux of CO_2 from cultivated peatlands. Of course, compromises will have to be made to

allow farming activity to continue. However, with the current rates of peat loss and associated fluxes, it is time to consider rewetting the peatlands.

Chapter 6. Discussion

6.1 Overview

The studies in this thesis set out to find out ways of mitigating the impact of intensive agriculture on lowland organic soils in the UK. The goal was to find mitigation measures that are effective in reducing carbon (C) loss from cultivated peatlands without affecting food production. As the population grows, estimated to be 10 billion in 30 years (Hickey et al., 2019), there will be a need for increased food production to meet demand (Godfray et al., 2010). This demand for food production will inevitably put increased pressure on cultivated peatlands and could lead to increased peat loss and more greenhouse gas (GHG) production, especially carbon dioxide (CO_2) (Foresight, 2011; Gilbert, 2012). Currently, in Europe, agriculture is the second-largest contributor of GHG emissions at 11%, this number has increased from 9.9 % in just 5 years and is expected to keep going up. An overview of the GHG emissions is summarised in Table 6.1. Overall, all GHGs from the various sources have been fluctuating with notable decreases from the energy sector. Agriculture GHG emissions for the year 2019 were higher than they were for the years 2010 to 2013 but slightly lower than the years 2014 to 2018 (European Environment Agency, 2016; 2021).

Table 6.1 Overview of European Union (EU) - Kyoto Protocol (KP) GHG emissions (in million tonnes carbon dioxide (CO_2) equivalent) for the main source and sink categories from 1990 to 2019 (European Environment Agency, 2021).

GHG source and sink	**1990**	**1995**	**2000**	**2005**	**2010**	**2011**	**2012**	**2013**	**2014**	**2015**	**2016**	**2017**	**2018**	**2019**
Energy	4 358	4 081	4 012	4 123	3 801	3 656	3 615	3 521	3 336	3 376	3 357	3 361	3 282	3 132
Industrial Processes	530	506	463	473	397	395	383	383	389	381	381	390	380	370
Agriculture	537	469	459	437	423	422	422	424	432	433	434	437	432	429
LULUCF	-184	-274	-285	-293	-300	-298	-304	-307	-288	-283	-282	-242	-247	-234
Waste	240	246	228	200	167	161	157	151	145	142	139	138	136	135
Other	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Indirect CO_2 emissions	4	4	3	3	2	2	2	2	2	2	2	2	2	2
Total (with net CO_2 emissions/removals)	5 485	5 031	4 881	4 943	4 490	4 339	4 274	4 174	4 015	4 052	4 030	4 085	3 985	3 833
Total (without LULUCF)	5 669	5 305	5 166	5 236	4 790	4 637	4 578	4 481	4 303	4 335	4 312	4 327	4 233	4 067

** *Land-Use, Land-Use Change and Forestry (LULUCF)*

Ideally, emphasis must be placed on increasing yields whilst simultaneously reducing the impact on the environment, but the most realistic approach in the current context is to either i) maintain current yields and reduce the associated environmental impact or ii) increase yields while maintaining current environmental impact on the environment. A decrease in GHG emissions from arable agriculture while maintaining current yields would achieve the first of these objectives and could contribute significantly to the UK's GHG emission reduction targets. The studies in this thesis investigated some of the potential measures which could be adopted to preserve the cultivated peatlands through mesocosm experiments.

The specific research aims were:

1. To assess the effects of adding crop residue or fresh organic matter (FOM) on the release of GHGs such as CO_2, methane (CH_4), and nitrous oxide (N_2O) in currently cultivated UK lowland organic soils.
2. Evaluate the response of GHG emissions when FOM is added into peat soil and assess this response to water table manipulation as a mitigation measure.
3. Assess how CO_2 from cultivated peatlands will respond to added FOM of a commercially important crop in the long term after the initial application and a season of being left fallow and the long-term impact of an elevated water table depth as a mitigation measure to CO_2 fluxes.
4. Assess the effectiveness of different chemical amendments in suppressing CO_2, and CH_4 and the added effect of water table manipulation.

In this thesis, mesocosms were set up to address these aims. In Chapter 3, this was conducted using unique mesocosms and automated chambers to address the first objective under conditions that were as representative of the farm conditions as possible in a mesocosm. Fluxes of CH_4, CO_2, and N_2O were recorded frequently using a Picarro Greenhouse Gas Concentration Analyser (G2508) and LI-COR LI8100a Infra-Red Gas Analyser (IRGA). The scope of Chapter 4 was limited due to the COVID-19 pandemic. The setup was modified from Chapter 2 to allow the growing of crops but as the site was under lockdown, it was not possible to grow the crops or alter any of the parameters. In the chapter, the original plan was to i) study the long-term impact of historical FOM application on a) future plant productivity and b) long term impact on GHG fluxes of CO_2 and CH_4, ii) assess the impact of elevated water table and FOM on crop yield. The experiment would have assessed the movement of DOC and nutrients in the dip wells used to maintain water table. The Los Gatos Research (LGR) Ultraportable

Greenhouse Gas Analyzer (UGGA) that was recording the CH_4 fluxes was faulty so only the CO_2 data collected by the IRGA was useable. Nevertheless, Chapter 3 provides an insight into what happens when peat soil is left fallow after FOM application. The evidence of the impact of calcium sulphate ($CaSO_4$), iron (II) chloride ($FeCl_2$), and iron (II) sulphate ($FeSO_4$) at suppressing CO_2 and CH_4 is provided in Chapter 5. This showed that time, substrate, and water table are important in the production and consumption of CH_4 when cultivated peatlands are flooded. The assumption based on current literature was that CH_4 will increase when the cores are flooded but that was not the case as CH_4 production was only observed when a substrate was added. The first experiment in Chapter 4 showed that there was no CH_4 increase in the 20 cm water table. A follow-up experiment at 0 cm also showed no increase of CH_4 fluxes without a substrate. Both experiments add to the understanding of the dynamics of cultivated peatlands when the water table is altered.

This discussion chapter first aims to place this thesis within the wider context of the preservation of cultivated lowland organic soils. It will then analyse the specific strengths and weaknesses of the methodology adopted e.g., GHG measurements using the static closed chamber, benefits of mesocosms and their limitations. It also considers the widespread use of peatlands for agriculture in the UK and their economic importance. Finally, it identifies avenues for further research and presents concluding remarks regarding how we can improve our understanding of cultivated peatlands.

6.2 Use of peatlands for agriculture in the UK and their economic importance.

Cultivated peatlands are commercially important, especially for food production. However, when the negative impact on the climate of the use of cultivated peatlands for food productions is taken into consideration then their continued use becomes more difficult to justify.. For countries such as the UK where a lot of our food is imported, ceasing arable agriculture on peatlands could lead to increased food imports, unless production is increased on mineral soils elsewhere in the UK. Furthermore, ceasing arable production will lead to significant monetary losses. Vegetable production in the UK occupies a small area compared to rearing lamb, beef, and dairy cattle but it is high value and is disproportionately done on peat soil (National Food Strategy, 2021; NFU, 2019). Therefore, there will be an enormous economic impact if peatlands were to be taken out of production. Total crop output from The Fens alone is worth £700 million (NFU, 2019). For these reasons, mitigation measures are needed if agricultural

production is to persist. In this thesis, different mitigation measures that could potentially be used to counter the effect of intensive peatland agriculture were assessed.

6.3 Water table manipulation

The findings from this thesis conclude unequivocally that the use of an elevated water table is the best solution to reduce the amount of CO_2 being released from arable peatlands. Significant decreases in CO_2 fluxes were observed during the studies conducted as part of this thesis in response to water table manipulation. The question here remains how elevated water tables can fit into commercial production. Compromises will be needed to attain the ideal water table depths that allow crops to grow in order to maintain current levels of production or farmers might have to move away from current practices and crops. Machinery is available that can work with water tables close to the surface, but the crops are a big issue. Most crops grown on peatlands are suited for drier conditions. Shifting to having a water table close to the surface means that farmers will need to adopt new crops. This is where paludiculture comes into play.

The paludiculture land use concept can allow for the cultivation of crops that can either be used for food, energy, or other industrial products. Shifting to paludiculture will allow peatlands to remain in agricultural production. However, paludiculture is not without its challenges such as increased CH_4 emissions from the flooded soil. Furthermore, there are only a few food crops that can be grown through paludiculture in colder climates such as the UK. Rice (*Oryza sativa* L.) which is a common paludiculture crop is intolerant to cold temperatures (Ghadirnezhad & Fallah, 2014; Khush, 1997). Most crops that can be grown through paludiculture in the temperate regions are used for fodder and biomass, particularly for building materials or bioenergy. Therefore, paludiculture might not offer for food security i.e., a demand for carbohydrate and protein; available evidence suggests that paludiculture would result in lower agricultural profitability than conventional agriculture within the current economic landscape and so it will not be immediately viable in all circumstances (Freeman et al., 2022). Genetically modified (GM) crops that are tolerant to excess water could offer the solution. However, research to create such crops is still ongoing. Hartman et al. (2019) studied *Arabidopsis thaliana* and their findings point to the potential of the development of flood-tolerant crops. Further to the challenge of developing flood-tolerant crops, greater acceptance of GMOs globally will be required which will not be an easy task and is likely to require legislation guidance (Garland, 2021).

6.4 Use of iron (II) chloride ($FeCl_2$), iron (II) sulphate ($FeSO_4$) and calcium sulphate ($CaSO_4$) as amendments

This thesis also explored the use of amendments to counter the GHG emissions released from cultivated peatlands. However, the use of these amendments ($FeCl_2$, $FeSO_4$, and $CaSO_4$) proved to be ineffective. Only $CaSO_4$ influenced the fluxes of CO_2, but the effect was not significant enough to warrant its use as a regular amendment to mitigate the effect of intensive arable agriculture.

6.5 Crop residue or fresh organic matter (FOM)

When added to cultivated peat soil, FOM proved to have some benefits. Most of the added FOM remained in the soil. If continued over time, this can provide some damage limitation. However, the amounts of C added here are minuscule (0.000000016 t C ha^{-1}) and cannot be used to justify the continuation of arable agriculture on cultivated peatlands where CO_2 emissions are estimated to be 39 t CO_2e ha^{-1} yr^{-1} (Evans et al., 2017). Perhaps if used in conjunction with other practices such as raising the water table depth or reduced tillage agriculture, it could be viable. Future studies could explore this further. Conversely, when FOM was coupled with a water table closer to the surface (20 cm), it released more N_2O into the atmosphere; 0.00000000017 t CO_2eq ha^{-1} compared to 0.00000000014 t CO_2eq ha^{-1} for the 50 cm cores. While this can be used to justify continued arable activity on cultivated peatlands, the quantities consumed were once again low and can in no way offset the enormous amounts of CO_2 being released into the atmosphere due to peatland cultivation. Furthermore, the time of this experiment was not long, a longer study would be necessary to justify upscaling these findings.

6.6 Methodology strengths and weaknesses

6.6.1 Mesocosms and static chambers

In this study, it was initially intended to deploy larger chambers using the SkyLine 2D system, Figure 6.1. The SkyLine 2D is a novel automated system (Ineson & Stockdale, 2014). The system would have allowed the deployment of an automated chamber on an active farm. Due to the size of the chamber, it would have enabled the measurement of fluxes over larger crops such as wheat (*Triticum aestivum* L.), Figure 6.1. However, after many weeks of setting up the systems and conducting trial runs, it proved to be unsuccessful. The system was still under

development and the equipment used were still prototypes. For this reason, all the experiments in this study were conducted in mesocosms. The use of static chambers in this thesis was ideal as all the experiments were conducted in mesocosms and they proved to be dependable and required less technical knowledge, unlike the SkyLine 2D. While the SkyLine 2D used might have failed, further iterations have been successfully deployed e.g., Keane et al. (2021).

Figure 6.1 SkyLine 2D automated system can be deployed onto an active field (top-right). It comes with a large clear chamber (left) which has enough clearance for large crops such as wheat (Triticum aestivum L.).

The mesocosms in Chapter 2, were set up to be as representative of the field as possible. To achieve this, larger cores were collected than previous studies to increase the soil volume and surface area e.g., Matysek, et al. (2019), Musarika, et al. (2017) and Wen et al. (2019). There were limits to the size of the cores as large cores can lead to the compaction and disturbance of soil structure. A trial of 2-metre-long cores was unsuccessful as the cores severely compacted the soil and most of them shattered when they were being driven into the soil using heavy machinery. Unlike previous studies (Matysek et al., 2019; Musarika et al., 2017), the mesocosm study was conducted outside so that there was no artificial atmospheric homogeneity provided by specialised growth chambers where everything is fully controlled.

Even though there were minor differences in the climatic conditions between the farm where the soil was sampled from and the facility where the experiments were conducted, these differences were not significant enough to affect the outcome of the experiments compared to the impact of moving the cores into a controlled indoor environment. Even though the daily weather conditions varied, there was enough rain that fell onto the cores to not require manual watering, which is necessary for indoor mesocosms.

Most experiments that were conducted in mesocosms recorded fluxes infrequently e.g., twice a week or once a month (Musarika et al., 2017; Taft et al., 2017). In Chapter 2 and Chapter 3, using automated chambers, fluxes were recorded once every hour for each core for the duration of the experiments. Chapter 3 used the same setup as Chapter 2 with modifications made to allow crop growth. Chapter 3 provides an insight into what happens when peat soil is left fallow after FOM is applied. In Chapter 4, the GHG fluxes were not collected as frequently as the experiments in Chapter 2 and Chapter 3. The fluxes were collected manually twice a week using an LGR Benchtop Greenhouse Gas Analyser (Model: 908-0010-0002) and a custom-made closed chamber. Furthermore, no N_2O fluxes were recorded in this experiment because no additional nitrogen (N) was added to the soil so there was no expectation of different N_2O fluxes in the soil.

6.7 Future research

The studies conducted in this thesis have answered some of the questions about the preservation of cultivated peatlands. What it has conclusively shown is that raising the water table depth is the best option for mitigating the impacts of intensive agriculture on lowland organic soils and the other proposed mitigation measures (i.e., $FeSO_4$, $CaSO_4$, $FeCl_2$, and FOM) are likely to offer at best marginal additional mitigation, and certainly will not substitute the raising of water levels. However, whilst the effects of the water table on the fluxes of CO_2 were definitive, its effects on CH_4 showed that further analysis is required. Long-term studies would be ideal, and the inclusion of plants, especially aerenchymous plants. This could be compared against non aerenchymous and common agricultural crops grown in lowland organic soils.

Evidence has shown that you rarely see CH_4 fluxes until the water table depth is closer to the surface, 20 cm or less (Evans et al., 2021). In the experiments in this thesis, CH_4 fluxes were small. These experiments support previous understanding and counter concerns that moderate raising of water table depth might lead to CH_4 emissions from cultivated peatlands. CH_4 fluxes

were only noticeable after the cores were flooded and crop residue was added. However, there was a considerable lag between raising water levels and the commencement of methanogenesis, Keane et al. 2021 reported a lag of 84-88 days between raising the water table increased CH_4 production. Realistically, on an active farm, such water table depth would not be possible. What this proves is that during the fallow period, the water table can potentially be raised close to the surface without producing enormous amounts of CH_4.

Before any recommendations can be made to the greater farming community, more studies need to be conducted around the longer-term impact on crop production under elevated water table depth especially the issue of plant diseases that can result due to the presence of water, such as water mould. Only a handful of studies exist that have looked at the impact of elevated water table on crop yield, such as Matysek et al. (2019), Musarika et al. (2017), Renger et al. (2002) and Stanley and Harbaugh (2002). However, more crops need to be studied to further our understanding of which crops will be negatively impacted by elevating the water table. Furthermore, there have been no studies in cultivated peatlands that have studied the impact of plant diseases that could arise from elevated water table depth.

6.8 Concluding remarks

This thesis addressed important questions regarding how the impact of intensive agriculture on lowland organic soils can be mitigated. It assessed the potential of adding of crop residue, $FeSO_4$, $FeCl_2$, $CaSO_4$, and raising the water table closer to the surface to mitigate the impact on GHG emissions. This thesis concludes that:

i. The addition of barley (*Hordeum vulgare* L.) straw as a source of FOM is beneficial when it comes to emissions of CH_4 and N_2O where consumption was observed in the 50 cm water table. However, when the water table is closer to the surface, more N_2O will be produced.
ii. The addition of FOM led to an initial increase in the fluxes of CO_2 upon application but there was no evidence of PE as the mass balance calculation showed that a substantial amount of the added FOM remained in the soil. At the very least, adding FOM is beneficial to cultivated peatlands as it has the potential to increase C stock. However, it will not be possible for FOM addition to offset the enormous amount of CO_2 emissions from the soil.

iii. The influence of FOM on the fluxes of CO_2 can be long-lived as in the long-term analysis, FOM still had some influence on the fluxes, however, this was not as strong as when it was applied. FOM would need to be applied frequently to accumulate and be beneficial.

iv. Apart from $CaSO_4$, the application of the amendments studied are ineffective against the fluxes of CO_2, and there is no justification for adding them given their greater negative environmental consequences.

v. Ultimately, water table depth proved to be the most effective mitigation measure. This demonstrates that the effects and benefits of water table depth on CO_2 fluxes are long-term making it the best approach at preserving cultivated peatlands. But this approach is replete with challenges that could make it unfeasible for crop production.

Appendices

Appendix A

Chapter 2: Crops that can be grown under paludiculture.

Numerous crops can be grown in the temperate zones and tropical zones and some of them can be used for food Table 0.1.

Table 0.1 Crops that can be grown for food, fibre, fodder, and other uses in the temperate and tropical zones.

Name	Area of viability	Usage
Redtop (*Agrostis gigantea* Roth) & (*A. stolinifera* L.)	(Temperate Zones) Europe, Asia, N. America	Fodder
Marsh foxtail (*Alopecurus geniculatus* L.)	(Temperate Zones) Europe, Asia	Fodder
Sloughgrass (*Beckmannia syzigachne* (Steud.) Fernald.)	(Temperate Zones) N. America	Fodder
Reedgrass (*Calamagrostis canadensis* (Michx.) P.Beauv.) or *C. Canescens* (Weber.) Roth.)	(Temperate Zones) Europe, Asia, N. America	Fodder
Giant manna grass (*Glyceria maxima* (Hartm.) Holmb.)	(Temperate Zones) Europe, Asia	Fodder
Marsh bird's-foot trefoil (*Lotus pedunculatus* Cav.)	(Temperate Zones) Europe, Asia	Fodder
Reed canary grass (*Phalaris arundinaceae* L.)	(Temperate Zones) Europe, Asia, N. America	Fodder
Brachiaria mutica Stapf.	(Tropical Zones)	Fodder
Echinochloa colona (L.) Link.	(Tropical Zones)	Fodder
Echinochloa crus-galli (L.) Beauv.	(Tropical Zones)	Fodder
Echinochloa polystacha (Kunth.) Hitchc.	(Tropical Zones)	Fodder

Hymenachne amplexicaulis (Rudge.) Nees.	(Tropical Zones)	Fodder
Swamp milkweed (*Asclepias incarnata* L.)	N. America	Fibre, stuffing, cosmetics
Soft rush (*Juncus effusus* L.)	Europe, N. America, Asia	Thatching, mats, baskets, fibre
Spiny rush, koga (*Juncus acutus* L.)	Southern Europe (Turkey), N. Africa	Spikes, baskets, mats
Schoenoplectus spp. (Rchb.) Palla.	Europe, Asia, America	Matting, chairs, thatching
Atlantic white cedar (*Chamaecyparis thyoides* (L.) Britton, Sterns & Poggenb.)	Eastern N. America	Wood
Jelutong (*Dyera polyphylla* (Miq.) Steenis.)	SE Asia (PSF)	Wood
Ramin (*Gonystylus bancanus* (Miq.) Kurz.)	SE Asia (PSF)	Wood
Sweetgum, sweet amber (*Liquidambar styraciflua* L.*)*	Eastern N. America	Wood
Water fir (*Metasequoia glyptostroboides* Hu & W.C.Cheng)	West-central China	Wood
Water-tupelo (*Nyssa aquatica* L.)	South-eastern N. America	Wood
Pinus mariana (Mill.) Münchh., *P. sylvestris* L., *P. contorta* Douglas., *P. taeda* L.	Europe, Asia, N. America	Wood
Swamp cypress (*Taxodium distichum* (L.) Rich.)	South-eastern N. America	Wood
Meranti (*Shorea balangeran* (Korth.) Burck.)	SE Asia (PSF)	Wood
Black chokeberry (*Aronia melanocarpa* (Michx.) Elliott.)	N. America	Food
Buffalo or sweet grass (*Hierochloe odorata* (L.) P.Beauv.)	Europe, N. Asia, N. America.	Food
Lotus (*Nelumbo nucifera* Gaertn.)	Asia; introduced to Europe, N. America	Food

Cranberry (*Oxycoccus palustris* Pers.)	Europe	Food
Cloudberry (*Rubus chamaemorus* L.)	N. Europe	Food
Water chestnut (*Trapa bicornis* Osbeck. & *T. natans* L.)	Europe, Asia, N. America	Food
Northern wild rice (*Zizania palustris* L.)	N. America	Food

Appendix B

Chapter 2: Data comparison between the LI-COR IRGA LI8100A and the Picarro G2508 analysers

Using both analysers together led to independent two sets of CO_2 fluxes being generated. Having the two sets of CO_2 data was an advantage as it allowed for data comparison to ensure they both were reporting similar fluxes. Figure 0.1 shows the CO_2 fluxes from both analysers were similar, thus providing a degree of confidence in the accuracy of the analysers.

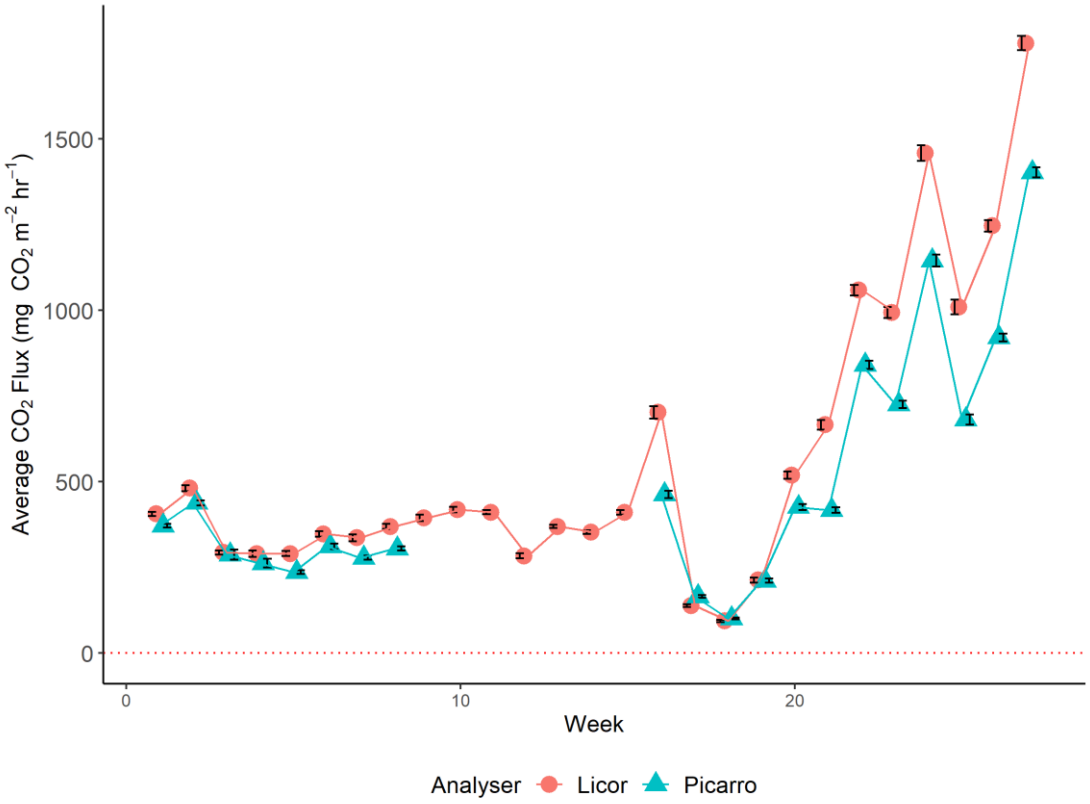


Figure 0.1 Weekly average carbon dioxide (CO_2) fluxes for all the cores from the LI-COR IRGA LI8100A compared to the fluxes read on the Picarro G2508. The G2508 stopped working as highlighted by the missing data. All values represent means ± SEM (n = 4).

Picarro G2508 CO_2 Fluxes

The G2508 gas analyser which was connected in series to the LI-COR LI8100 IRGA recorded comparable CO_2 flux results. Figure 0.3 shows the average fluxes for the duration of the

experiment which is comparable to the IRGA fluxes shown in Figure 0.2. Similarly, comparable results were observed in Figure 0.4 and Figure 0.5 which show the average fluxes every week, except for the period when the G2508 stopped working, leaving short gaps in the data. The missing data led to overall values being marginally lower but still comparable.

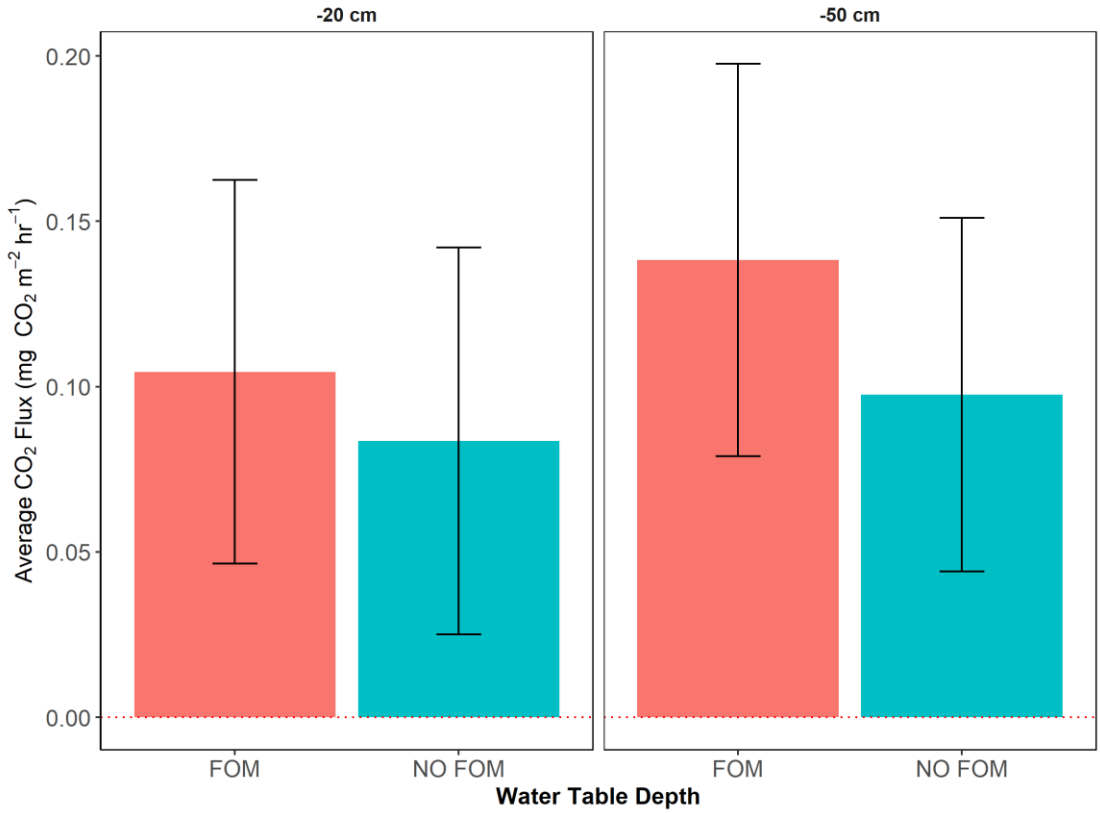


Figure 0.2 Average carbon dioxide (CO_2) fluxes for the duration of the study recorded with the LI-COR IRGA either in the presence or absence of fresh organic matter (FOM) added in the form of barley (Hordeum vulgare L.) straw. All values represent means ± SEM (n = 4).

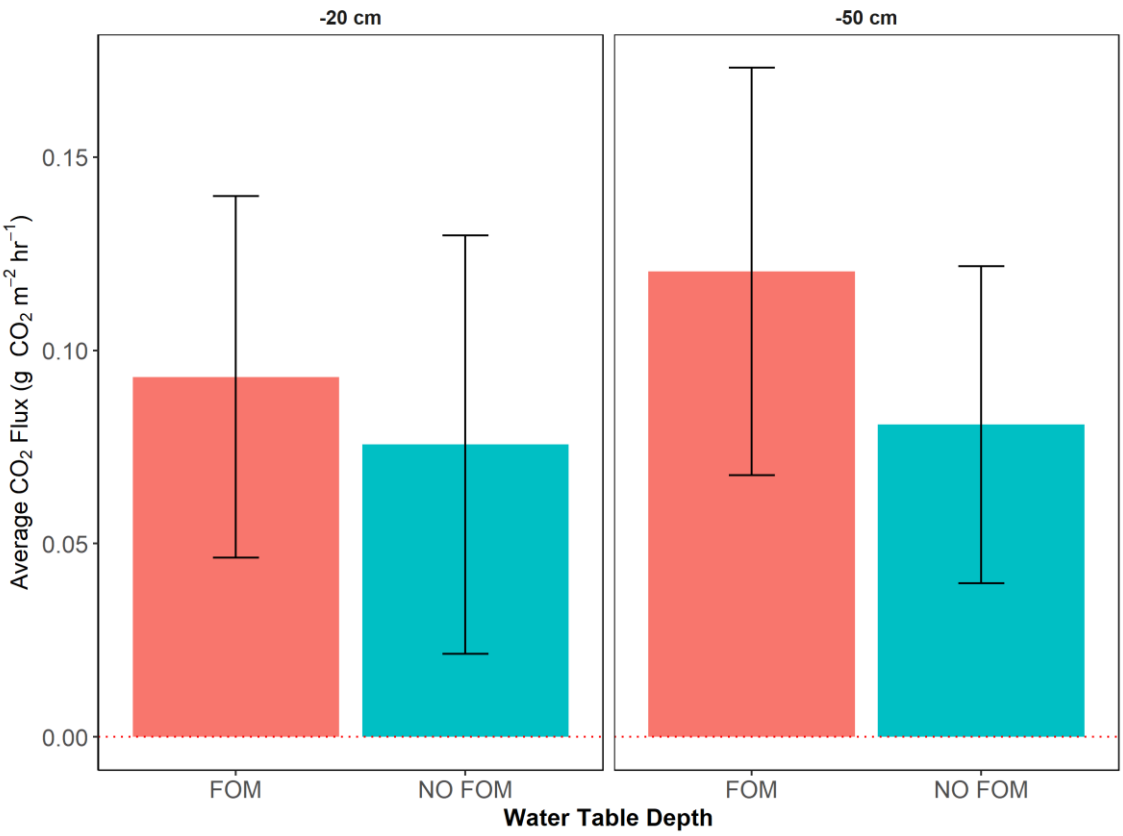


Figure 0.3 Average carbon dioxide (CO_2) fluxes from the Picarro G2508 gas analyser either in the presence or absence of fresh organic matter (FOM) added in the form of plant residues. All values represent means ± SEM (n = 4).

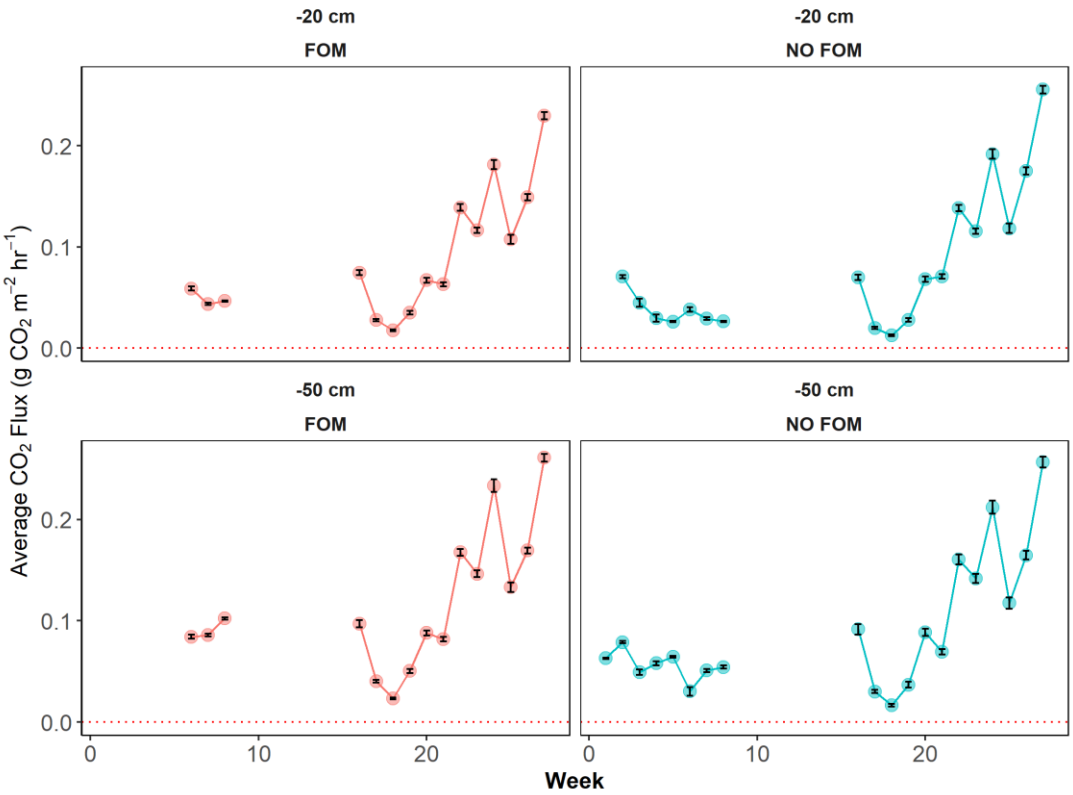


Figure 0.4 Average soil carbon dioxide (CO_2) fluxes from lowland peat mesocosms either in the presence or absence of fresh organic matter (FOM). Data was recorded using a Picarro G2508 gas analyser. The missing data is due to a fault with the analyser between weeks 9 and 15. All values represent means ± SEM (n = 4).

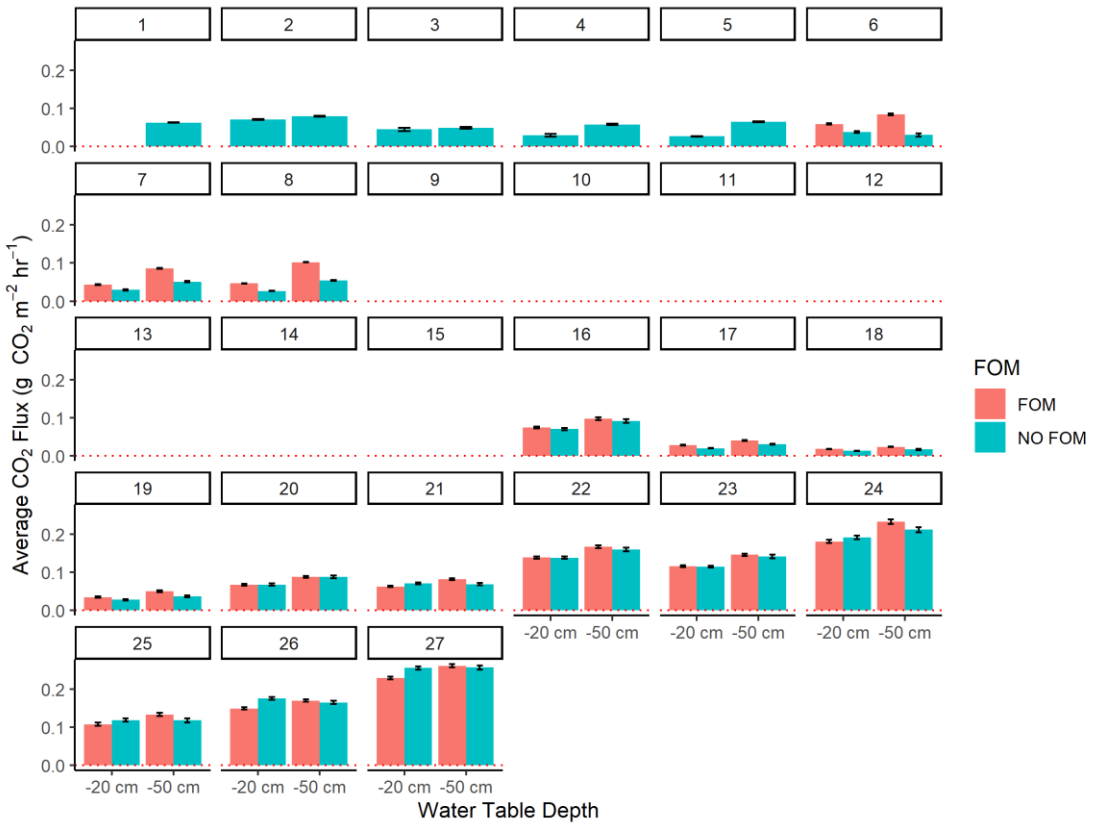


Figure 0.5 Average carbon dioxide (CO_2) fluxes for each week measured using a Picarro G2508 gas analyser either in the presence or absence of fresh organic matter (FOM) added in the form of plant residues. The data for weeks 9-15 is missing due to the G2508 gas analyser breaking down. Values represent means ± SEM (n = 4) except for week 1 (n = 16) and week 2 to 5 (n = 8).

Appendix C

Chapter 3: Data normality tests

The statistical tests were conducted using the following models in R. Normality tests were performed on the data using library(ggpubr) and library(moments). The packages showed that the data was positively skewed. As there was a strong positive skew, data transformation was needed. Consequently, the data was log(x) transformed. The log-transformed data did fit the normality test.

```
> skewness(fluxes.one$CO2dry_Licor, na.rm = TRUE)

[1] 2.725222
```

CO_2 fluxes from the LI-COR LI8100

The R model below is the model that was used to calculate the statistical significance of the observed CO_2 fluxes from the LI-COR LI8100 Infra-Red Gas Analyser (IRGA). As the data is measured repeatedly on the same cores over time, a Repeated Measures ANOVA (Analysis of Variance) approach was most appropriate. The model assesses the effect of the water table and fresh organic matter (FOM) whilst considering the chambers as a random variable.

```
fluxmodel1Licor <- lmer(CO2dry_Licor ~ Water.Table * FOM
+(1|Chamber), REML = TRUE,data=fluxes.one,
na.action=na.exclude)
```

The diagnostic plot in Figure 0.6 shows the result of the normality test done before the data was transformed.

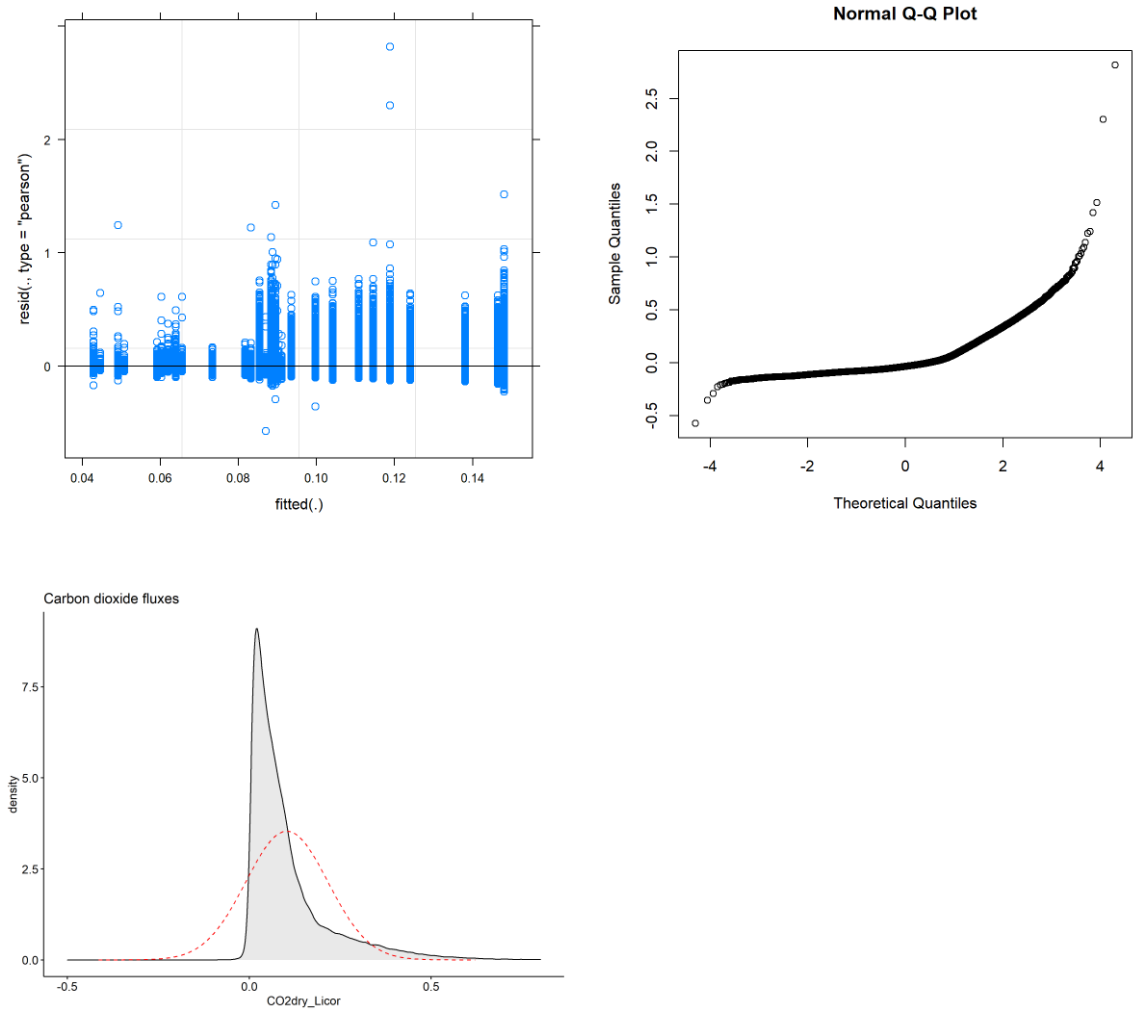


Figure 0.6 Results of a normality test on the carbon dioxide (CO_2) data from the LI-COR LI8100 Infra-Red Gas Analyser (IRGA) before it was transformed. The image top right shows the residuals whilst the top right image shows the QQ plot of the data, and the bottom image shows the skew distribution.

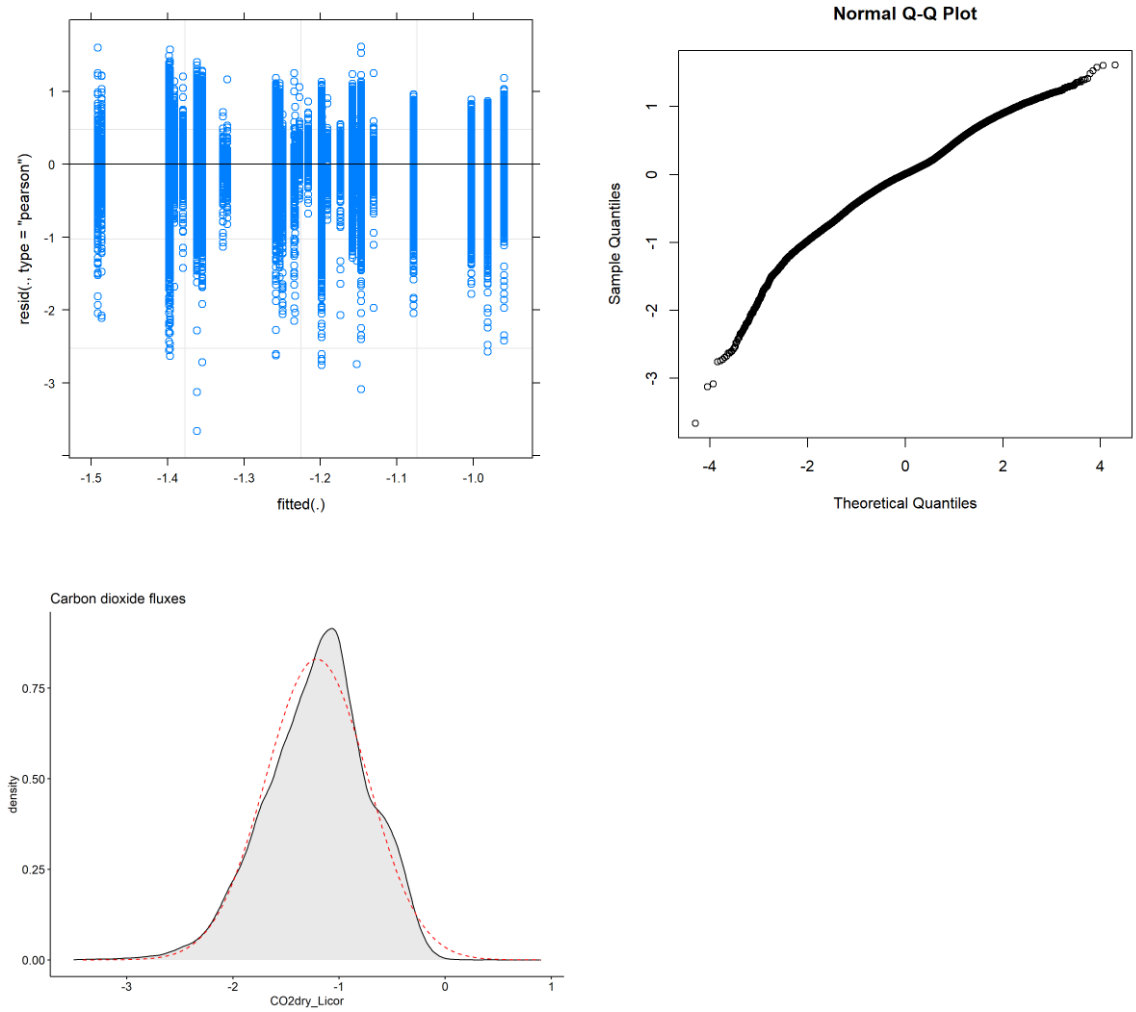


Figure 0.7 Results of a normality test on the carbon dioxide (CO_2) data from the LI-COR LI8100 Infra-Red Gas Analyser (IRGA) after it was log(x) transformed. The image top right shows the residuals whilst the top right image shows the QQ plot of the data, and the bottom image shows the distribution without a skew.

CO_2 fluxes from the Picarro G2508

For comparison purposes, the data from the Picarro G2508 was also subjected to the same approach as the LI-COR LI8100 Infra-Red Gas Analyser (IRGA) data. A similar model, as shown below was used to calculate the statistical significance of the observed CO_2 fluxes from the Picarro G2508 GHG analyser.

```
fluxmodel1Picarro <- lmer(CO2dry_Picarro ~ Water.Table * FOM
+(1|Chamber), REML = TRUE,data=fluxes.one,
na.action=na.exclude)
```

The diagnostic plot in Figure 0.8 shows that the data from the Picarro G2508 was distributed in similar patter to the data from the LI-COR analyser.

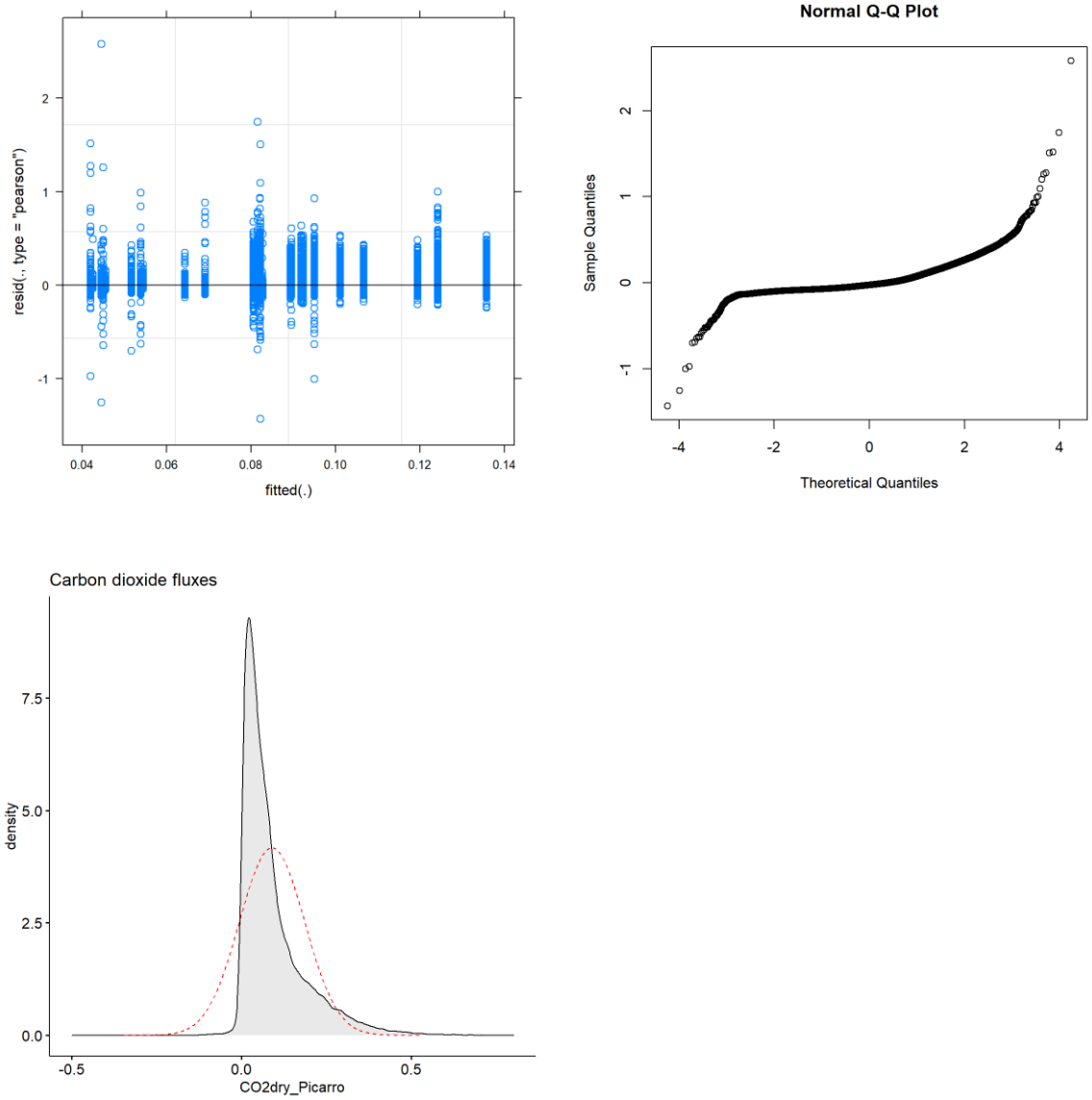


Figure 0.8 The results of a normality test on the carbon dioxide (CO_2) data from the Picarro G2508 data before it was transformed showed that it was similar to the LI-COR LI8100 IRGA data. The image top right shows the residuals whilst the top right image shows the QQ Normal plot of the data, and the bottom image shows the skew distribution.

The diagnostic plot in Figure 0.9 shows the result of the normality test done before the data was transformed. The data fits the normality assumptions.

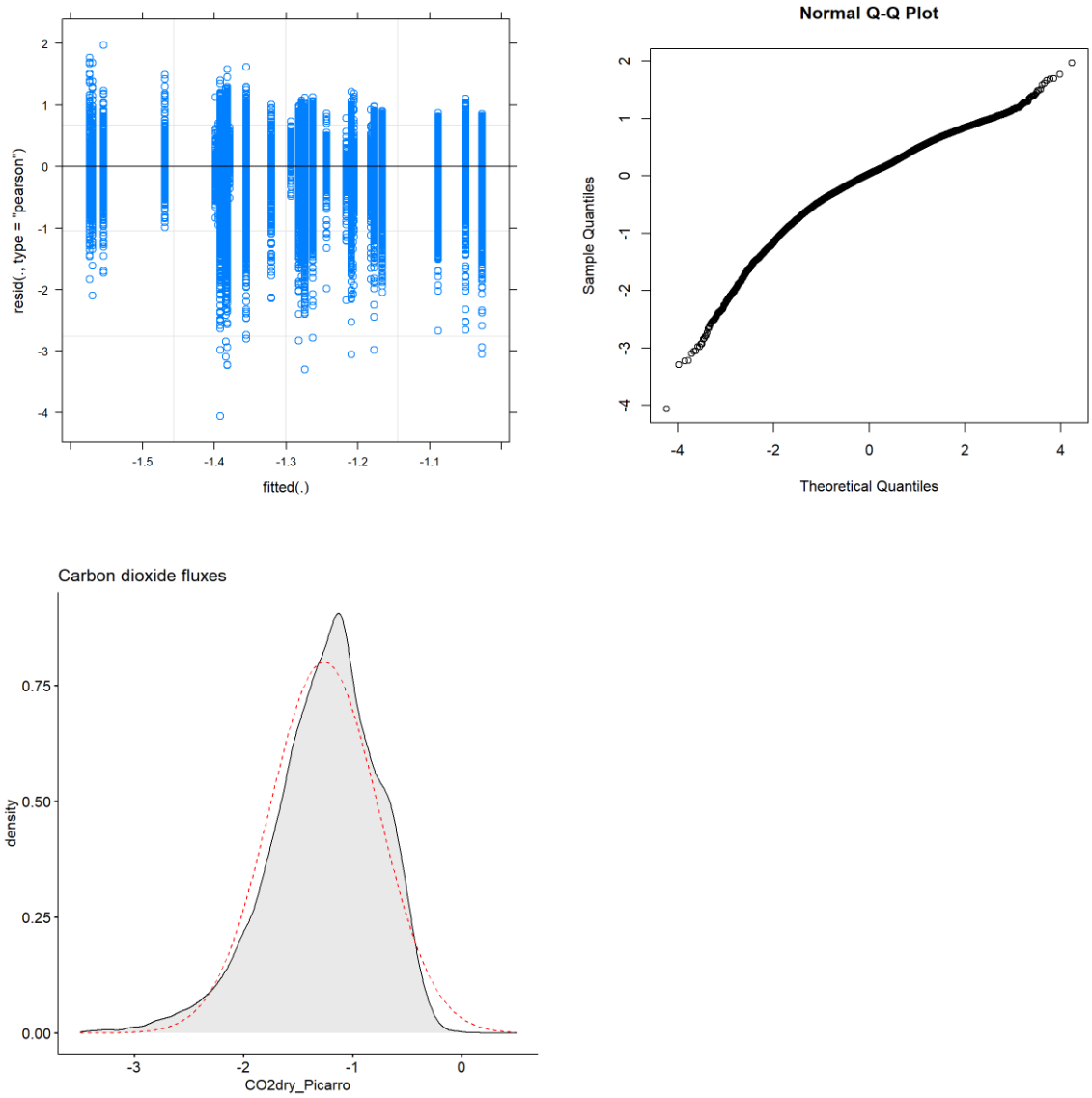


Figure 0.9 Results of a normality test on the carbon dioxide (CO_2) data from the Picarro G2508 after it was log(x) transformed. The image top right shows the residuals whilst the top right image shows the QQ plot of the data, and the bottom image shows the distribution without a skew.

CH_4 fluxes from the Picarro G2508

CH_4 fluxes were analysed using the same model as that used for CO_2 fluxes from both the LI-COR LI8100 IRGA and the Picarro G2508. The model was arranged as follows:

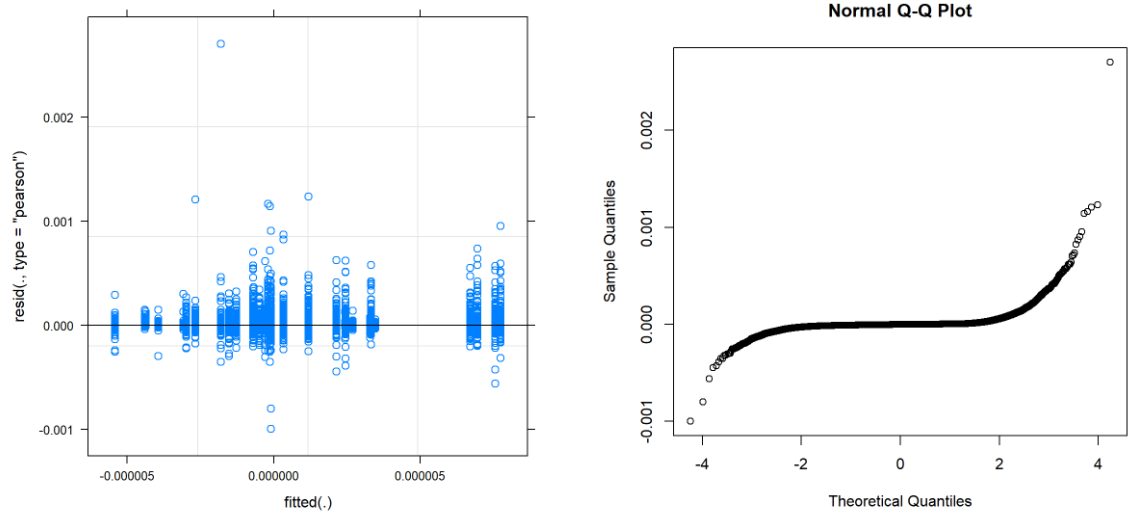
```
fluxmodel2PicarroCH4 <- lmer(CH4dry_Picarro ~ Water.Table * FOM
+(1|Chamber), REML = TRUE,data=core.data,
na.action=na.exclude)
```

The diagnostic plot as shown in Figure 0.10 indicates that the CH_4 data a similar distribution to the CO_2 data, i.e., it was not normally distributed, therefore there was need to transformation. Furthermore, a skewness test done on the data also confirmed that there was a positive skew.

When the CH_4 data was assessed for skewness, it showed a strong positive skew.

```
> skewness(fluxes.one.picarroB4$CH4dry_Picarro, na.rm = TRUE)

[1] 16.72481
```



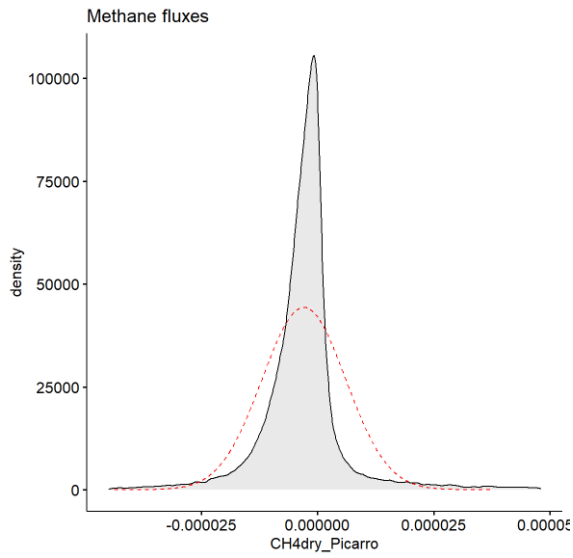


Figure 0.10 The distribution of the methane (CH_4) was not normally distributed similar to the CO_2. The image top right shows the residuals whilst the top right image shows the QQ Normal plot of the data, and the bottom image shows the skew distribution.

After the data was log (x) transformed, fitted the normality assumption Figure 0.11.

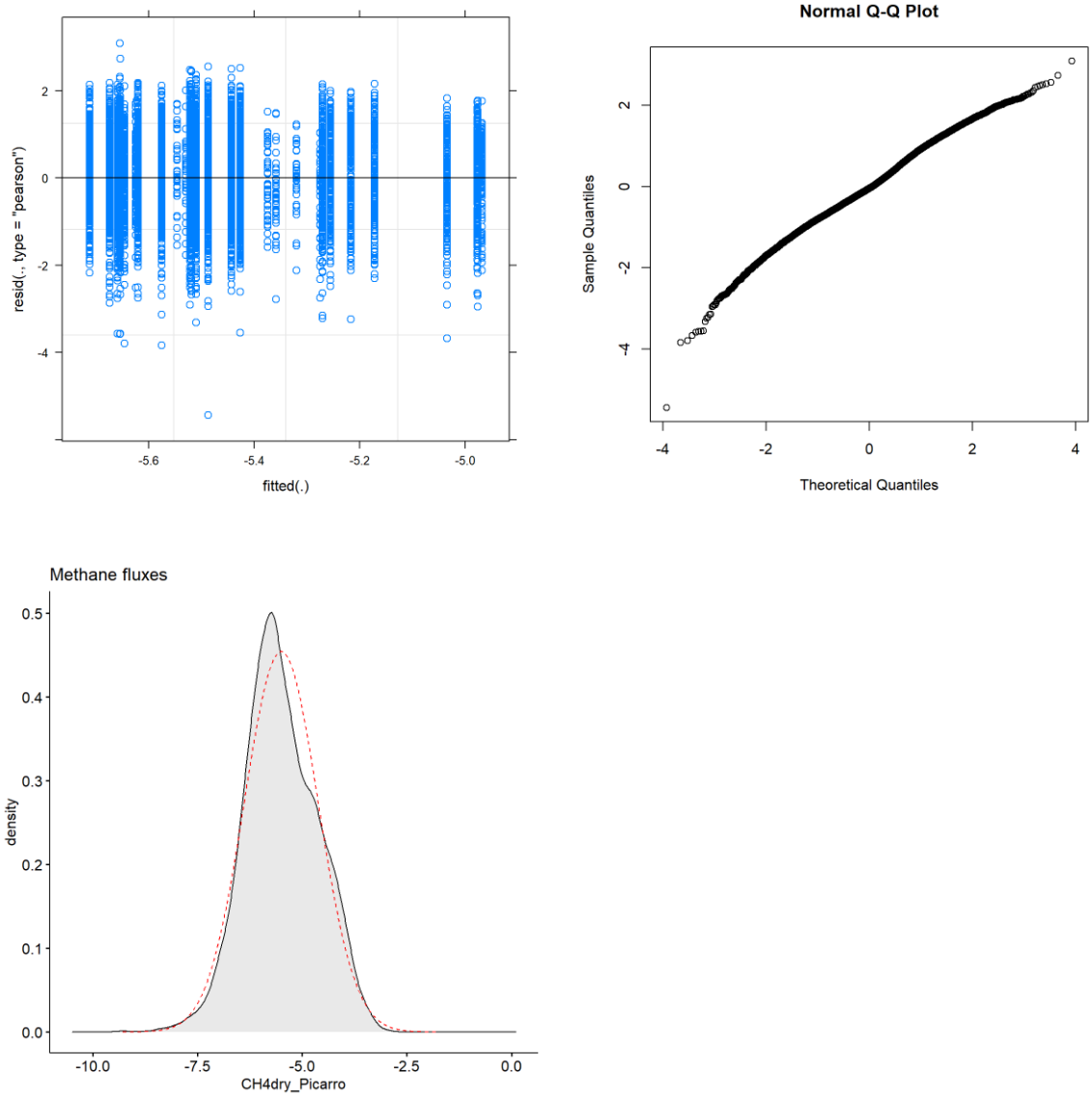


Figure 0.11 Results of a normality test on the methane (CH_4) data from the Picarro G2508 after it was log(x) transformed. The image top right shows the residuals whilst the top right image shows the QQ plot of the data, and the bottom image shows the distribution without a skew.

N_2O fluxes from the Picarro G2508

The N_2O fluxes were calculated using a similar model to the fluxes of CO_2 and CH_4. The full model is as follows:

```
fluxmodel2PicarroN2O <- lmer(N2Odry_Picarro ~ Water.Table * FOM
+(1|Chamber), REML = TRUE,data=core.data,
na.action=na.exclude)
```

The diagnostic plot in Figure 0.12 shows the result of the normality test conducted on the model above. Similar to the CO_2 and CH_4 Data, the data was not normally distributed therefore required transformation. A skewness test on the data showed that there was a strong positive skew.

```
> skewness(fluxes.one.picarroB4$N2Odry_Picarro, na.rm = TRUE)

[1] 9.344394
```

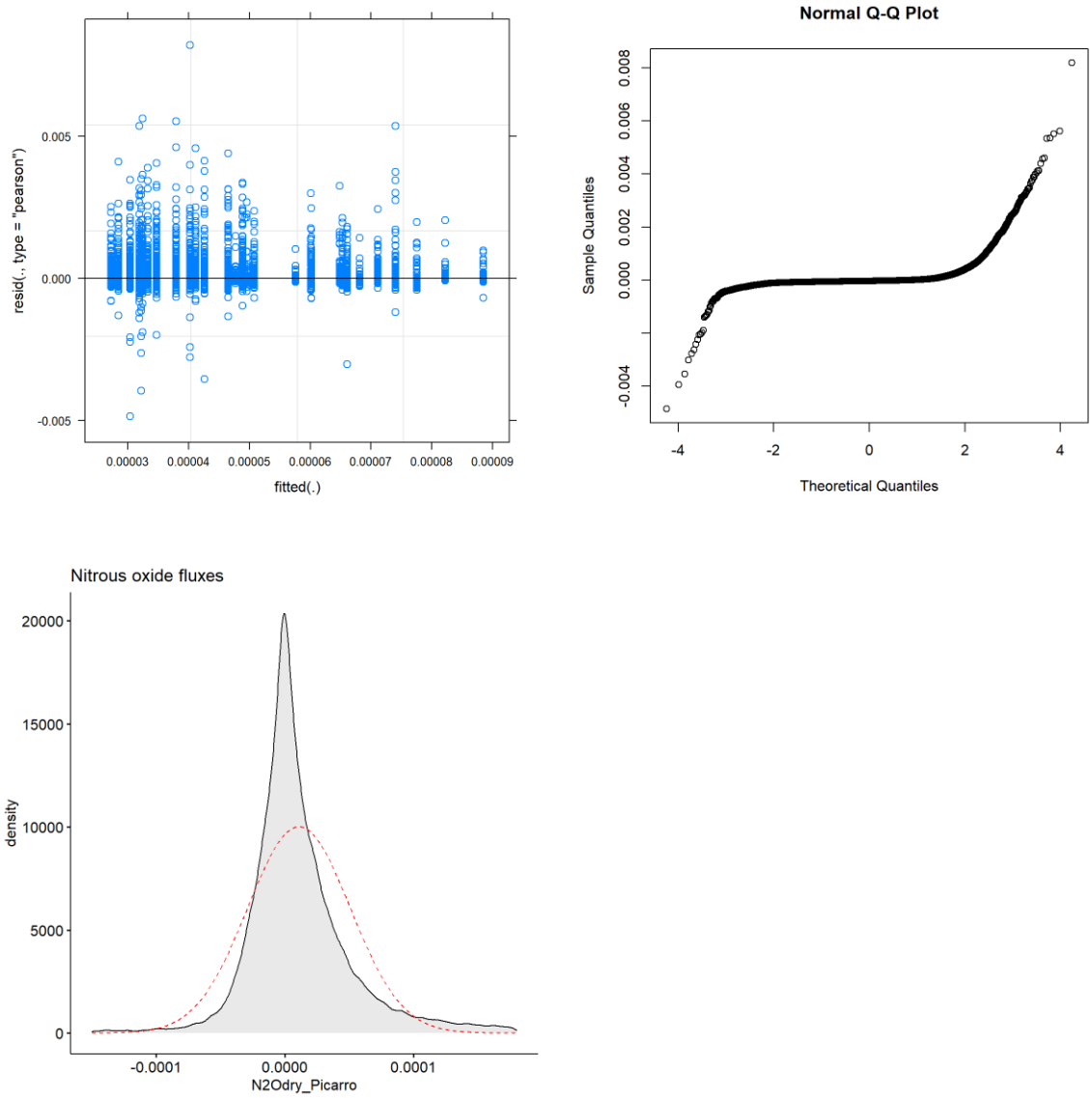


Figure 0.12 The normality tests on the nitrous oxide (N_2O) flux data from the Picarro G2508 showed the data distribution was not normal. The image top right shows the residuals whilst the top right image shows the QQ Normal plot of the data, and the bottom image shows the skew distribution.

After the data was log (x) transformed, it fitted the normality assumptions Figure 0.13.

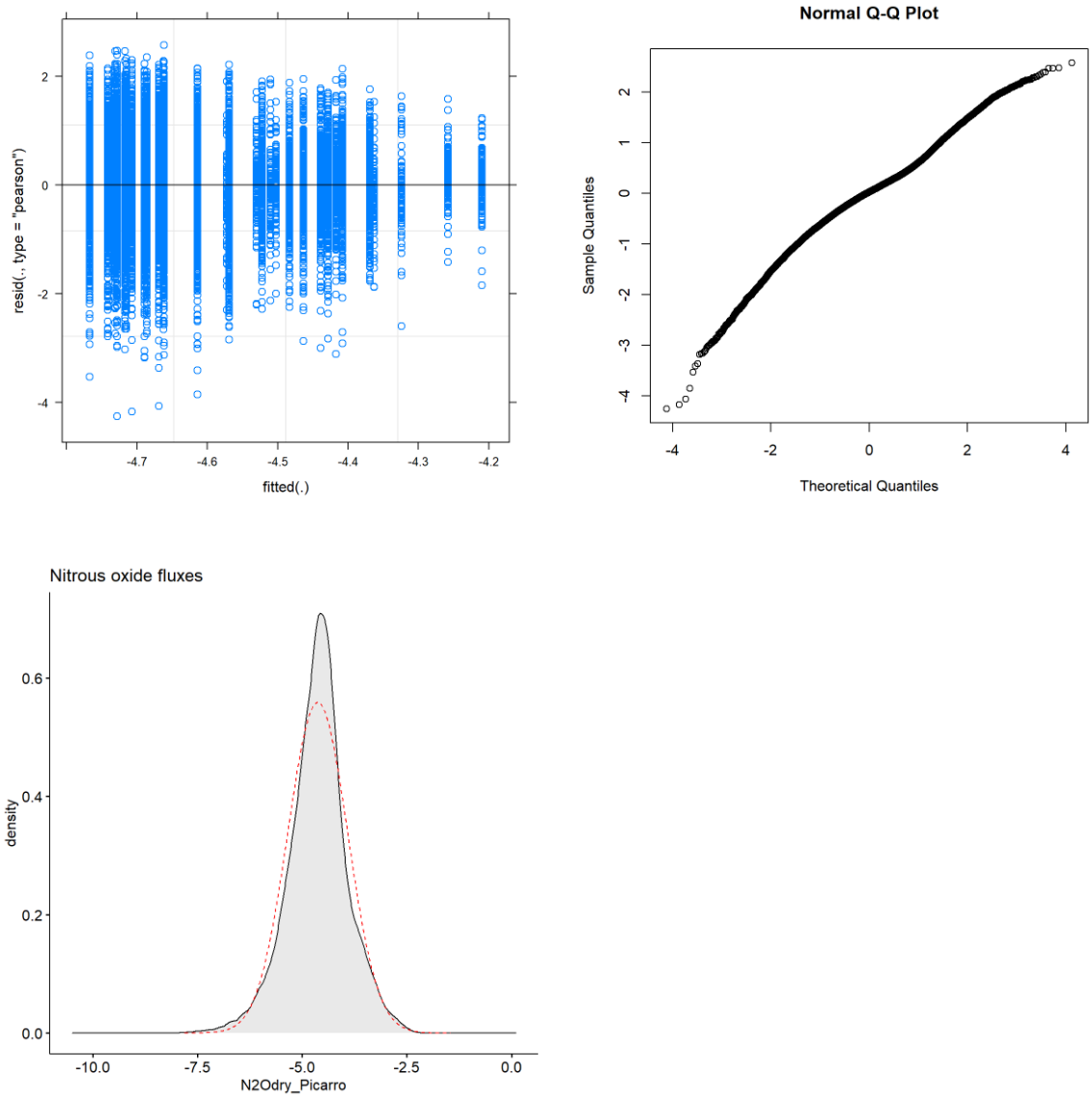


Figure 0.13 Results of a normality test on the nitrous oxide (N_2O) data from the Picarro G2508 after it was log(x) transformed. The image top right shows the residuals whilst the top right image shows the QQ Normal plot of the data, and the bottom image shows the distribution without a skew.

Appendix D

Chapter 3: LI-COR offsets

The LI-COR IRGA requires the offsets of the collars to be defined. The LI-COR collar offsets for this study are as shown in Table 0.2 and were calculated as per instruction from the LI-COR manual. The chamber offset for the Long-Term Chambers is measured by the distance between the soil surface and the upper edge of the chamber base plate. The first step is to measure the distance from the soil surface to the top of the soil collar and then subtract the distance between the upper edge of the chamber base plate and the top of the soil collar to obtain the offset value.

According to the LI-COR manual, this value should be as close to 1 cm as possible since the default Long-Term chamber volume value in the software accounts for displacement by the collar. That displacement is estimated assuming that the collar is 7 mm thick and protrudes 1 cm above the chamber base plate. However, in this study, due to minor variations in the topography, it was not possible to get every collar within the recommended 1 cm. However, this variation in offsets does not affect the flux calculations. The SoilFluxPro™ software that is provide by LI-COR to calculate the fluxes can compensate the offset values in the flux calculations.

Table 0.2 Offset values for all the collars on the LI-COR autochambers (Long Term Chamber –Model 8100-104). LI-COR recommends the offset value be as close to 1 cm as possible. However, due to the unique setup and topography, it was not possible to get every collar within the recommended 1 cm.

Chamber Number	Depth to soil surface – collar Exposed height	Offset (cm)
1	4 – 4.5	-0.5
2	4.5 – 3	1.5
3	4.0 – 3.5	0.5
4	3.5 – 3.5	0
5	3.2 – 3.5	-0.2
6	4 – 3.7	-0.3
7	4 – 3.3	0.7
8	3.7 – 4	-0.3
9	4.0 – 3.0	1
10	4.5 – 3.5	1
11	4 – 3	1
12	7 – 3.5	3.5
13	7 – 4	3
14	5- 3	2
15	6.2 – 4.5	1.7
16	4.5 – 3.5	1

Appendix E

Chapter 4: Data normality tests

The normality tests on the data used on the cumulative statistics showed that the data was normally distributed as shown in xx. The skewness test showed that there was a minor positive skew.

```
> skewness(cumweekly$meanCO2licor, na.rm = TRUE)

[1] 0.5983457
```

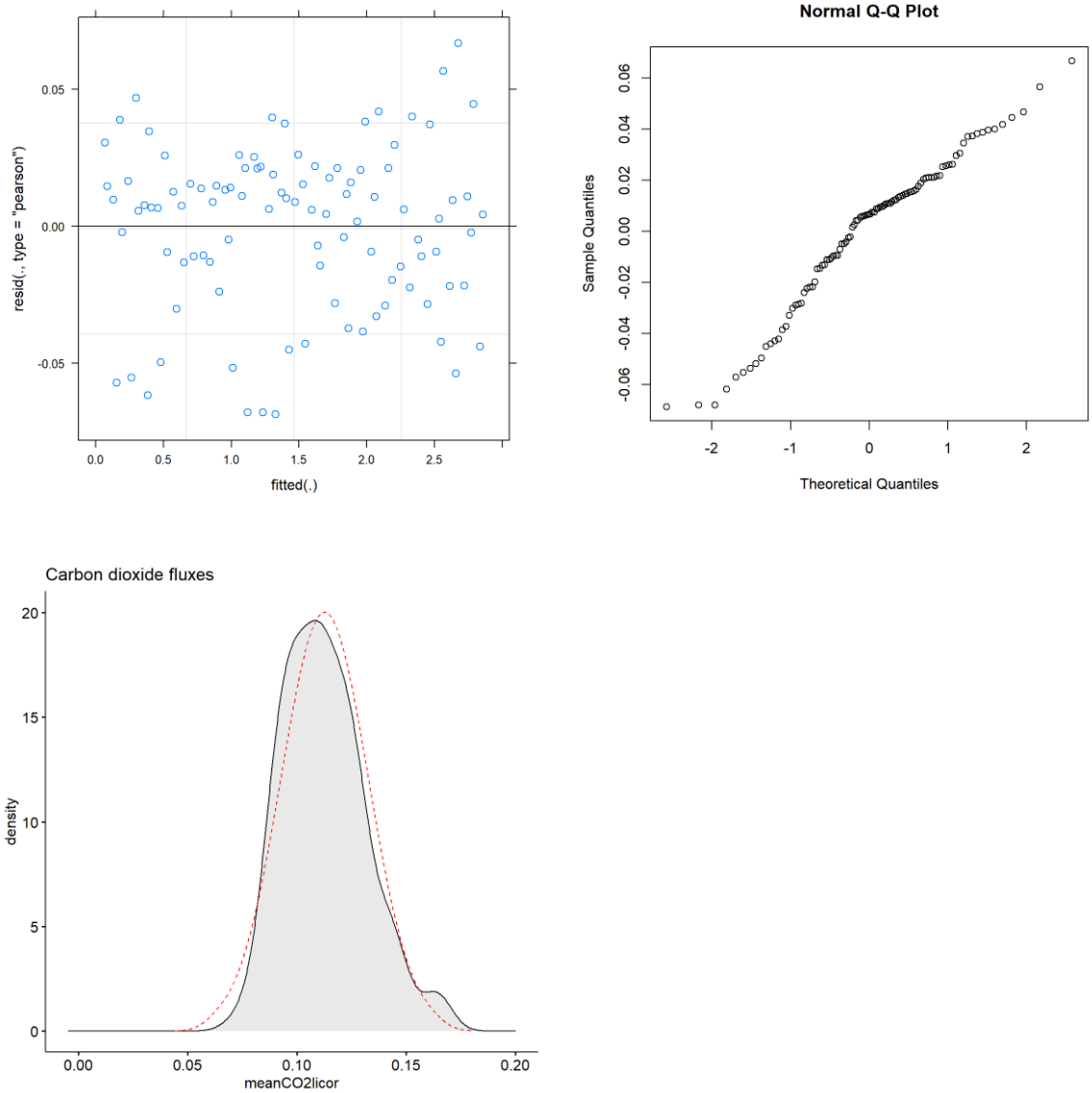


Figure 0.14 The data used for the cumulative statistics showed that the CO_2 data was normally distributed. Top left shows the residual, top right shows the QQ Normal plots, and bottom right shows the skewness distribution.

Chapter 4: UK population growth

The UK population is currently at 67.1 million people with a steady growth rate of 0.41% annually (ONS, 2021). Under conservative estimates (Figure 0.15), by the year 2100, the population is estimated to be close to 80 million, but this could be closer to 95 million if the current trajectory is maintained (United Nations, 2019).

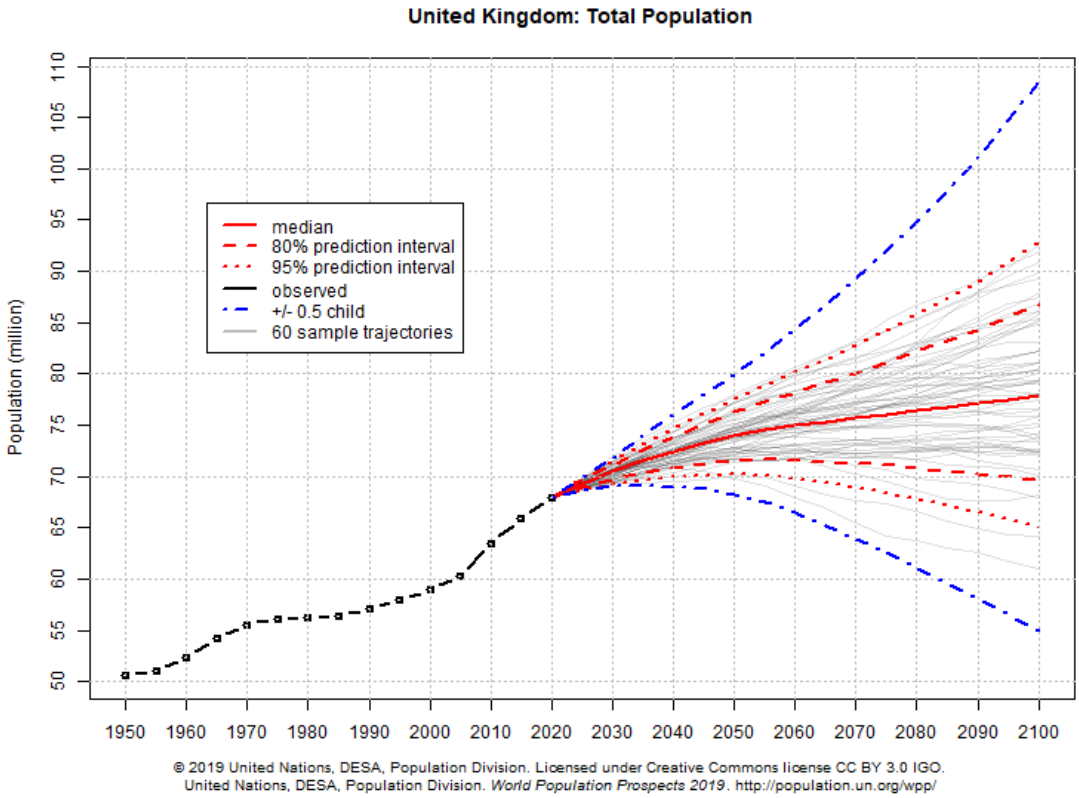


Figure 0.15 UK population growth trajectory up to the year 2100. The estimated median puts the UK population at just under 80 million people. However, if the current growth rate is maintained, this figure could end up closer to 95 million people.

Appendix F

Chapter 4: Power outage

During the study, there was a power outage at the site that stopped the instruments from collecting data. However, the instrument has an auto-resume function, so it resumed data collection when the power was restored to the site. These outages occurred on the following dates shown in Table 0.3. Since it was not possible to regularly attend to the equipment, it is not possible to ascertain the reason for the loss of power. Power cuts are not entirely unexpected in Abergwyngregyn, so the loss of power is most likely due to a power cut rather than equipment failure. The longest time the equipment was offline due to power loss was one hour and forty-four minutes. As expected, the equipment resumed data collection when the power supply was restored. The time the equipment was offline is limited so did not affect the overall outcome of the study.

Table 0.3 Dates and times when the equipment lost power.

Date	Time power lost	Time power restored	Elapsed time
19/06/2020	09:19:00	10:53:00	01:34:00
08/07/2020	13:29:00	14:53:00	01:24:00
15/07/2020	13:29:00	13:53:00	00:24:00
17/07/2020	12:06:00	12:53:00	00:47:00
12/08/2020	11:06:00	11:53:00	00:47:00
13/08/2020	07:32:00	07:53:00	00:21:00
19/08/2020	11:09:00	12:53:00	01:44:00

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