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Greenhouse gas flux response to restoration management in UK lowland peatlands

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Greenhouse gas flux response to restoration management in UK lowland peatlands

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by

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Acknowledgements

Summary

Peatland restoration seeks to re-establish a number of ecosystem services, with carbon (C) sequestration key following the inclusion of emissions from rewetted peatlands in national GHG reporting. Few studies address the effects of restoration on GHG fluxes or quantify annual emissions from peatlands undergoing restoration. This research focuses on the response of CO_2 and CH_4 fluxes from degraded lowland peatlands in Wales to restoration measures; vegetation removal, topsoil removal and rewetting. Fluxes were quantified from three locations using closed-chamber measurements, with annual fluxes estimated using empirical models. The effect of vegetation removal on CO_2 and CH_4 fluxes was closely coupled to water table depth, with the method of vegetation removal determining plant community composition. The removal of nutrient-enriched topsoil resulted in a net C emission, but this was assumed to be temporary as a small net C sink was observed where vegetation recolonisation occurred. The effect of rewetting was studied at a cutover raised bog and a grassland on peat. CO_2 and CH_4 fluxes from microforms created during historical hand-cutting were analogous to natural hummock-hollow complexes; CO_2 fluxes were greater from higher drier areas, whereas lower wetter conditions favoured higher CH_4 emissions. Rewetting at the cutover bog increased CH_4 fluxes where the water table was close to the surface and aerenchymatous vegetation was present, however under inundated conditions where vegetation was absent, CH_4 fluxes were relatively low. The rewetted grassland on peat was a significantly stronger CO_2 sink compared to a shallow-drained grassland, attributed to high nocturnal CO_2 emissions from the latter. CH_4 emissions were greater from the rewetted grassland, though the high CO_2 uptake resulted in a net GHG balance. Annual emissions derived from this research are comparable to UK lowland peatlands, but lower than IPCC emission factors, highlighting the importance of deriving UK-specific emission factors for future reporting.

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Abbreviations

a	Year
AWS	Automatic weather station
AFOLU	Agriculture, forestry and other land use
C	Carbon
Ch.	Chapter
CH_4	Methane
CO_2	Carbon dioxide
CO_2-eq	Carbon dioxide equivalents
Defra	Department of Agriculture, Food and Rural Affairs
DOC	Dissolved organic carbon
EC	Electrical conductivity
EF	Emission factor
GHG	Greenhouse gas
GPP	Gross primary productivity
GWP	Global warming potential
IPCC	Intergovernmental panel on climate change
LULUCF	Land use, land use change and forestry
NEE	Net Ecosystem Exchange
O_2	Oxygen
PAR	Photosynthetically active radiation
POC	Particulate organic carbon
R_{ECO}	Ecosystem respiration
SE	Standard error
W	Watt

Introduction

There has been a rise in atmospheric GHG concentrations since the Industrial Revolution, attributed to anthropogenic activity, and is strongly linked to global climate change (IPCC, 2014). The agriculture, forestry and other land use (AFOLU) sector, in which peatland GHG emissions are included, is estimated to account for 24% (~10 to 12 Gt CO_2-eq a^{-1}) of all anthropogenic GHG emissions (Smith et al., 2014). In 2015, the agriculture sector was the fifth largest source of GHG emissions within the UK, accounting for 10 % of national GHG emissions (49.1 Mt CO_2-eq). Although the Land Use, Land Use Change and Forestry (LULUCF) sector acted as a small net sink (-7.4 Mt CO_2-eq) during 2015, wetlands were a net source of GHGs (Evans et al, 2017b).

Peatlands are long term stores of carbon, acting as net sinks of atmospheric carbon in their natural state due to high water tables limiting aerobic decomposition of soil organic matter. Globally organic soils are estimated to store 600 Gt of carbon (Wilson et al., 2016a) whilst covering only 3% (~4 million km^2) of the Earth's surface (Joosten 2009). The usage of peatlands (e.g. for agriculture, forestry or fuel production) typically involves drainage which increases organic matter decomposition and accelerates carbon loss through CO_2 emissions, whilst CH_4 emissions decrease. Approximately 10% of all GHG emissions from the AFOLU sector have been attributed to drained organic soils (Smith et al., 2014), making them a significant source of GHGs to the atmosphere. The UK is within the top 20 countries with the largest CO_2 emissions from degrading peat, estimated at 9.6 Mt CO_2 a^{-1} (2008 data; Joosten 2009). Although many drained peatlands are still under agricultural production or peat extraction (though the latter is reducing), there is an increasing effort to restore abandoned peatlands with the primary intervention being rewetting. The raising of the water table seeks to reduce CO_2 emissions however higher water tables are also associated with higher CH_4 emissions, which has a radiative forcing at least 25 times greater than CO_2 (Forster et al., 2007). The GHG flux dynamics of rewetted peatlands may still differ from those of pristine peatlands, even if water levels are similar due to the impact of drainage on the soil. Although emissions from organic soils are specifically considered in the IPCC Wetlands Supplement (IPCC, 2014) and divided into wet or drained organic soils, there is no disaggregation of peatlands based on land use management. The wide range of land use on dry and wet organic soils, together with differences in local conditions such as climate and soil, may result in considerable errors using this approach to calculate GHG emissions.

UK peatlands cover just over 17 000 km^2 (~15 % of total land area), with an estimated carbon stock of 32 000 ± 300 Mt C (Worrall et al., 2010). Peatlands are predominately found in the UK uplands, with many areas subjected to drainage to improve grazing for sheep. In contrast, lowland peatlands account for ~15% of the total UK peat area, and have been subjected to more intense land use pressure compared to the uplands due to their potential for agricultural production and peat extraction following drainage (Evans et al., 2017). Due to the disproportionate pressures on lowland peatlands, it is estimated they may account for up to 50% of UK peatland GHG emissions (Worrall et al., 2012), however past research has predominately focused on upland blanket bogs. The Defra Lowland Peat Project sought to address this discrepancy, measuring GHG budgets from a range of lowland peatlands from near-natural fens and grasslands on peat to arable peatlands and extraction sites. The sites within the Lowland Peat Project were under long-term stable management, however restoration efforts of abandoned agricultural and cutover lowland peatlands across the UK involves a wide range of management interventions not captured by the project (e.g. vegetation management and removal of eutrophic topsoil). Furthermore, the two Welsh peatlands included in the Lowland Peat Project were both near-natural fens, however just over a quarter (~27.6 %) of Welsh lowland peatlands have been modified, with over half of this lowland bog (Blackstock et al., 2010).

The UK has both national and international commitments to reducing GHG emissions. Reduction targets set out by the Kyoto Protocol demand a 20% reduction in GHG emissions by 2020 compared to the 1990 baseline (UNFCCC); a joint commitment between European Union countries. Additional domestic legislative targets have been made through the UK Climate Change Act (2008), which requires GHG emission reductions of 34 % by 2020, and at least 80 % by 2050 compared to the 1990 baseline (UK Parliament, 2008). The achievement of these reduction targets is essential for mitigation against global warming and it is therefore crucial to address the knowledge gaps with regards to national estimates of GHG emissions. Improving our knowledge of GHG fluxes from UK lowland peatlands under restoration will not only inform future restoration efforts with regards to the effects of management on GHG fluxes, but will also aid accurate quantification when reporting national GHG estimates.

The primary aim of this thesis is to investigate the response of CO_2 and CH_4 fluxes to restoration management in Welsh lowland peatlands. In order to achieve this aim, a number of research questions and specific objectives will be investigated.

The specific research questions are:

1. To investigate the effect of restoration interventions, namely vegetation removal and topsoil removal, on CO_2 and CH_4 fluxes from a temperate lowland fen.
2. To investigate the effect of rewetting through the construction of low-lying bunds on CO_2 and CH_4 fluxes from a cutover lowland raised bog
3. Do GHG fluxes from microtopography created by hand cutting of peat behave in the same way as hummock-hollow complexes in natural peatlands?
4. To investigate the effect of rewetting on CO_2 and CH_4 fluxes from a grassland on peat.
5. Are diurnal fluctuations in CO_2 fluxes higher in shallow-drained grassland on peat compared to a rewetted grassland on peat?

The specific research objectives which will be met to answer the above research questions are:

1. To measure CO_2 and CH_4 fluxes from peatlands under restoration management, as well as unrestored peatlands.
2. To identify the controlling environmental drivers of CO_2 and CH_4 fluxes.
3. To produce annual CO_2 and CH_4 balances for the peatlands studied using modelling with environmental drivers.
4. Compare annual CO_2 and CH_4 balances to the Defra Lowland Peat Project, IPCC Tier 1 emission factors and draft UK specific Tier 2 emission factors.

The thesis structure is summarised below. The individual chapters address the research questions and objectives above as described below.

Chapter one reviews the published and grey literature. The biogeochemistry of lowland peatlands with regards to the carbon balance is discussed, together with the effects of peatland modification and restoration techniques on CO_2 and CH_4 fluxes. The methods used to measure and calculate CO_2 and CH_4 fluxes are reviewed, followed by a discussion on upscaling flux measurements to annual carbon budgets. Finally, the importance of peatland carbon budgets in national GHG reporting is considered.

Chapter two addresses the first research question and comprises of a field study investigating the effects restoration interventions on the CO_2 and CH_4 fluxes of a tall fen, a short fen and a restored pasture. CO_2 and CH_4 fluxes were measured on a monthly basis using the closed-chamber method on sites subjected to vegetation removal through hand cutting and burning as well as topsoil removal. Corresponding unmanaged, semi-natural control sites were also monitored. The environmental conditions driving fluxes were identified and used to model annual CO_2 and CH_4 flux estimates for each restoration intervention.

Chapter three focuses on the effects of rewetting on CO_2 and CH_4 fluxes from a cutover lowland raised bog using the closed-chamber method; the second research question. Comparisons are made between rewetted and non-rewetted areas of the cutover bog for one growing season as well as the before and after effects of rewetting. The experimental design included gas flux measurements from microtopographic features created when the bog was cut by hand for peat to address the third research question.

Chapter four compares CO_2 and CH_4 fluxes from a shallow-drained and rewetted grassland on peat in order to answer the fourth research question. Automated chambers were used to measure fluxes on a 1.5 hourly basis, enabling the diurnal pattern in CO_2 fluxes to be identified and address the fifth research question. Annual CO_2 and CH_4 fluxes were calculated from empirical models driven by environmental measurements.

Chapter five provides a summary of the key findings from each field study and addresses the strengths and weakness of the chamber methods used. The work is analysed within the wider context through comparing annual emissions measured within this study with global default and UK specific emission factors, as well as the recent Defra Lowland Peat Project with regards to the potential development of Tier 3 emission factors. Finally, the potential policy implications of this work are highlighted, together with recommendations for future research.

Chapter 1

Greenhouse gas fluxes from semi-natural peatlands: controls, measurements and restoration effects

1.1. Definition of peatlands

Peatlands are the most extensive type of wetland habitat, accounting for 50 to 70 % of global wetlands (Joosten and Clarke, 2002). The waterlogged nature of peatlands together with recalcitrant plant litter creates conditions highly unfavourable for decomposition. This imbalance between photosynthesis and slow decomposition rates, results in the accumulation of partially decomposed plant remains, with carbon being retained within the highly organic soils formed. A variety of definitions are used to classify peat soils, with organic matter contents varying from 20-25 % for 'peaty' soils to over 50 % for 'peat' (Evans et al., 2011), whilst the minimum peat depth required for 'peatland' classification ranges from 30 cm to 1 m (Lindesay et al., 2014). The vertical peat profile is typically divided into two layers; the permanently saturated catotelm containing largely decomposed plant remains, and the oxic surface acrotelm layer where material is only partially decomposed and periodically inundated (Clymo, 1984).

Peatlands can be broadly classified into two categories; bogs or fens, depending on the origin of inputs into the ecosystem. Ombrotrophic bogs receive nutrient and water inputs solely from precipitation or atmospheric moisture, resulting in a highly acidic and low nutrient environment (Lai, 2009), whereas minerotrophic fens receive additional inputs from groundwater flow and thus are more base rich (Holden, 2005). Bogs can be further divided into either blanket bogs or raised bogs, characterised by their underlying topography where the former comprises of a 'blanket' of peat covering the landscape and the latter consists of a dome of peat overlying an infilled lake or fen (Baird et al., 2009). Fens on the other hand are found in depressions in the landscape, for example, forming basin fens in topographic hollows or valley fens and floodplain fens (Charman, 2002). The greater nutrient content of fens compared to bogs, as well as differences in elevation, leads to contrasting plant community composition (Lai, 2009) and thus distinct ecosystems which require separate consideration.

1.2. Peatland extent

Global peatland extent is estimated to be 4 million km^2, with the majority of peatlands found within northern temperate regions (Joosten, 2009) though tropical peatlands are considered to hold up to a quarter of the global peat volume (Page et al., 2011). Furthermore, recent discoveries of large peat deposits within central Africa suggest that the global peatland inventory is still incomplete (Dargie et al., 2017). It is estimated up to 20 % of the global peat

area has been lost through climate change and human activity since 1800 (Joosten and Clarke, 2002), with the latter predominately through drainage for agriculture, forestry and extraction for fuel provision. European peatlands account for approximately 12 % of the global peatland area (51 million ha) yet have suffered the largest loss of mire extent (Joosten and Clarke, 2002). Within the UK, peatlands cover an area of ~ 23 000 km^2 (Fig. 1.1.), however, shallow and deep peaty soils are much more extensive (~80 000 km^2) indicating past peatland habitats were once more widespread (Evans et al., 2011).
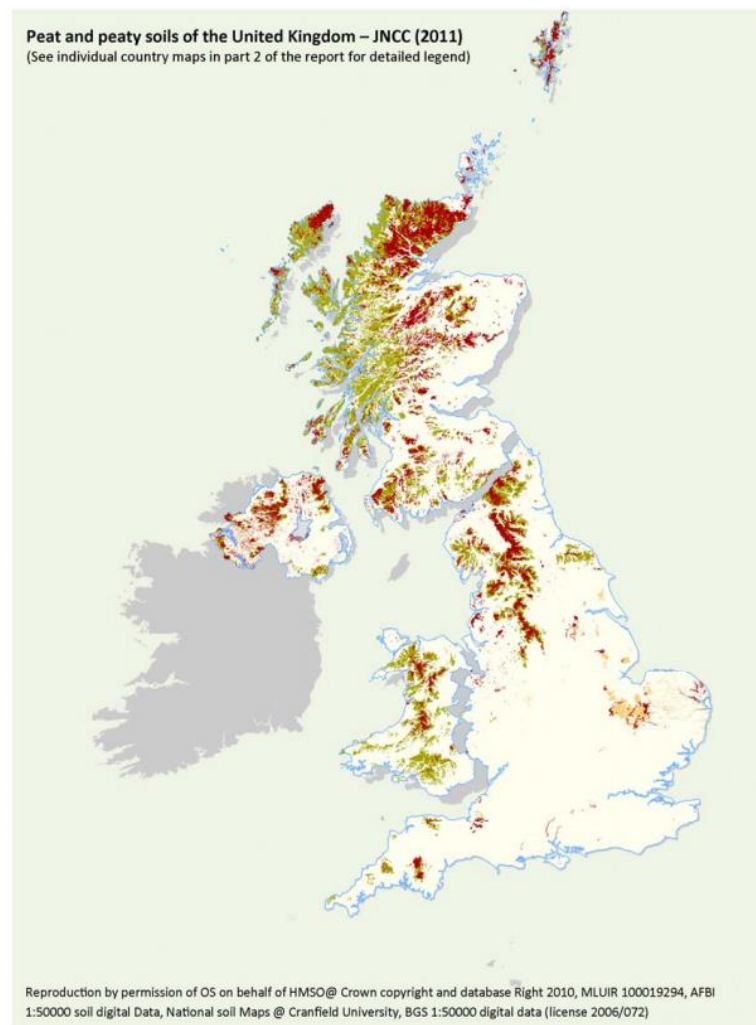
Figure 1.1. Distribution of peat and peaty soils in the UK (Evans et al., 2011).

Peatlands within the UK are estimated to hold 3.2 billion tonnes of carbon (Billett et al., 2010) with blanket bogs the most extensive type of UK peatland, followed by raised bogs and fens. As with peatlands globally, UK peatland extent (both spatially and peat depth) has declined as a result of human influence. In England less than 1 % of deep peatlands are considered as

undamaged, with ~28 % classified as wasted arising from uses such as crop cultivation and pasture (Table 8; Evans et al., 2011). Whilst the cultivation of peatlands in Wales is less prevalent, 44 % of Welsh peatlands are classed as modified, having been subjected to drainage for afforestation, grazing, arson fires and for peat-cutting for fuel provision (Blackstock et al., 2010). Similarly, all peatland types within Scotland and Northern Ireland are considered to be in slow decline (Evans et al., 2011).

1.3. Peatlands and the carbon cycle

Peatlands play a crucial role in the global carbon cycle, naturally acting as a net sink of atmospheric carbon due to the imbalance between photosynthesis and ecosystem respiration. Atmospheric CO_2 is taken up from the atmosphere by vegetation through photosynthesis (Fig. 1.2.), though only a small proportion of carbon is sequestered in accumulating peat (Frolking et al., 2006). Carbon is lost from peatlands through the decomposition of organic matter. Aerobic autotrophic (plant) and heterotrophic (microbial) respiration occur within oxic zones of the peat profile, producing CO_2, which diffuses through the peat profile to the atmosphere. (Clymo and Pearce, 1995). The combination of these processes is known as ecosystem respiration (R_{ECO}). CO_2 can also be produced through the oxidation of CH_4 by methanotrophic bacteria within aerobic pockets of the peat profile such as the rhizosphere (Watson et al., 1997). The balance of CO_2 uptake (gross primary productivity, GPP) and R_{ECO} is known as the net ecosystem exchange (NEE). Due to the saturated, and often cool conditions of natural peatlands, decomposition is restricted and thus plant litter accumulates on the surface, being buried by further litter and incorporated in the peat profile. Although northern peatlands have relatively low CO_2 exchange rates compared to other ecosystems, it is the persistent dominance of GPP over R_{ECO} which has led to northern peatlands acting as a net CO_2 sink over the past 5000 – 10 000 years. Over this time it is estimated 200 – 450 Pg C have been sequestered by northern peatlands at an average rate of 0.02 – 0.03 kg C m^{-2} a^{-1} (Frolking et al., 2001). Although this is inclusive of UK peatlands, the maritime climatic conditions are likely to cause greater interannual variability in carbon exchange compared to northern boreal and sub-arctic peatlands (Billett et al., 2010).
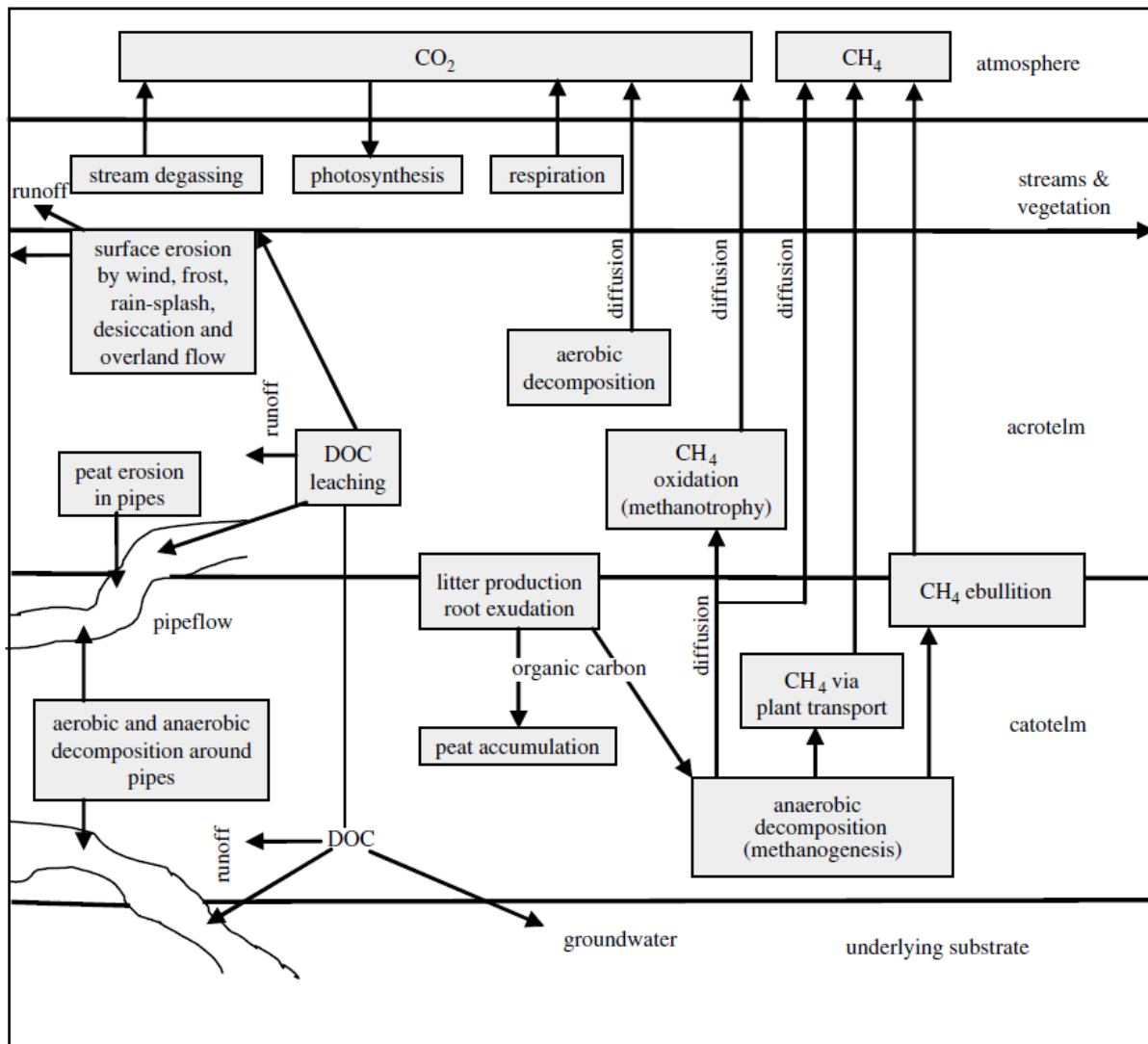
Figure 1.2. Schematic summary of the carbon cycle in peatlands (Holden, 2005).

Wetlands contribute the largest individual source of microbial CH_4 to the global CH_4 cycle; 69% (Conrad, 2009), with an estimated $20 - 50$ Tg a^{-1} of CH_4 emitted from northern peatlands (Mikaloff Fletcher et al., 2004). CH_4 is produced by the reduction of organic matter in strictly anaerobic conditions by methanogenic microorganisms (Van den Pol-Van Dasselaar et al., 1999), with three main pathways of release to the atmosphere as shown in Fig. 1.3. Firstly, CH_4 can diffuse vertically through the peat profile along a concentration gradient from the site of production to the atmosphere. Optimum CH_4 production is found at the water table boundary and thus diffusion of CH_4 through aerobic layers or pockets within the peat profile increases the likelihood of oxidation by methanotrophic bacteria, producing CO_2 (Van den Pol-Van Dasselaar et al., 1999; Lai, 2009). The thickness of the aerobic peat layer, as well as the

presence of aerobic pockets within the rhizosphere, therefore controls the proportion of CH_4 produced which reaches the atmosphere through diffusion (Frenzel & Rudolph, 1998). Secondly, CH_4 can undergo plant mediated transport by vegetation which possess aerenchyma; specialised tissue which aids gas exchange between the leaves and roots (Rydin and Jeglum, 2006). Whilst the primary function of aerenchyma is to transport O_2 to the roots, CH_4 may also be transported from the anoxic zone of production directly to the atmosphere, bypassing the oxic zone, with the plants acting as a chimney or conduit (Whalen, 2005). In addition to this passive transportation, high CH_4 fluxes have been associated with the convective through flow in *Phragmites* plants, where a humidity induced pressure gradient results in pressurized air flow from the sheaths into the rhizomes and back to the atmosphere through dead stems still connected to the rhizomes (van den Berg et al., 2016). Finally, CH_4 often forms gas bubbles in saturated peat due to its low solubility. CH_4 bubbles may accumulate within the peat profile where pore diameters are too small for the bubble to pass through, potentially blocking the release of other smaller bubbles (Strack et al., 2005). The release of bubbles to the atmosphere is known as ebullition, and may be triggered by changes in atmospheric pressure, the lowering of the water table or high turbulence (Strack et al., 2005; Tokida et al., 2007; Hendriks et al., 2010). Although ebullition may significantly contribute to CH_4 emissions from peatlands, ebullition events have been found to be highly variable both spatially and temporally and thus are a challenge to quantify (Tokida et al., 2007; Stamp et al., 2013).
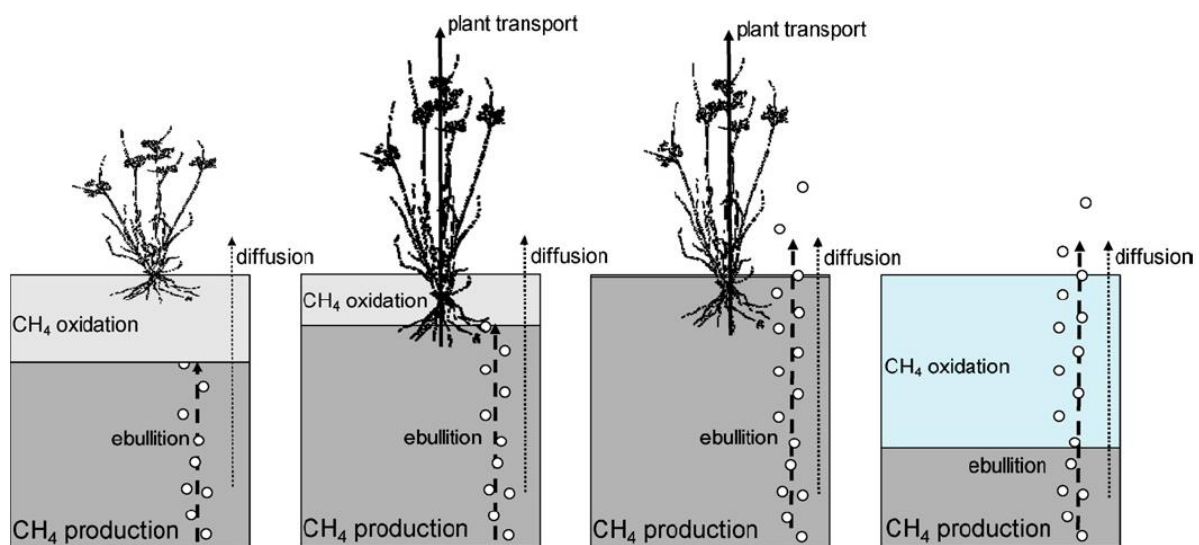
Figure 1.3. CH_4 transport mechanisms (Hendriks et al., 2010). CH_4 production occurs in anaerobic peat (dark grey) and is subsequently oxidised as it diffuses through aerobic peat (light grey). CH_4 may be transported directly to the atmosphere via aerenchymatous tissue in

vegetation, potentially bypassing the oxic peat or CH_4 bubbles may be released where the water table reaches the surface through ebullition.

Carbon losses from peatlands also occur through aquatic fluxes with many peatlands, particularly in the uplands, having streams supersaturated in CO_2 and CH_4 which may be lost through evasion (Hope et al., 2001). Dissolved organic carbon (DOC) makes up the largest of the aquatic fluxes, predominately exporting carbon downstream and out of the catchment, though some is also oxidised to CO_2 and lost to the atmosphere through evasion. DOC production is regulated by the availability of soluble carbon within the soil as well as vegetation with some vegetation producing more labile carbon than others (e.g. in root exudates). Finally, physical erosion within peatlands results in particulate organic carbon (POC) losses, which is either exported via streams or windblown. Although a minor component of the peatland carbon budget, POC is highly variable in both space and time, with increases observed in response to storm flow events (Hope et al., 1997). Furthermore, the POC flux is likely to be higher from disturbed peatlands compared to those in semi-natural or pristine condition and therefore could be an important component of the carbon balance, though there is still a lack of understanding regarding its fate in terms of GHG budget (Worrall et al., 2010).

The carbon balance of an ecosystem is the total of all the carbon fluxes entering and leaving a specific landscape unit (Billett et al., 2010), with the wide range of carbon species present (gaseous, dissolved and particulate) making it a challenge to comprehensively quantify. In addition to this, there are a number of pathways by which carbon may enter or leave the ecosystem, with few studies measuring all components and pathways of the carbon budget. In the past many peatland carbon budget studies only measured during the growing season, neglecting winter carbon losses and therefore the annual switch between source and sink (Roulet et al., 2007). Finally, temperate climates, particularly in maritime regions are susceptible to considerable interannual variability both natural and climate change induced meaning prolonged data sets are required to accurately produce a peatland carbon balance and identify long term trends (Billett et al., 2010).

1.4. Controls on CO_2 and CH_4 fluxes from peatlands

The large scale components of peatlands such as their climatic zone and hydrogeomorphic context determine local environmental conditions which drive carbon and nutrient cycling,

including vegetation, hydrology and soil characteristics (Bridgham et al., 2013). This section will explore the principal controls on CO_2 and CH_4 gaseous fluxes from peatlands with regards to their magnitude and patterns on diurnal to seasonal cycles.

1.4.1. Temperature

Temperature is often found as the dominant control on variations in R_{ECO}, with higher temperatures stimulating microbial activity and therefore rates of CO_2 emissions (Lloyd and Taylor, 1994; Silvola et al., 1996; Lafleur et al., 2005). The increase in CO_2 emissions with temperature was initially described as an exponential relationship (Q10), however studies have shown that above certain temperatures R_{ECO} fluxes plateau or start to decline (e.g. Juszczak and Augustin, 2013); indicative that other controls are limiting R_{ECO}. The temperature sensitivity of R_{ECO} has also been shown to vary due to the interaction with other environmental variables such as substrate quality (Blodau et al., 2004) and microbial community structure (Makiranta et al., 2009). Temperature has a similar control on the microbial production of CH_4, with increased rates of CH_4 production at higher temperatures under anoxic conditions (Frenzel and Karofeld, 2000; Bubier et al., 2005). Similarly, methanotrophic activity (CH_4 consumption) is considered to increase with increasing temperatures, though a mesocosm study by van Winden et al. (2012) suggested that higher rates of CH_4 oxidation cannot match the concurrent increase in CH_4 production.

1.4.2. Water table depth

The position of the water table is considered to be one of the most important regulators of wetland biogeochemistry (Waddington & Roulet, 1996; Ellis et al., 2009), controlling the size of aerobic and anaerobic zones within the peat profile and therefore the magnitude and type of decomposition occurring (Clymo, 1984). A lowering of the water table increases O_2 diffusion into the soil, facilitating aerobic decomposition as well as root respiration, increasing CO_2 production and therefore emission (Bubier et al., 2003; Riutta et al., 2007; Juszczak et al., 2013). Although greater CO_2 emissions with lower water tables have been reported under laboratory conditions (Waddington et al., 2001; Blodau et al., 2004), the interaction of water table with temperature, vegetation and microtopography complicates the response of CO_2 fluxes to water table experiments under field conditions. Lafleur et al. (2005) reported a weak correlation between water table depth (WTD) and R_{ECO} from a Canadian ombrotrophic bog which was attributed to a consistently low water table (usually at least 30 cm below the surface), and thus changes in water table had little effect on the soil moisture conditions of the uppermost peat profile where CO_2 production is greatest. In contrast, R_{ECO} has also been found

to be insensitive to water table fluctuations near the surface, potentially due to fluctuations being too small (< 7 cm) to significantly alter optimum conditions for respiration (Juszczak et al., 2013). Where water table changes have been long term, for example the rewetting of drained peatlands, R_{ECO} under rewetted conditions has been significantly lower compared to that from drained peatlands, substantially reducing net carbon losses (e.g. Strack and Zubak, 2013; Tuittila et al., 1999).

Methanogenesis relies upon anaerobic conditions and thus a water table closer to the surface often leads to larger CH_4 emissions due to the greater thickness of the production zone and reduced opportunity for oxidation in the smaller overlying aerobic layer (Lai, 2009). Indeed, a rewetting mesocosm experiment by Dinsmore et al. (2009) showed higher CH_4 emissions from mesocosms under high water table conditions, with a pulse of CH_4 observed between 1 and 2 days after rewetting. A lowering of the water table reduces the size of the anaerobic zone within the peat profile and thus reduces potential methanogenic activity whilst increasing methanotrophic activity. Strack et al. (2006) found lower CH_4 emissions following experimental drawdown of the water table in a Canadian poor fen, however, high temporal and spatial variability in CH_4 fluxes together with changes in vegetation composition obscured the relationship between water table and CH_4 flux. A meta-analysis of GHG emissions from temperate European peatlands found annual CH_4 fluxes to be negligible (< 2 kg CH_4 ha^{-1} a^{-1}) where water tables were at least 20 cm below the surface, however, emissions were also low where there was a lack of aerenchymatous vegetation (Couwenberg et al., 2011). Despite high variability of CH_4 fluxes from field studies, the mean annual water table level has been identified as the best single explanatory variable of annual GHG fluxes from a global peatland meta-analysis (Couwenberg et al., 2011).

1.4.3. Microtopography

The peatland surface is comprised of small features known as microforms which have different functional, hydrophysical and ecological properties (Belyea & Baird, 2006). The underlying nature of the landscape topography encourages preferential accumulation of peat, resulting in the development of hummock and hollow microforms (Holden, 2005). In combination with different rates of production and decomposition of organic matter, the distinct properties of these microforms becomes embedded within the newly forming peat (Bubier et al., 2006), altering the soil chemistry and resulting in their continual development and persistence, eventually creating a self organised peatland structure at the microtopographic scale (Belyea

& Baird, 2006). Microforms create a difference in elevation across the peatland surface on a scale of tens of centimeters, leading to small scale heterogeneity in the water table depth, as well as microclimatology and consequently vegetation composition (Bubier et al., 2006). The influence of microtopography on water table position creates a gradient from higher and drier areas such as hummocks where the acrotelm may be more than 20 cm thick to lower and wetter microforms including hollows and pools, with an acrotelm of only a few millimeters (Frenzel & Karofeld, 2000). Subsequently the thickness of the oxic layer and associated microbial decomposition is also affected by microtopography, with typically higher CH_4 and lower CO_2 emissions from hollows and vice versa from hummocks.

1.4.4. Biotic controls: Enzymes

Peat decomposition is ultimately controlled by soil microbial processes and the associated community composition (Preston et al., 2012), which differs across both geographic and microsite scales (Mitchell et al., 2003). Although peatland microbial community composition remains relatively understudied, enzyme activity in wetland soils has been well explored, providing an indication of carbon cycling rates (Dunn et al., 2014). Together with gaseous products such as CO_2 and CH_4, peatland microbial processes release soluble products including phenolics (humic) and organic acids (Freeman et al., 2004). Phenol oxidase acts to catalyse the oxidation of recalcitrant phenolic compounds into readily available substrates, however, is limited by the low O_2 availability in saturated peat soils. The buildup of recalcitrant materials such as lignin inhibits the activity of hydrolase enzymes which are responsible for substrate decomposition in peat soils. The role of phenol oxidase as an enzymic 'latch' preventing peat soils from releasing stored carbon relies upon anaerobic conditions being maintained, with increases in enzymic activity observed following drainage (Freeman et al., 2001). Whilst it would be expected that raising the water table would suppress phenol oxidase activity, this was not observed by Peacock et al. (2013) following ditch blocking in a Welsh blanket bog, which was attributed to constant soil moisture conditions post blocking and the potential legacy of enhanced enzyme activity stimulated by previous aerobic conditions (Fenner et al., 2011).

1.4.5. Plant functional type

The vegetation component of peatlands has an important influence on gas exchange, as well as the peat structure through the chemical and physical composition of litter inputs. Peatland vegetation has specific functional traits to tolerate waterlogged anoxic conditions as well as the low nutrient availability and cold, slightly acidic conditions (De Deyn et al., 2008). The spatial distribution of vegetation throughout the peatland landscape is dependent on a number of

environmental factors such as water table depth, pH and electrical conductivity (EC), with the former being the strongest driver of plant species composition (Bubier et al., 2006). To aid vegetation assessments at a landscape level, categories of plant functional type (PFT) are used rather than individual species; a necessary simplification for ecosystem modelling especially when considering plant response to global climate change (Bubier et al., 2003). UK peatlands are typically dominated by three PFT; ericoid dwarf-shrubs, graminoids and bryophytes, each with specific adaptations which impact the exchange of greenhouse gases between the peat soil and atmosphere.

Ericoid dwarf shrubs, such as *Calluna vulgaris* are long lived, retaining carbon in their woody shoots and producing nutrient poor, recalcitrant litter (De Deyn et al., 2008). They dominate degraded peatlands due to their ability to withstand acidic, low nutrient environments. Ericoid tissues are also rich in phenolics which inhibit decomposition through the enzymic control described above (Ward et al., 2009). Removal experiments by Ward et al. (2009) found that *Ericoids* had the greatest effect on CO_2 fluxes; their presence hindering the photosynthetic ability of other peatland vegetation such as graminoids, most likely due to shading effects.

In contrast to shrub vegetation, graminoids have the highest NEE within peatland ecosystems (Clay et al., 2012), sequestering more carbon than other PFTs (Bubier et al., 1999). Graminoids are characterised by deep roots which penetrate the anoxic zone and aerenchyma; specialised stem tissue which aids the transportation of gases between the atmosphere and roots (Lai, 2009). Aerenchyma has a considerable influence on the peatland carbon budget, acting as a conduit for CH_4 to be transported directly to the atmosphere. Numerous experiments have reported increased CH_4 emissions from graminoid dominated plots e.g. Greenup et al. (2000), Tuittila et al. (2000) and Frenzel & Rudolph (1998). Sedge biomass was found to explain 80% of the variation in CH_4 fluxes between plots in field campaigns carried out by Bellisario et al. (1999) across a large peatland complex in Manitoba, Canada. The significant relationship found between CH_4 emissions and net ecosystem production (NEP) strongly indicates plant mediated activities are an important driver within *Carex*-dominated peatland communities (Bellisario et al., 1999), likely due to the presence of a vascular transport system. In addition to this, the supply of CO_2 to the roots is increased, providing substrate for methanogens in the anoxic zone to produce CH_4 (Greenup et al., 2000).

Bryophytes are the final PFT to be considered here, and include non vascular plants such as *Sphagnum* species, which acquire nutrients through rainfall and symbiotic relationships with N fixing bacteria (Ward et al., 2012). Bryophytes are adapted to waterlogged conditions, commonly found in hollows and their distribution across the landscape follows the water table position due to their low tolerance for dry conditions (Bellisario et al., 1999). Mosses are less productive than most peatland vegetation, and their slow rate of decomposition and ability to grow indefinitely (Clymo & Pearce, 1995) makes them one of the most important peat forming species (Fenner et al., 2007).

1.5. Peatland modification

Globally, peatlands have been heavily utilized for agriculture, forestry and peat extraction for fuel and horticulture (Chapman et al., 2003), with additional pressures such as erosion, fire and urbanization also causing losses. Whilst the lack of accurate information regarding peatland extent means it is difficult to quantify the anthropogenic impact on peatlands, it is estimated ~ 16 % of the global peatland area has been lost through human induced activity and degradation (Joosten and Clarke, 2002). The saturated conditions of near natural peatlands means that drainage is typically the first step in peatland utilisation. Within the UK lowland peatlands have been subjected to greater pressure compared to upland systems, with drained peatlands principally used for agriculture and to a lesser extent peat extraction. This section will discuss the effects of drainage and subsequent agriculture and extraction on GHG fluxes.

1.5.1. Drainage

At the global scale, drained organic soils are estimated to account for 10 % of all GHG emissions from the agriculture, forestry and other land use (AFLOU) sectors; an estimated emission of 0.9 Gt CO_2-eq a^{-1} in 2010 (Smith et al., 2014). Drainage of UK peatlands intensified in the 17th century in order to improve lowland fens for agriculture (e.g. the East Anglian fens), whilst upland peatlands were drained more recently (1960s and 1970s) primarily for grazing (Holden et al., 2004). The lowering of the water table increases the size of the aerobic layer of the soil profile, with O_2 penetrating previously saturated areas of peat. This increases the area in which aerobic decomposition can occur, resulting in an increase in CO_2 produced and therefore released to the atmosphere (Blodau et al., 2004). As the water table drops, the zone of CH_4 production is moved down the soil profile and there is a greater opportunity for oxidation as CH_4 diffuses through the larger overlying aerobic layer.

1.5.2. Agricultural peatlands

Cultivated peatlands have among the highest CO_2 emissions from any land use type (Couwenberg et al., 2011) and therefore quantifying emissions is crucial for mitigating impacts on climate change whilst providing for an increasing population (Giltrap et al., 2010). In addition to increased CO_2 emissions driven by the lowering of water tables, agricultural practices such as soil tillage have been found to increase soil respiration rates through increasing aeration and substrate availability (Elder and Lal, 2008). Carbon losses also arise from the harvesting of crops in arable peatlands, as the carbon taken up during crop growth does not make it back into the soil. Although the rate of CO_2 uptake is found to differ between crops, for example CO_2 sequestration was greater from maize and wheat compared to lettuce in measurements from arable peatlands in the East Anglian Fens, the allocation of much of this carbon to above ground biomass means once harvesting has been taken into account, the net loss of carbon was similar (Evans et al., 2017). The removal of biomass and therefore carbon also occurs at agricultural grasslands on peat soils where hay cutting occurs, with the carbon lost through biomass being taken offsite a substantial proportion of the NECB (Beetz et al., 2013). Cutting also affects CO_2 exchange, with the loss of leaf area temporarily eliminating GPP which allows R_{ECO} to dominate the CO_2 balance. Generally, intensively managed grasslands on peat soils are found to be net sources of CO_2 with the pattern in NEE governed by the cutting regime (Beetz et al., 2013; Veenendaal et al., 2007).

Agricultural peatlands are typically associated with negligible CH_4 emissions due to the low water table level, with a small net uptake of CH_4 reported from some studies (Taft et al., 2017). However, measurements from drainage ditches have found extremely high CH_4 emissions; up to 366 mg CH_4 m^{-2} hr^{-1} in an intensively managed Dutch grassland on peat, likely created by a mixture of high wind and turbulent flow conditions (Schrier-Uijl et al., 2009). CH_4 emissions originating from ditches dominate the CH_4 balance in agricultural peatlands and are widely found to be significantly greater than those from fields, though variability is often high. (Hendriks et al., 2007; Schrier-Uijl et al., 2009; Peacock et al., 2017).

1.5.3. Peat extraction

Commercial peat extraction involves the removal of peat for horticultural and energy production, and occurs on an industrial scale across Central Europe and North America (Grand-Clement et al., 2015). Extraction of peat involves extensive drainage and the removal of the surface layer of vegetation to provide a bare surface from which peat is typically milled in a

number of layers up to $\sim$ 200 mm a^{-1} (Lindsay et al., 2014). The removal of vegetation eliminates CO_2 uptake as no photosynthesis occurs, leading to R_{ECO} dominating the CO_2 balance. Furthermore, extensive drainage leads to enhanced organic matter decomposition in the newly exposed aerobic layer and thus increased CO_2 emissions. A recent study by Wilson et al. (2013) quantified CO_2 emissions from peat extraction sites within the UK and Republic of Ireland (ROI) with annual estimates ranging from 93 to 304 g C m^{-2} a^{-1}. Similar annual emissions have been reported from other European peatlands under extraction; 267 g C m^{-2} a^{-1} in Estonia (Järveoja et al., 2016) and 697 g CO_2 m^{-2} a^{-1} in Finland (Maljanen et al., 2010). As with peatlands drained for agriculture, CH_4 emissions from peatlands under extraction are minimal due to the lower water table level and continual removal of the surface peat layers which contain microbial communities. Drainage ditches can, however, act as hotspots for CH_4 emissions, with ditch emissions from mined peatlands in Sweden ranging from 1.1 to 25 mg CH_4 m^{-2} hr^{-1} in a study by Sundh et al. (2000); significantly higher than from the bare peat surface (0.01 to 1 mg CH_4 m^{-2} hr^{-1}). In addition to CO_2 losses, bare peat surfaces are more susceptible to wind and water erosion, and losses of windblown particulate organic carbon may be exacerbated.

Peat extraction also occurs on a small scale, with peat cutting by hand for domestic fuel provision historically widespread throughout the UK, though this has generally declined and in some areas superseded by mechanical extraction. Whilst the drainage implemented for domestic peat extraction is on a much smaller scale than that of industrial sites, the lowering of the water table has similar effects on CO_2 and CH_4 emissions. Furthermore, drainage and domestic extraction at the margins of peatlands (typically lowland raised bogs) results in a lowering of the water table within the centre of the peatland, causing subsidence of the main dome and potentially increasing decomposition across a wider area (Wilson et al., 2015, Lindsay et al., 2014). Emission factors from domestic and industrial peat extraction sites in the ROI and the UK are similar; 1.70 and 1.64 t C ha^{-1} a^{-1} respectively (Wilson et al., 2015), though there is a lack of data regarding the extent and emissions from domestic extraction sites. Despite this, emissions are notably lower than the Tier 1 emission factor reported in the Wetlands Supplement (2.8 t C ha^{-1} a^{-1}).

1.6. Peatland restoration

Peatland modification for agriculture and extraction has led to degraded systems which have lost their hydrologic functioning, biodiversity and net carbon sink ability. The restoration of abandoned peatlands has therefore become critical to preserve existing carbon stocks for climate change mitigation, as well as to combat the loss of biodiversity, enhance flood alleviation and provide recreational spaces (Andersen et al., 2017). Peatland restoration broadly aims to re-establish peat forming vegetation, and thus a carbon sink function, as well achieving favourable nature conservation status (Lunt et al., 2010). However, the nature and level of restoration intervention needed to achieve this outcome differs depending on the degree to which the hydrology, peat characteristics and vegetation of the site have been modified. The principal areas of peatland restoration include water management (e.g. the blocking of drainage ditches), revegetation (of bare soil following peat extraction or arable harvesting) and vegetation management (e.g. removal of undesirable vegetation or trees), though in extreme cases topsoil removal may be required where the surface peat has become enriched from agricultural inputs. The following section will discuss the effects of these main restoration measures on the greenhouse gas fluxes.

1.6.1. Water management: Rewetting

Water management is considered the highest priority for restoring drained peatlands (Schumann and Joosten, 2008) with the rewetting, i.e. raising the water table, crucial for re-establishing a functioning peatland system. Rewetting is often achieved through the blocking of drainage ditches, either with plastic or peat dams, which minimizes runoff and creates localised ponding with water shedding laterally. Where peatlands have a low surface slope, such as in cutover lowland raised bogs, additional dams or bunds may be created to form a network of cells in which excess precipitation becomes impounded. The use of bund cells raises the water table within defined areas, creating conditions ideal for Sphagnum colonisation as well as buffering against summer drought (Money & Wheeler, 1999). Rewetting aims to create conditions where the soil is saturated or inundated for all or part of the year (IPCC, 2014), consequently inhibiting aerobic respiration and therefore reducing CO_2 emissions. Comparisons between shallow-drained and rewetted grasslands on peat show reductions in CO_2 emissions from the latter (e.g. Renou-Wilson et al., 2016 during ungrazed conditions), with emissions substantially lower than those reported for deeply drained systems (e.g. Renou-Wilson et al., 2014). The annual CO_2 balance of rewetted grasslands on peat varies widely within the literature, with those used for extensive grazing near often near neutral (e.g. Beetz

et al., 2013; Renou-Wilson et al., 2016), whilst those under no management can act as strong net CO_2 sinks (e.g. Hendriks et al., 2007).

Whilst CO_2 emissions are lowered following the creation of anaerobic conditions, CH_4 oxidation is also limited due to the reduced size of the aerobic layer, and thus CH_4 emissions are likely to be higher where water tables are closer to the surface (e.g. Hendriks et al., 2007; Wilson et al., 2009). Furthermore, the higher water table position results in higher temperatures within the zone of CH_4 production and so rates of CH_4 production may also increase following rewetting. Due to the higher radiation forcing of CH_4 compared to CO_2 over a 100 year time period, it is possible that rewetting may increase the GHG warming potential of the ecosystem if the reductions in CO_2 emissions are offset by an increase in CH_4 release (e.g. Beyer and Hoper, 2015).

A water table within 10 cm of the surface is considered optimum for mitigation against GHG emission (Couwenberg et al., 2011); achieving a balance between reduced CO_2 emissions and increased CH_4 emissions. However, the complex hydrological nature of peatlands may result in inundated conditions occurring following restoration. Although winter water tables at or above the surface are common in natural peatland systems, year round inundation is not recommended because it hinders the establishment of peat forming species and potentially results in high CH_4 emissions. Few studies exist on the effect of rewetting on the GHG balance of fens, with many rewetting studies focusing on upland blanket bogs, lowland cutover bogs or grassland systems. Shallow flooding following rewetting in fens has been reported as common, with subsequent colonization by emergent species such as *Phragmites australis* and floating tall sedge reeds (Minke et al., 2016). Minke et al. (2016) measured GHG fluxes across an inundated mesotrophic and eutrophic fen in Belarus and found the CO_2 balance to be near neutral or negative across the two sites. However, the high CH_4 emissions observed (3.8 to 37.7 t CO_2-eq ha^{-1} a^{-1}) generally resulted in an overall net warming effect (i.e. positive GHG balance), with CH_4 emissions significantly higher from the eutrophic fen.

1.6.2. Vegetation management

The high biodiversity of natural peatlands gives them an important role in nature conservation, however, the drainage of peatlands and subsequent abandonment encourages vegetation succession, with increased coverage of shrubs and woody species (Fen Management Handbook, 2011). Consequently restoration often seeks to encourage the establishment of

desirable vegetation communities such as peat forming species or fen plant communities through the removal of rank or overgrown vegetation. The manual cutting of biomass (e.g. with the use of strimmers) enables local vegetation management which preserves habitat continuity and microtopography, and thus is suitable for peatlands with high conservation value. Typically, the cut biomass is left in situ, resulting in an increased availability of fresh labile carbon and therefore an increase in R_{ECO}. In addition to biomass removal to promote the development of herbaceous communities, hay cutting also takes place in grasslands on peat soils under agricultural management. The removal of biomass results in a substantial decrease in leaf area and therefore photosynthesis, with R_{ECO} dominating the carbon balance immediately after cutting. Beetz et al. (2013) found it took an average of 22 days for GPP to exceed R_{ECO} following biomass removal; equating to 40 % of the growing season over the study period. In addition to a reduction in GPP, the carbon allocated to above ground biomass is removed from the system and therefore presents a significant offsite carbon loss.

Burning is a controversial form of fen management and often only implemented where it was traditionally used in the past (Fen Management Handbook, 2011). Historically, burning was carried out on fens in north Wales to ensure the fresh growth of vegetation in the spring for grazing animals, and more recently has been used as an effective method of removing overgrown and dead biomass (Fen Management Handbook, 2011), particularly woody species (Middleton et al., 2006). Burning is more commonly carried out on moorland (upland bogs) for grouse management, however, there is still little in the literature on carbon balance impacts (Worrall et al., 2009). Studies by Worrall et al. (2007) showed significant decreases in water table depth in burnt plots compared to unburnt, likely due to the removal of vegetation during burning. Burning does not completely remove surface vegetation and the nutrients from the remaining ash deposits may result in increased recovery rates (Garnett et al., 2000), however this is not necessarily great enough to offset the biomass lost in burning (Worrall et al., 2010).

1.6.3. Topsoil removal

The rewetting of peatlands where the top layer of peat has become severely nutrient enriched, e.g. from agricultural runoff or fertilisation, can lead to eutrophication and significant losses of phosphorous and ammonium (van de Reit et al., 2013; Zak et al., 2017). Thus in heavily degraded fens, top soil removal is increasingly used as a restoration technique to remove the

enriched and mineralized surface layer of peat (Klimkowska et al., 2010). This 'ecosystem reset' creates nutrient poor conditions analogous to pristine peatlands and removes any seed bank of invasive species (Emsens et al., 2015; Patzelt et al., 2001), reducing the likelihood of competitive fast growing species becoming established (Klimkowska et al., 2015). The spontaneous recolonisation of fen species occurs through wind-, animal- and water- dispersal from local refugia, however, the fragmentation of fens limits the effectiveness of these methods and propagules may be required for rare species (Malson et al., 2008). Covering bare soil with straw mulch or hay has also been used as an effective method for increasing soil moisture and encouraging the colonization fen species where seeds are present within the hay (Patzelt et al., 2001).

Though the effect of topsoil removal on the restoration of ecological functions of peatlands has been studied, little information exists on the response of GHG fluxes to this intervention. Furthermore, identifying the effect of topsoil removal on GHG fluxes in the field is a challenge as the process is often accompanied by rewetting which also alters the biogeochemical processes within the soil. A mesocosm experiment by Harpenslager et al. (2015) measured GHG fluxes from rewetted subsoil and topsoil cores from a former agricultural peatland and found CH_4 fluxes to be 99 % lower from the subsoil cores compared to the topsoil cores. This was attributed to the lower availability of labile organic matter and nutrients in the subsoil which resulted in limited substrate and P availability for methanogenic populations. In contrast, little difference was observed in CO_2 fluxes from the topsoil and subsoil cores, with all topsoils and most subsoils acting as net CO_2 sources and only a few subsoils showing net CO_2 uptake. Whilst topsoil removal appears to reduce C emissions and encourage the establishment of fen species, there is a significant carbon loss to the system in topsoil removed from the site. Harpenslager et al. (2015) suggest the removed topsoil should be relocated to agricultural peatlands still in use to increase the surface level, however no studies were found to have quantified the carbon emissions associated with this peat layer.

1.6.4. Cutover bogs

Peatlands drained and used for extraction require a combination of rewetting and revegetation in order to restore a functioning ecosystem. The spontaneous recolonisation of peatlands following peat extraction is a slow process due to the exposed conditions of bare peat surfaces (Poulin et al., 2005), though early colonisers such as *Eriophorum vaginatum* and *Calluna*

vulgaris are often present within a few years of abandonment (Trinder et al., 2008). Revegetation of cutover bogs may use similar hay spreading and seed transfer techniques as in peatlands following topsoil removal (Jarveoja et al., 2016), however, for Sphagnum colonisation rewetting is also crucial for maintaining water tables at the surface. Rewetting through ditch blocking in cutover bogs raises the water table, creating anaerobic conditions and therefore reducing CO_2 emissions. However, drained peatlands are susceptible to large water table fluctuations due to their lower water retention capacity, which may lead to high CO_2 emissions due to the increased decomposition of residual and more recent organic substrate (Wilson et al., 2016b), as well as creating unfavourable conditions for Sphagnum colonization (Quinty and Rochefort, 2003).

Whilst the restoration of cutover bogs has been taking place for decades, there is little long term monitoring of GHG fluxes to determine whether a net C sink has successfully been re-established. Furthermore, whilst studies compare adjacent restored and extraction sites (e.g. Vanselow-Algan et al., 2015), there is no evidence on GHG fluxes immediately after the cessation of peat extraction and rewetting. It would be expected that the topsoil would be devoid of microbial communities due to the lack of vegetation and continual extraction and therefore there would be a lag in the establishment of new populations following rewetting. Consequently, CH_4 fluxes might be low, despite the prevalence of anaerobic conditions (e.g. Wilson et al., 2016b). The longest reported GHG flux data set for a rewetted cutover bog is in Ireland where 5 years of measurements have taken place (Wilson et al., 2016b). On average over the five years, the rewetted microsites acted as net CO_2 sinks (-104 ± 80 g C m^{-2} a^{-1}), however, there was considerable interannual variability with occasional annual net CO_2 emissions observed in years when the water table was low (up to 29 cm below the surface). Annual CH_4 emissions were high (5.3 ± 3.0 to 11.2 ± 9.0 g C m^{-2} a^{-1}), leading to the majority of microsites having a positive GHG balance (i.e. a net warming effect), with the exception of the *Eriophorum* spp. dominated plot where a very high CO_2 uptake was found (Wilson et al., 2016b). Similar findings have been reported from rewetted cutover bogs in Germany, where high CH_4 emissions dominate the GHG balance (Vanselow-Algan et al., 2015), particularly where inundation occurs (Beyer and Hoper, 2015). Although some studies have reported smaller CH_4 emissions, these are generally where, although rewetting has taken place, mean annual water tables remain relatively low and consequently net CO_2 emissions are also observed (Strack and Zuback, 2013; Jarveoja et al., 2016). The rewetting of cutover bogs where water tables are maintained just below the surface appear to be successful in reducing CO_2

emissions, however, due to increased CH_4 emissions, considerable variability is found both interannually and between vegetation types.

1.7. Measuring GHG fluxes

Quantifying the peatland carbon balance requires a field-based approach, with two principle techniques used; eddy covariance flux tower measurements and gas chamber measurements. Eddy covariance towers take direct measurement of the rate of vertical transport of the gas species being studied (Denmead, 2008); usually CO_2 and CH_4. The instantaneous vertical flux density of a gas is measured at a frequency of ~10 Hz and averaged over a time period, usually 15 minutes to 1 hour, to include all the effective transporting eddy sizes. The area over which fluxes are measured, the flux footprint, is governed by wind direction and velocity and can extend hundreds of meters upwind of the flux tower, providing an integrated flux at the ecosystem scale (Baldocchi, 2003). Whilst flux tower measurements are generally non-intrusive following initial set up of the tower and can provide continuous ecosystem scale NEE measurements, they rely on assumptions such as flat terrain and horizontal homogeneity, which are not always met in peatland ecosystems (Laine et al., 2006; Schrier-Uijl et al., 2010).

Gas flux measurements using chambers are based on the principle that gas exchange within a restricted volume of air magnifies changes in the concentration of gas in the head space (Denmead, 2008). Closed chamber measurements are more commonly used than flow-through chambers, which are open to the atmosphere, as changes in gas concentration are easier to detect. Closed chambers are placed over a known area of soil with an airtight seal, following which a change in gas concentration is observed within the chamber due to the gas exchange processes between the soil, vegetation and chamber headspace (Fig. 1.4.). Fluxes are calculated based on the change in concentration with time, using Eq. 1.1 (Denmead, 2008);

$$F_{GHG} = \frac{\partial c}{\partial t} \frac{V}{A} \quad [1.1]$$

Where F_{GHG} is the flux of CO_2 or CH_4 in g GHG m^{-2} hr^{-1}, $\delta C/\delta t$ is the change in gas concentration over time (mol m^{-3} hr^{-1}), V the total volume of the chamber and the aboveground portion of the collar (m^3), A the area of the collar (m^2). The processes contributing to CO_2 exchange is governed by the type of chamber used; transparent chambers measure NEE (i.e.

GPP and R_{ECO}), whilst chambers covered in an opaque shroud only measure R_{ECO}. GPP fluxes can therefore be calculated as the difference between NEE and R_{ECO} fluxes.
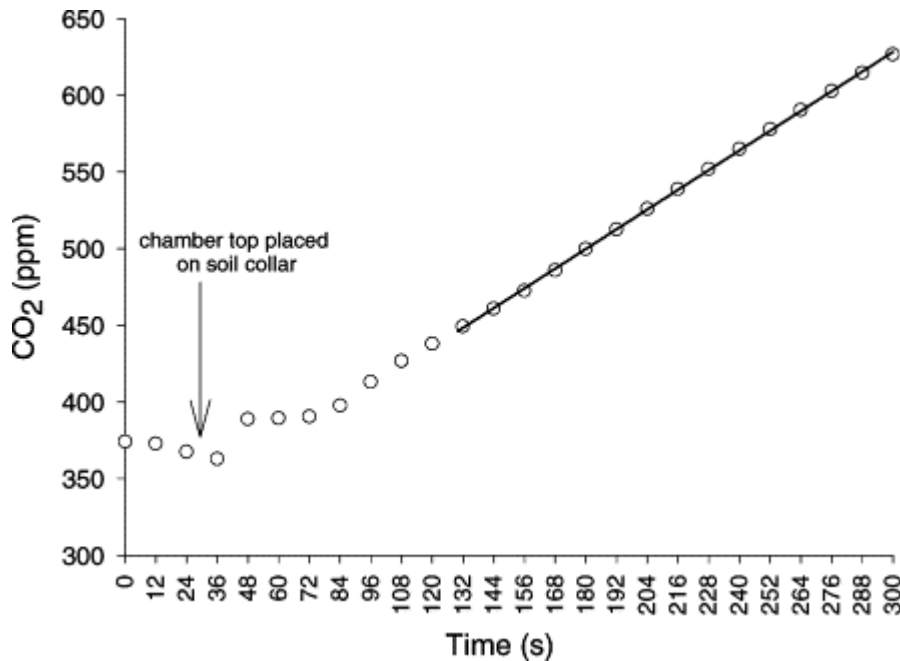
Figure 1.4. Example CO_2 gas flux measurement using a closed chamber (Davidson et al., 2002). Points represent measured CO_2 gas concentrations, the solid line represents the linear regression used to calculate the flux. The 'noise' in CO_2 concentration at the time of chamber placement is presumed to be caused by chamber placement disturbance.

Closed chambers can be either static, where there is no air circulation between the chamber and sensor, or dynamic where air is circulation from the chamber to analyser in a closed loop (Denmead, 2008), with the latter becoming more common as field portable GHG analysers become cost effective. Maintenance of a gas-tight seal between the chamber and collar or ground surface is crucial for accurate gas flux measurements using closed chambers, without which gas concentrations and therefore fluxes will be underestimated (Pihlatie et al., 2013). Further flux underestimations may occur as the accumulation of CO_2 or CH_4 within the chamber headspace reduces the concentration gradient within the soil profile, therefore reducing the rate of gas efflux (Davidson et al., 2002). This effect is considerably reduced using closed dynamic chambers where gas concentrations can be measured at high frequencies (every $1 - 2$ seconds) thus the most appropriate regression function can be applied, omitting initial disturbances created by chamber placement and feedback effects after long chamber closures (Fig. 1.4., Davidson et al., 2002). The presence of fieldworkers can also result in disturbance

which is evident in the gas concentration measurements, for example ebullition events caused by fieldworkers approaching collars can result in the systematic overestimation of CH_4 fluxes, particularly in locations where the water table is near the surface (Hargreaves and Fowler, 1998). The risk of user induced disturbance can be reduced through the use of boardwalks, reducing the impact on the peat.

Due to the small size of chambers (< 0.5 m^2) the small scale spatial heterogeneity of peatlands can be captured, enabling fluxes to be described specific to environmental conditions such as vegetation type or land use management, which cannot be achieved by eddy covariance systems. However, the high fieldworker requirements associated with chamber flux measurements severely restricts their frequency. The development of automatic chamber systems has sought to overcome this challenge, enabling high frequency gas flux measurements on small spatial scales. Comparisons of CO_2 flux measurements by manual and automated chambers did not reveal significant differences in a study by Burrows et al. (2005) at a temperate poor fen, however, manual chambers did record higher maximum estimates of gross photosynthesis. Ideally automated chamber measurements, as with eddy covariance systems, would allow continuous gas flux measurements, however logistical constraints and technical challenges mean this is rarely the case and modelling is still required for annual flux estimates. GPP modelling relies on measuring CO_2 fluxes at a range of light levels, a task which can be imitated using various levels of shading with manual chambers (e.g. Elsgaard et al., 2012; Gatis et al., 2016), however is easily captured using automatic chambers which can measure throughout the diurnal cycle. Night time measurements have, however been found to overestimate CO_2 fluxes where low atmospheric conditions have persisted (e.g. Schneider et al., 2009; Juszczak et al., 2012; Koskinen et al., 2014). Calm conditions result in CO_2 accumulating at the peat surface, reducing the concentration gradient between the soil and the atmosphere and therefore decreasing the rate of the diffusive flux (Schneider et al., 2009). As the automated chambers close, this stratified surface layer is disturbed, and the concentration gradient and therefore CO_2 efflux is increased. This effect can be further enhanced by the presence of chamber fans mixing the headspace (Lai et al., 2012; Görres et al., 2016). The use of nocturnal chamber measurements in R_{ECO} modelling, therefore requires caution, with fluxes omitted where turbulent conditions are not present (Juszczak et al., 2012).

1.8. Modelling of GHG emissions

Chamber flux measurements provide a 'snapshot' of the GHG exchange at a particular point in space and time. To estimate GHG fluxes between measurement dates, it is necessary to model fluxes based on relationships with environmental factors such as photosynthetically active radiation (PAR), soil or air temperature, water table and vegetation cover which vary both diurnally and seasonally. Modelling of CO_2 and CH_4 fluxes enables annual budgets to be compared between sites and years, as well as contributing towards land use emission factors when reporting to the United Nations Framework Convention on Climate Change.

GPP is usually modelled using a light response parameter which produces a rectangular hyperbola, similar to the commonly used Michaelis-Menten curve which describes enzymes kinetics (Fig. 1.5). The initial steep part of the curve indicates a rapid increase in CO_2 uptake as irradiance increases from zero, followed by a peak of the asymptote with further irradiance increase leading to a diminishing increase in CO_2 uptake. The light response curve varies according to environmental parameters including temperature and vegetation (type and age) and therefore is most successfully fitted over short time periods (days to weeks) using flux measurements from a range of light conditions for specific vegetation types (Hoffmann et al., 2015). Alternatively, seasonal or phenological variables may be included in the model to account for the changing relationship between GPP and irradiance throughout the year. Tuitilla et al. (1999) constructed an effective temperature sum index (ETI) to represent the ratio of the sum of cumulative temperature to the number of temperature sum days, whilst Burrows et al. (2005) included a ratio vegetation index to represent the quantity of green leaves present.
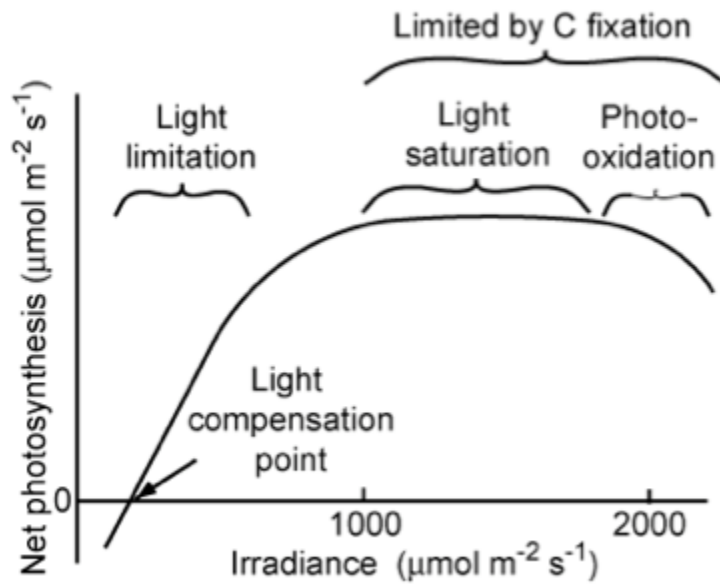
Figure 1.5. GPP light response curve: The light limitation area represents the linear increase in photosynthesis with increasing irradiance, where light use efficiency is constant. The light compensation point is the point where a net uptake of CO_2 begins (Chapin et al., 2011).

R_{ECO} modelling is typically based on the relationship between CO_2 emissions and air or soil temperature, though this again is often modified using additional explanatory variables such as water table (Bellisario et al., 1999; Samaratini et al., 2011), ETI (Tuittila et al., 1999) or leaf area index (Shaver et al., 2013). The Lloyd and Taylor model has been widely applied to R_{ECO} modelling in peatlands (Lloyd and Taylor, 1994; Beetz et al., 2013; Vennendaal et al., 2007; Eickenscheidt et al., 2015) using soil temperature from a range of depths (2, 5 and 10 cm) as well as air temperature. Uncertainty still surrounds the effect of high temperatures on R_{ECO} fluxes, when other parameters may limit CO_2 production and efflux such as soil moisture availability and microbial kinetics. Field studies have found logistic models to improve the modelling of R_{ECO}, enabling the plateauing of fluxes at temperatures above $\sim$ 20 °C to be captured (Lee et al., 2017) and reducing overestimation of fluxes at high temperatures. The relatively slow changes in temperature and associated R_{ECO} response means models tend to be more successful when fitted over a longer time periods covering either seasonal or annual changes, rather than just the diurnal cycle.

The fine scale controls on CH_4 fluxes are still not fully understood, with CH_4 production, consumption and transportation contributing to the fluxes measured within chambers. Many studies linearly interpolate CH_4 fluxes between measurement campaigns, though this may lead to the overestimation of annual fluxes, especially where summer fluxes are high (Schrier-Uijl et al., 2010). Additionally, interpolation risks missing pulses of CH_4 emissions such as those associated with rainfall or land management events. Studies which model annual CH_4 fluxes typically use simple non-linear regression with temperature as the sole explanatory variable (Schrier-Uijl et al., 2010), however model uncertainty remains high, and may result in no statistically difference annual flux estimate to that derived through interpolation (Hendriks et al., 2007).

1.9. National GHG reporting and emission factors

The UK has committed to reducing GHG emissions under the Kyoto Protocol and the UK Climate Change Act (2008) and is obliged to report GHG emissions on a national scale to the United Nations Framework Convention on Climate Change (UNFCCC). The methods for calculating emissions is provided by the Inter-governmental Panel on Climate Change (IPCC), with emissions calculated for specific land use categories from an emission factor (EF) multiplied by activity data. In the case of peatlands, the EF represents the annual flux in t CO_2-C ha^{-1} a^{-1} (or kg CH_4-C ha^{-1} a^{-1}) and activity is the national spatial extent of the land use category. The IPCC reporting method comprises of three tiers; Tier 1 values are global defaults based on measurements from the different climate zones (boreal, temperate, tropics); Tier 2 values use more country specific data, whilst Tier 3 uses simple empirical models based on additional data. Tier 1 EFs were published by the IPCC for peatlands and other wetlands in the 2013 supplement to the 2006 GHG inventory guidelines (IPCC, 2014), enabling GHG emissions and removals from drained and rewetted peatlands to be included in national reporting. The widespread application of Tier 1 values, as well as limited data availability, resulted in EFs based on broad climate zones, peat types ('nutrient rich' fen peat and 'nutrient poor' bog peat) and land-use categories, and therefore are not fully representative of peatlands within the UK. For example, blanket bogs were not considered as a distinct peat type despite their different ecological functioning compared to other peatland types and yet are the most extensive peatland type within the UK. Furthermore, the 'temperate' climate zone included studies carried out in dryer continental regions, thus are unsuitable for comparison with the maritime conditions of the UK. The development of Tier 2 emissions from UK peatlands is

currently under review, and improves upon Tier 1 EFs by including UK specific land cover categories (e.g. specifying peatlands which have been modified but retain semi-natural vegetation), and using data sources solely from climatically similar regions (humid temperate).

1.10. Conclusion

Natural peatlands act as long term carbon stores and net sinks of atmospheric CO_2, however their drainage for agriculture, forestry and extraction has led to net losses of stored carbon. The recognition of peatlands as potential carbon sinks and vulnerable carbon stores, as well as their biodiversity value, has led to increased restoration efforts through rewetting, vegetation management and topsoil removal. The magnitude of carbon fluxes from peatlands depends upon a range of environmental variables including temperature, WTD, vegetation and land use management, though current studies on GHG fluxes from UK lowland peatlands are sparse. Whilst significant progress was made with the recent Defra Lowland Peat Project, the effects of restoration interventions on the carbon balance of lowland peatlands is understudied. Further research is crucial to determine the immediate response and long term changes in peatland GHG fluxes with restoration, especially to assess whether the carbon sink function of peatlands can be re-established, together with other ecosystem services such as flood mitigation and biodiversity.

1.11. References

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Chapter 2

Heterogeneous greenhouse gas flux response to restoration in a temperate lowland fen

Abstract. Peatlands can act as a net source or sink of atmospheric carbon (C) depending on a range of environmental conditions including water table level, temperature, vegetation cover and land use. The drainage of peatlands for agriculture, plantation forestry and extraction has led to substantial net losses of stored C across the UK, other areas of northern Europe and more recently in tropical peatlands. Recognition of the importance of peatlands as both vulnerable C stores and potential C sinks has led to increased efforts to restore degraded peatlands, especially in Europe. However, rewetting of peatlands may cause large increases in emissions of methane, a potent greenhouse gas (GHG), which may negate or reverse any benefit attained by increasing CO_2 uptake. This study estimates CO_2 and CH_4 fluxes across a partially degraded lowland fen subjected to different restoration measures; a tall fen and short fen underwent vegetation removal through burning and hand-cutting whilst enriched topsoil was removed at a former improved pasture. Annual CO_2 and CH_4 fluxes were modelled for control and intervention plots based on monthly manual chamber measurements made over a one year period. The control and intervention plots in the tall fen acted as net C sinks, with vegetation productivity and water table driving differences between plots. The short fen control plot acted as a net C sink, whereas the intervention plots acted as net C sources with ecosystem respiration rates driving differences in CO_2 exchange, and water table and vegetation cover driving differences in CH_4 emissions. Following topsoil removal at the former improved pasture, the bare soil acted as a net C source; however, a small net C sink was observed where natural recolonisation had occurred within two years of topsoil removal. The opposing response in GHG fluxes from the tall and short fen following intervention has implications for the management of degraded fens and highlights the importance of taking into consideration initial conditions in restoration decision making.

2.1. Introduction

Globally, organic soils are estimated to store 600 Gt of carbon (C); twice that held by the entire forest biomass worldwide, yet only cover 3 % of the Earth's surface (Wilson et al., 2016a; Joosten et al., 2016). In their natural state, peatlands provide a wealth of ecosystem services including water regulation, habitat provision and C sequestration and storage, with their value recognised under international agreements such as the Kyoto Protocol and the Ramsar Convention on Wetlands (Bonn et al., 2014). The very presence of peatlands demonstrates their function as a long term net sink of atmospheric C, with pristine peatlands currently estimated to sequester ~ 100 Mt C yr^{-1} (Joosten et al., 2016). Drainage of peatlands (e.g. for agriculture) increases soil aeration which accelerates the decomposition of organic matter, converting drained systems into net sources of atmospheric CO_2. In contrast, undrained peatlands are associated with higher CH_4 emissions due to methanogenesis within the anaerobic layer of the soil, and restricted methanotrophy within the thin overlying aerobic layer. Natural wetlands are estimated to account for 30 % of global CH_4 emissions (Stocker et al., 2013), with default emission factors for rewetted organic soils ranging from 1.86 to 9.79 t CO_2-eq ha^{-1} yr^{-1} (Wilson et al., 2016a).

European peatlands account for approximately 12 % of the global peatland area, yet have undergone the largest loss of mire extent (Joosten and Clarke, 2002). Within the UK, peatlands cover ~ 23 000 km^2, however, peaty soils are much more extensive (~ 80 000 km^2) indicating past peatland habitats were more widespread (Evans et al., 2011). UK peatlands are broadly categorised into bogs and fens, with the former receiving inputs solely through precipitation, whilst fens receive additional inputs through groundwater. Whilst upland blanket bogs prevail within the UK, lowland peatlands have been subjected to more intense land use pressure due to their potential for agricultural production and peat extraction, and thus likely contribute disproportionately towards greenhouse gas (GHG) emissions despite their relatively small extent (Evans et al., 2017). The majority of UK GHG studies are based on upland blanket bogs, with steady-state lowland peatlands, including fens, only recently being addressed (Evans et al., 2017).

The modification of peatlands primarily starts with drainage to lower the water table, enabling agricultural or industrial activities such as peat extraction to begin. Drainage increases O_2 availability in the soil leading to increased microbial and root respiration and therefore an increase in CO_2 produced and released to the atmosphere (Maljanen et al., 2007; Renou-Wilson

et al., 2014). Concurrently, the zone of methanogenesis moves down the soil profile, resulting in a greater opportunity for CH_4 oxidation in the newly exposed overlying aerobic layer, thus a reduction in CH_4 released to the atmosphere is observed (Wilson et al., 2009). In addition to significantly changing terrestrial C fluxes, peatland drainage ditches have been found to be hotspots of CH_4 emissions in productive high nutrient sites such as agricultural grasslands, where water flow is low and labile C inputs are high (Peacock et al., 2017). Although water table level is considered the most important control on peatland GHG fluxes, agricultural practices such as tillage and fertilisation as well as intensive management of agricultural grasslands through grazing or biomass cutting, have been found to increase soil respiration rates and therefore CO_2 emissions to the atmosphere (Beetz et al., 2013; Vennendaal et al., 2007). Unlike peatlands drained for agriculture, those undergoing peat extraction have significantly smaller C turnover rates; however, the absence of vegetation means that respiration is the sole component of NEE and thus cutover peatlands act as net C sources (Waddington et al., 2002).

Restoration of drained peatlands is increasingly common, particularly where sites have been abandoned and still retain some semi-natural vegetation, or where extraction has ceased and land been acquired by conservation bodies. Peatland restoration broadly aims to re-establish peat forming vegetation, and thus promote C storage as well as achieving favourable nature conservation status (Lunt et al., 2010). However, the nature and level of intervention needed to achieve this outcome differs depending on the degree to which the hydrology, peat characteristics and vegetation of the site have been modified. Methods used to restore peatlands include water management (e.g. blocking drainage ditches), revegetation (of bare soil) and vegetation management (e.g. biomass removal by cutting or burning) to encourage the growth of peat forming species (Lunt et al., 2010). In cases of extreme ecosystem modification such as enrichment by agricultural nutrients, topsoil removal has been used as a method to recreate more oligotrophic conditions and allow wetland species to re-establish (Emsens et al., 2015; Leonard et al., 2016). Different restoration management techniques are often used in combination, and in some cases it may be necessary to maintain lower-intensity management such as grazing or regular mowing beyond the original restoration period.

There is little evidence on the effects of these restoration measures on the C balance in fens, with studies focusing on the impact on plant communities (e.g. Kotowski et al., 2013; Menichino et al., 2016) rather than GHG fluxes. Vegetation management typically seeks to

remove rank vegetation or prevent succession of woody species in favour of fen plant communities. The cutting of vegetation which is then left in situ is typically accompanied by an increase in ecosystem respiration due to the increased availability of fresh labile C inputs (Limpens et al., 2008). Furthermore, the cutting of biomass substantially reduces or eliminates productivity, thus immediately after cutting, a temporary net emission of CO_2 is likely. If biomass is removed offsite, there is an additional C loss to the system and off-site CO_2 emissions may arise from the burning of biomass (Wilson et al., 2016a). Although at a much slower rate, grazing also acts to remove biomass from peatlands, reducing leaf area, productivity and therefore C sequestration. Livestock may also graze selectively (e.g. cows avoid *Juncus* spp. if other food sources are available) resulting in a shift in species composition which in turn will affect GHG dynamics. For example, Renou-Wilson et al. (2016) observed high CH_4 emissions following the cessation of grazing due to the dramatic growth and colonisation of *Juncus* spp. in a shallow drained grassland on peat in Ireland, although there was no direct effect of grazing on net CH_4 fluxes, with vegetation growth corresponding with higher than average air temperature and photosynthetic photon flux density (PPFD). After grazing ceased, both gross primary productivity (GPP) and ecosystem respiration (R_{ECO}) increased, resulting in little change in net ecosystem exchange (NEE) under shallow-drained conditions, whilst rewetted areas of the grassland saw a large increase in GPP relative to R_{ECO}, resulting in an increased C sink (Renou-Wilson et al., 2016).

The rewetting of peatlands during restoration is critical for re-establishing peat forming vegetation, which is highly susceptible to desiccation (McNeil and Waddington, 2003). Raising the water table creates anaerobic soil conditions, inhibiting ecosystem respiration and therefore reducing CO_2 release to the atmosphere. A decrease in CO_2 emissions has been observed from rewetted peatlands (Beetz et al., 2013; Renou-Wilson et al., 2016); however, a water table close to the surface can promote high CH_4 emissions (Beetz et al., 2013; Hendriks et al., 2007; Wilson et al., 2009) due to the suppression of methanotrophic processes. Consequently, rewetting may not necessarily result in successful mitigation of GHG emissions if reductions in CO_2 emissions are offset by an increase in CH_4 release, particularly given the higher climate forcing of CH_4 (Beyer and Hoper, 2015); it is therefore possible that rewetting may increase the GHG warming potential of the landscape, contributing more to global warming and climate change, despite drawing down more C annually.

In cases where the required peat hydro-geochemical conditions can only be obtained through an 'ecosystem reset' involving the removal of highly trophic topsoil (Emsens et al., 2015; Klimkowska et al., 2015), conditions are created analogous to cutover bogs, with peat forming vegetation being established on bare peat. GHG fluxes from bare soil in cutover bogs are typically characterised by moderate CO_2 emissions and negligible CH_4 fluxes (Waddington et al., 2002; Waddington et al., 2010; Vanselow-Algan et al., 2015), suggesting that a temporary net emission of C will occur following topsoil removal. Indeed, Harpenslager et al. (2015) showed that removal of topsoil reduced CH_4 emissions by 99% in a mesocosm experiment that examined the impact of restoration of peat soils previously under agricultural management. Following topsoil removal, vegetation colonisation is encouraged through rewetting and mulching (Price et al., 1998), with studies showing former cutover or agricultural peatlands under restoration management may become either C sources or sinks (e.g. C. Beyer & Höper, 2015; Vanselow-Algan et al., 2015).

At least 20,000 ha of UK peatland are currently under restoration management, with ecological and hydrological restoration the two principal drivers identified during the 2009 Peat Compendium (Holden et al., 2008). Although the monitoring of GHG dynamics from UK lowland peatlands under restoration management is limited, emission reductions provided by peatlands can be included in national GHG reporting to the UN Framework Convention in Climate Change, thus it is critical to improve our understanding of the effects of restoration activities on GHG fluxes. The aim of this study is therefore to investigate the effects of three restoration interventions: (i) vegetation removal by hand-cutting; (ii) vegetation removal by burning; and (iii) topsoil removal; on CO_2 and CH_4 fluxes across a partially degraded lowland fen which encompasses tall fen (ecologically degraded) and short fen (ecologically good condition) vegetation and an area of restored pasture. The hypotheses of this study are that (1) plots with vegetation removed by hand-cutting and burning will have larger CO_2 emissions but smaller CH_4 emissions compared to unmanaged control areas; (2) following topsoil removal bare peat plots will act as a net CO_2 source with negligible CH_4 emissions; (3) revegetated plots which have undergone topsoil removal will be neutral with regards to CO_2 and CH_4 fluxes compared to the unmanaged control plot which will be a net source.

2.2. Methods

2.2.1. Experimental design

The study was located at Cors Erddreiniog (53°18' N, 4°17' W), a 289 ha calcareous valley head fen complex located within the Anglesey fens Special Area of Conservation, North Wales, UK. The site has a temperate maritime climate with an average annual air temperature of 10.4 °C (1981 – 2010) and average annual precipitation of 841 mm (1981 – 2010). The Anglesey Fens were extensively drained during the 19th century (Meade and Blackstock, 1988) and have been historically managed through grazing, burning and mowing for domestic peat cutting and harvesting for thatching. To ascertain the impact of restoration activities on GHG fluxes, three sites were selected within Cors Erddreiniog: (i) a tall fen (TF) considered botanically poor and dominated by sedges and grasses (*Phragmites australis*, *Cladium mariscus*, *Molinia caerulea)*; (ii) a short fen (SF) with botanically rich vegetation (brown moss hollows, *Sphagnum* hummocks and *Juncus* species); and (iii) an area of former improved pasture which has now been restored to fen (hereafter referred to as restored pasture; RP). Spatial heterogeneity in nutrient conditions are found across Cors Erddreiniog due to past management as well as the presence of a ridge ~70 m wide separating the SF and TF. The limestone hillslope to the east of the fen results in base-rich groundwater feeding directly into the SF, which together with spring upwellings, supports fen meadow vegetation. In contrast, a number of perimeter drains surround the TF resulting in the site being hydrologically isolated from lateral base-rich inputs, thus it is slightly more acidic and botanically degraded compared to the SF (Table 2.1).

Cors Erddreiniog was included in a large scale European Union-funded restoration project aiming to restore the Anglesey Fens (LIFE07 NAT/UK/000948), with two different restoration interventions conducted at the TF and SF sites in 2012: (i) vegetation removal by burning, and (ii) vegetation removal by hand-cutting. This resulted in two intervention plots within the TF and SF, with areas of unmanaged vegetation acting as controls (Fig. 2.1). At the RP site, enriched topsoil was removed in winter 2011/2012 over an area of approximately 5 ha, which together with re-profiling, re-established oligotrophic peat with marl layers at the surface as well as high water tables and diffuse soligenous inputs across the site. Hay spreading with *Schoenus nigricans* seed heads was used to promote the development of alkaline fen vegetation at part of the RP, with some areas left to naturally recolonise. Here, bare soil and recolonised vegetation plots were identified, and compared to an area where topsoil was not removed as a control (Fig. 2.1).
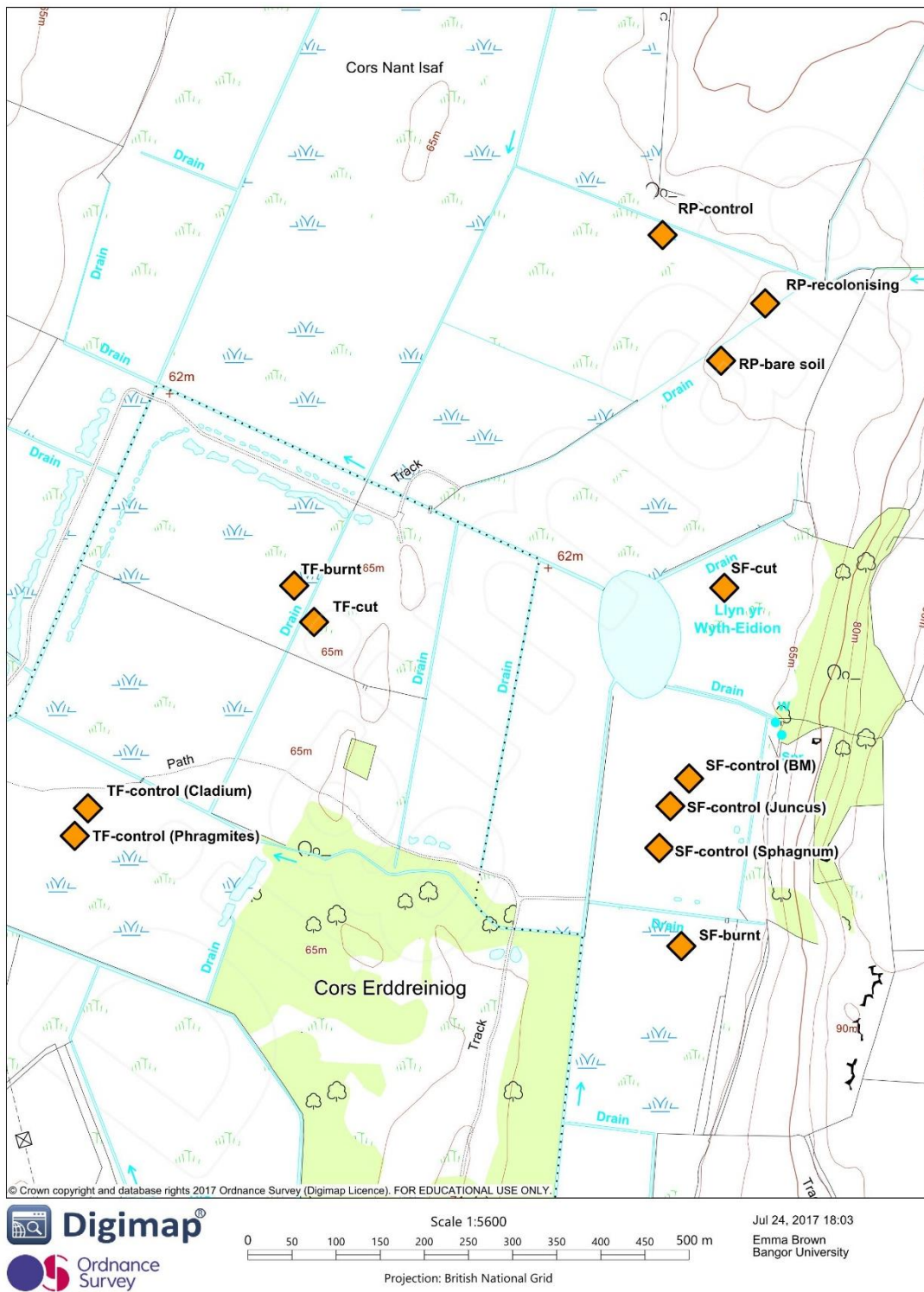
Figure 2.1. Site map of Cors Erddreiniog, with collar locations identified within the Tall Fen (TF); Short Fen (SF) and Restored Pasture (RP).

Table 2.1. Site description: dominant vegetation, number of sampling collars, mean soil properties and mean annual water table depth with ± 1 standard error of the mean shown in brackets. A negative water table denotes below the surface.

Measurement plot	Dominant vegetation (no. collars)	pH	Electrical conductivity ($\mu S\ cm^{-1}$)	Organic matter (%)	Mean water table depth (mm)
Tall Fen (ecologically degraded) (TF)					
Control	*Cladium mariscus* (3)	5.94 (0.1)	81 (7.5)	69.9 (3.2)	-71 (0.7)
	Phragmites australis (3)	5.73 (0.1)	83 (12)	70.6 (1.2)	-51 (0.8)
Burnt	*Phragmites australis* (4)	5.65 (0.02)	92 (27)	84.6 (9.9)	+69 (0.7)
Hand cut	*Molinia caerulea* (4)	5.92 (0.1)	118 (36)	83.0 (2.1)	-32 (0.7)
Short Fen (ecologically good condition) (SF)					
Control	*Sphagnum* hummock (3)	6.67 (0.04)	73 (7.1)	89.7 (0.7)	-95 (1.1)
	Brown moss hollow (3)	6.77 (0.1)	142 (16)	84.1 (1.0)	+12 (0.2)
	Juncus species (3)	6.75(0.1)	142 (12)	90.5 (3.7)	-43 (0.6)
Burnt	*Phragmites australis* (4)	6.76 (0.1)	218 (44)	75.6 (1.1)	-100 (1.3)
Hand cut	*Molinia caerulea* (4)	6.70 (0.1)	126 (2.7)	74.0 (1.3)	-67 (0.8)
Restored Pasture (RP)					
Control	Wet Grassland/rush pasture (3)	7.45 (0.2)	148 (15)	35.6 (1.8)	-114 (0.6)
Recolonising	Short sedges e.g. Black-bog rush (3)	7.36 (0.1)	184 (34)	64.5 (15)	+1.8 (0.1)
Bare soil	Bare soil (3)	7.30 (0.1)	318 (77)	31.5 (1.3)	-109 (1.1)

2.2.2. Laboratory methods

2.2.2.1. Soil moisture and organic matter content

Triplicate soil samples were collected from the dominant vegetation within each measurement plot in March 2015 using a 15 cm length soil core. Soils were transported back to the laboratory at the Centre for Ecology and Hydrology in re-sealable plastic bags and stored at 4 °C overnight then homogenised before analysis. The loss on ignition method develop by Ball (1964) was used to determine soil moisture content and soil organic matter (SOM) content of the samples. A sub-sample of field-moist soil was weighed using an analytical balance (Reflex HP 220C Avery Weigh Tronix, Smethwick, England) and then heated at 105 °C for 16 hours. Samples were then cooled in a desiccator and re-weighed to determine soil moisture content. The dry samples were then placed inside a chamber furnace (Carbolite CWF1100, Carbolite Ltd, Hope Valley, UK) at 450 °C for a further 16 hours, then re-weighed after cooling to determine the SOM content.

2.2.2.2. Electrical conductivity and pH

A 10 g soil subsample was mixed with 25 ml deionised water and left for 30 mins. The electrical conductivity (EC) was measured using a 4320 Jenway electrical conductivity meter (Bibby Scientific Ltd, UK) and pH with a pH SevenEasy pH probe (Mettler-Toledo AG Analytical, Switzerland), which was calibrated beforehand using a pH 4 and 7 buffer.

2.2.3. GHG measurements

CO_2 and CH_4 fluxes were measured every four weeks from March 2014 until March 2015 between 10:00 and 16:00, using closed chambers with an Ultraportable Greenhouse Gas Analyser (Los Gatos Research Inc., USA, see Table A1 for equipment precision and calibration). A minimum of three replicate collars were installed within each treatment, and where vegetation heterogeneity necessitated it, replicates were installed in each major vegetation type within a treatment (Table 2.1). Collars (0.6 x 0.6 m) were installed in each plot 0.15 m deep into the soil and approximately 3 m apart at least three weeks prior to the start of measurements. A modular chamber system (0.6 x 0.6 x 0.3 m) was used that could be increased in height using additional (0.6 x 0.6 x 0.5 m) sections to allow measurement of tall vegetation and equipped with a pressure equalisation balloon and small fans to ensure sufficient mixing of air. Transparent Perspex chambers were used to measure net ecosystem exchange (NEE) followed by a chamber covered in an opaque shroud for ecosystem respiration (R_{ECO}), with each chamber placement lasting up to 5 min. During gas exchange measurements, air pressure

(Extech Instruments Ltd., USA) and soil temperature were measured at the start of each chamber placement, and chamber headspace temperature and humidity were logged every 10 s (Hobo External Data loggers; Onset Computer Corp., USA). Boardwalks were used within plots to minimise damage to vegetation and reduce soil disturbance or compression during GHG measurements. Throughout the measurement campaign, WTD was measured at each sampling location using dipwells installed next to chamber plots. Soil temperature at 10 cm and 30 cm depth and air temperature at 1.2 m were monitored using an on-site automatic weather station (AWS) by Reference Thermistor sensors (Probe T107, Campbell Scientific, Logan, UT, USA). Irradiance was measured at 1.2 m by the AWS using a CM3 pyranometer (Kipp & Zonen, Campbell Scientific, USA) from which photosynthetically active radiation (PAR) was calculated using a conversion value of 0.42.

2.2.4. Flux calculations

CO_2 and CH_4 flux rates were calculated using the linear change in chamber headspace gas concentration over time (Denmead, 2008) as shown in Eq. 2.1:

$$F_{GHG} = \frac{\partial c}{\partial t} \cdot \frac{V}{A} \quad [2.1]$$

where F_{GHG} is the flux of CO_2 or CH_4 in g GHG m^{-2} hr^{-1}, $\delta c/\delta t$ is the change in gas concentration over time (mol m^{-3} hr^{-1}), V the total volume of the chamber and the aboveground portion of the collar (m^3), A the area of the collar (m^2). Fluxes were accepted if the r^2 of the linear regression between gas concentration and time was greater than 0.7 and p value was ≤ 0.05. Fluxes which did not meet these criteria were visually inspected and retained if small but non-zero (i.e. $p \leq$ 0.05) as the high frequency measurements by the analyser allowed detection of low fluxes despite high short-term scatter (i.e. low r^2), avoiding a bias towards large fluxes. Negative fluxes denote the uptake of gas by the ecosystem and positive fluxes a loss to the atmosphere. Gross Primary Productivity (GPP) was calculated as the difference between NEE and R_{ECO}.

2.2.5. Carbon dioxide flux modelling

2.2.5.1. Ecosystem respiration

Annual flux estimates of R_{ECO} were modelled using the Arrhenius equation approach of Lloyd and Taylor (1994) (Eq. 2.2) parameterised with the soil temperature at a depth of 10 cm.

$$R_{ECO} = R_{ref} \exp\left\{E_0\left(\frac{1}{T_{ref}-T_0} \frac{1}{T_{soil}-T_0}\right)\right\}, \quad [2.2]$$

where R_{ECO} is measured ecosystem respiration (g CO_2 m^{-2} hr^{-1}), R_{ref} is the respiration rate (g CO_2 m^{-2} hr^{-1}) at the reference temperature T_{ref} (283.15 K), T_0 is the temperature constant (227.13 K) and T_{soil} is the temperature of the soil at a depth of 10 cm in Kelvin. E_0 is an activation-like parameter which is estimated along with R_{ref} using non-linear least squares regression. R_{ECO} was modelled for each vegetation group after measurements from the entire year were pooled.

2.2.5.2. Gross Primary Productivity

The relationship between PAR and GPP is often modelled using a rectangular hyperbolic saturation curve based on Michaelis-Menten kinetics (Burrows et al., 2005; Elsgaard et al., 2012; Beetz et al., 2013). Here, a modified version of this equation was used to model GPP (Eq. 2.3), which includes soil temperature measured at a depth of 10 cm as an additional explanatory variable.

$$\text{GPP} = \frac{GP_{max} \cdot \alpha \cdot \text{PAR}}{GP_{max} + \alpha \cdot \text{PAR}} * T_{soil} \quad [2.3]$$

where GPP is the measured gross primary productivity (g CO_2 m^{-2} hr^{-1}), PAR the photosynthetically active radiation (W m^{-2}) and T_{soil} the soil temperature at a depth of 10 cm (°C). The parameters α and $GPmax$ were estimated for each vegetation group using non-linear least squares regression, where α is the initial slope of the regression curve (g CO_2 m^{-2} hr^{-1} / W m^{-2}) and $GPmax$ is the limit of GPP when approaching infinite PAR (g CO_2 m^{-2} hr^{-1}).

2.2.6. CH$_4$ flux modelling

Annual CH$_4$ fluxes were modelled using multiple linear regression parameterised with soil temperature at 10 cm depth, PAR and WTD (Eq. 2.4).

$$\text{CH}_4 = T_{soil} \cdot \text{PAR} \cdot \text{WTD} \quad [2.4]$$

where $Tsoil$ is the soil temperature at a depth of 10 cm (°C), PAR the photosynthetically active radiation (W m^{-2}), and WTD the water table depth in mm. If a satisfactory model fit (according to the criteria of Hoffmann et al., 2015) could not be achieved using Eq. 2.4, annual CH$_4$ fluxes were calculated by linear interpolation between measurement campaigns (e.g. Beetz et al., 2013).

2.2.7. Annual flux estimates

Hourly estimates of R_{ECO}, GPP and CH_4 fluxes were reconstructed using continuous measurements of soil temperature at depth of 10 cm, PAR and WTD for each plot using Eq. 2.2, 2.3 and 2.4 respectively. Annual GPP, R_{ECO} and CH_4 flux estimates were calculated by summing hourly values from 1 March 2014 to 28 Feb 2015, along with annual NEE (NEE = GPP + R_{ECO}). Where more than one vegetation type existed, annual flux estimates for each plot was weighted by the area of each vegetation type; 75% *Phragmites* 25% *Cladium* for the TF control plot and an equal distribution of the three vegetation types at the SF control plot.

2.2.8. Net GHG balance (in CO₂-equivalents)

Due to the small area of measurement plots and complex spatial heterogeneity across the study area, it was not possible to quantify waterborne C exports which corresponded solely to the areas where terrestrial fluxes were measured. Therefore dissolved organic C, particulate organic C and dissolved inorganic C fluxes were not considered in the net GHG balance. Furthermore, N_2O fluxes were not measured as they are assumed to be negligible from semi-natural fens (Hendriks et al., 2007). To account for the stronger radiative forcing of CH_4, annual CH_4 fluxes were converted into CO_2-equivalents according to Forster et al. (2007) using the 100 year global warming potential (GWP) for methane of 25, in line with the approach used for UK Greenhouse Gas Inventory (Brown et al., 2016). CH_4 fluxes (t CO_2-eq ha^{-1} yr^{-1}) were added to the net CO_2 balance to calculate the annual net GHG flux expressed in CO_2-equivalents for each plot. Whilst considering only terrestrial C fluxes does not represent the full GHG balance, this terminology is used to refer to the net gaseous C exchange expressed in CO_2-equivalents for simplicity.

2.2.9. Statistical procedures

Transformation of fluxes did not enable the assumptions of parametric analyses to be met, therefore differences between measured CO_2 and CH_4 fluxes from measurement plots and vegetation groups were identified using the non parametric Friedman test. Correlations between fluxes and environmental variables were determined using Spearman's rank statistics. All data processing and analysis was carried out using the statistical programme R (R Core Team, 2015), with annual flux models fitted using packages 'nlme' (Pinheiro et al., 2016) and, 'zoo' (Zeileis and Grothendieck, 2005). Model performance was rated using the thresholds outlined by Hoffmann et al. (2015) including mean absolute error, RMSE-observations standard deviation ratio, co-efficient of determination, modified index of agreement, percent BIAS and Nash-Sutcliffe efficiency (Table A2 in Appendix A). Based on these measures, the

agreement between modelled and measured fluxes were classified from unsatisfactory to excellent. GPP and R_{ECO} model uncertainty was estimated using the approach outlined by Renou-Wilson et al. (2016), with the model's standard error expressed as a percentage of the mean fluxes which is then applied to the annual balance (Eq. 2.5).

$$E_r = \sqrt{\sum_{i=1}^{n} \frac{(F_{obs} - F_{mod})^2}{(n-1)*n}} \quad [2.5]$$

where Er is the model standard error, F_{obs} is the measured flux, F_{mod} is the modelled flux and n the total number of measured fluxes. The law of error propagation was used to calculate the uncertainty of the annual NEE flux estimate (the square root of the sum of squared standard errors of GPP and R_{ECO}) as it was not directly modelled.

2.3. Results

2.3.1. Site conditions

The SOM content of the tall and short fens ranged from 70 % to 91 % (Table 2.1) whereas the SOM content for the restored pasture plots was notably lower, consistent with the presence of marl across the site. The SF and TF control plots had consistently higher SOM contents than their respective intervention plots, however, differences were not statistically significant. Throughout the study period, the water level across the sites corresponded to the seasonal pattern of precipitation with the highest water tables in winter and lowest in summer. The mean annual water table position was within 10 cm of the surface in the majority of the plots (Table 2.1), with the brown moss hollow (SF-control), the RP-recolonising and TF-burnt plots being inundated for the majority of the year. A sustained period of water table drawdown during the summer was observed in most plots, with water levels occasionally more than 30 cm below the surface at the SF-burnt, SF-control Sphagnum and RP-bare soil plots.

2.3.2. Model performance

Model performance was rated as at least "satisfactory" using the criteria of Hoffmann et al. (2015) for all plots, with the exception of R_{ECO} at the RP-bare soil plot, where fluxes were consistently low. No other model tested could provide an improved fit and consequently the Lloyd and Taylor model was used. The modelled R_{ECO} fluxes generally matched measured fluxes well (Fig. 2.2); however, measured values tended to fall faster than the modelled values in autumn, indicating that model fits could be improved further. Although the inclusion of water table improved R_{ECO} model fits for some plots (RP-control, RP-bare soil, RP-

recolonising and SF-burnt), a decision was made to apply a common model across all measurement plots to ensure comparability, and to apply the simplest model possible which provided at least a satisfactory fit; therefore R_{ECO} was modelled solely using soil temperature. The model ratings are provided in Table A3 in Appendix A. The GPP models also provided a generally good fit of modelled against measured fluxes; however, modelled fluxes showed a slight underestimation (Fig. 2.2) in the TF and SF, where peak productivity during the summer was not fully captured. The spatial scale of this study meant that it was not possible to obtain sufficient measurements to derive light-response curves for each sampling interval, but incorporation of temperature as an additional explanatory variable (Eq. 2.3) significantly improved GPP model performance, and thus appears to have provided a suitable proxy for seasonal variability in light response. More complex seasonality indices, such as the effective temperature sum index of Tuittila et al. (2000), did not lead to a significant additional improvement in model fit.

The relationship between observed and modelled CH_4 fluxes showed some underestimation of higher fluxes (Fig. 2.3) and was rated at least 'satisfactory' for all vegetation types except at the TF-control and RP-bare soil plots (Appendix A). In the TF-control plot the model failed to capture high summer CH_4 fluxes, whilst the small variability in fluxes measured at the RP-bare soil plot (-0.05 to 0.03 mg CH_4 m^{-2} hr^{-1}) meant that no relationship with environmental variables could be established. Thus linear interpolation was used to produce annual flux estimates for the TF-control and RP-bare soil plots.
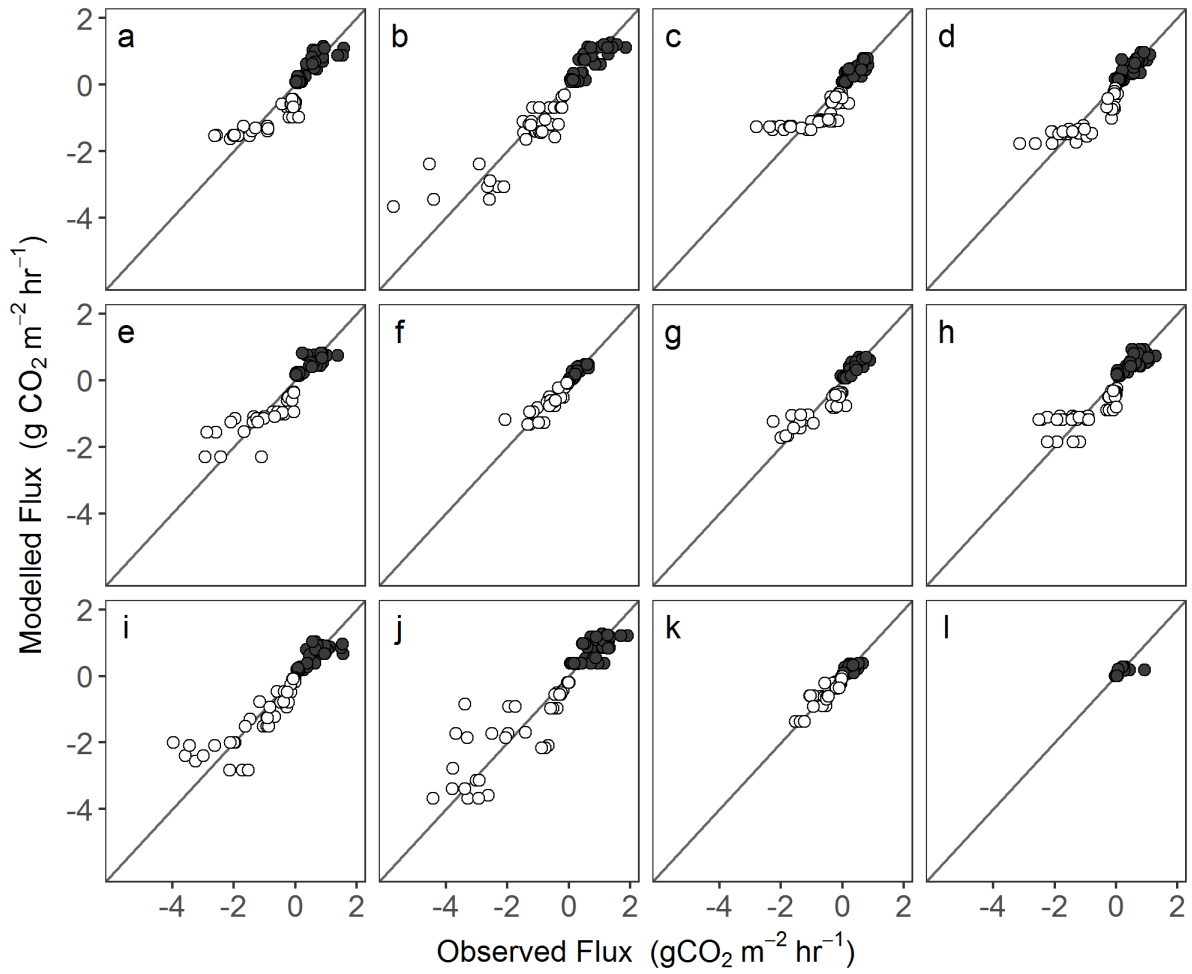
Figure 2.2. Modelled v Measured CO₂ fluxes (filled circles represent R_{ECO}, open circles represent GPP) for (a) TF-control Phragmites; (b) TF-control Cladium; (c) TF-burnt; (d) TF-cut; (e) SF-control Sphagnum; (f) SF-control brown moss; (g) SF-control Juncus; (h) SF-burnt; (i) SF-cut;(j) RP-control; (k) RP-recolonising; (l) RP-bare soil. The solid line represents the 1:1 line.
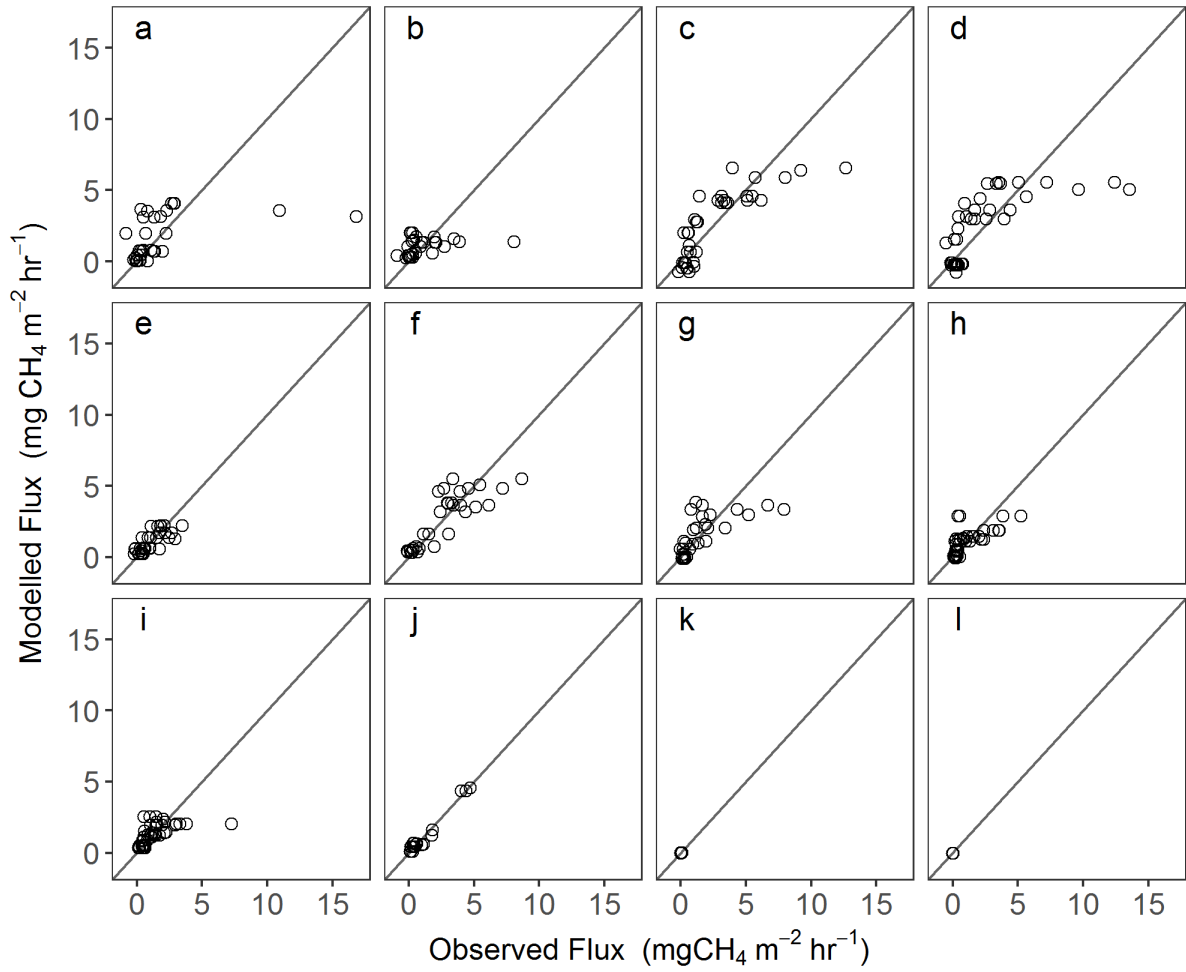
Figure 2.3. Modelled v measured CH₄ fluxes for (a) TF-control Phragmites; (b) TF-control Cladium; (c) TF-burnt; (d) TF-cut; (e) SF-control Sphagnum; (f) SF-control brown moss; (g) SF-control Juncus; (h) SF-burnt; (i) SF-cut;(j) RP-control; (k) RP-recolonising; (l) RP-bare soil. The solid line represents the 1:1 line.

2.3.3. CO_2 fluxes

CO_2 fluxes showed a clear seasonality throughout the study period, with CO_2 uptake greatest between May and September (Fig. 2.4). The TF-control plot acted as a net sink of CO_2 (-178 ±16 g C m^{-2} yr^{-1}), with an increase in net CO_2 sequestration observed at both the burnt and hand-cut plots (-305 ±49 g C m^{-2} yr^{-1} and -266 ±47 g C m^{-2} yr^{-1} respectively; Table 2.2, Fig. 2.6). The SF-control plot also acted as a net CO_2 sink (-130 ±48 g C m^{-2} yr^{-1}) whilst the plots which underwent vegetation removal were small to moderate net sources of CO_2 (28 ±5 and 235 ±52 g C m^{-2} yr^{-1} for the SF-burnt and SF-cut plot respectively). Although no significant differences in NEE were observed between the control and intervention plots in either the tall or the short fen, the *Cladium* dominated collars in the TF-control plot acted as a greater net CO_2 sink compared to the TF-cut plot ($p < 0.05$). R_{ECO} was greater in the SF-cut plot compared to the SF-control plot ($p < 0.05$), with further investigation showing these differences were largely associated with the *Juncus* collars ($p < 0.05$) and brown moss collars ($p < 0.01$). Within the restored pasture, the bare soil plot acted as a net source of CO_2, whereas a net CO_2 sink was observed at the control and recolonising plots, though the latter was small (Table 2.2, Fig. 2.6). The RP-control plot acted as a significantly greater CO_2 sink compared to the RP intervention plots ($p < 0.01$).

2.3.4. CH_4 fluxes

CH_4 fluxes showed a similar but less pronounced seasonal pattern compared to the CO_2 fluxes, with the highest fluxes seen from April to September (Fig. 2.5). All measurement plots acted as a net source of CH_4, with the highest fluxes occurring in the SF-control and TF intervention plots (Table 2.2, Fig. 2.6). Again, significant differences in CH_4 fluxes were observed between vegetation types rather than between treatments within the tall and short fens, with the TF-burnt plot having higher CH_4 fluxes compared to the *Cladium* collars within the TF-control plot ($p < 0.01$). Within the restored pasture, the control plot was a moderate source of CH_4 (5.6 g C m^{-2} yr^{-1}), and fluxes were significantly higher than those from the intervention plots ($p < 0.01$) which were negligible (0.1 and -0.1 g C m^{-2} yr^{-1} for the RP-recolonising and RP-bare soil plot respectively).
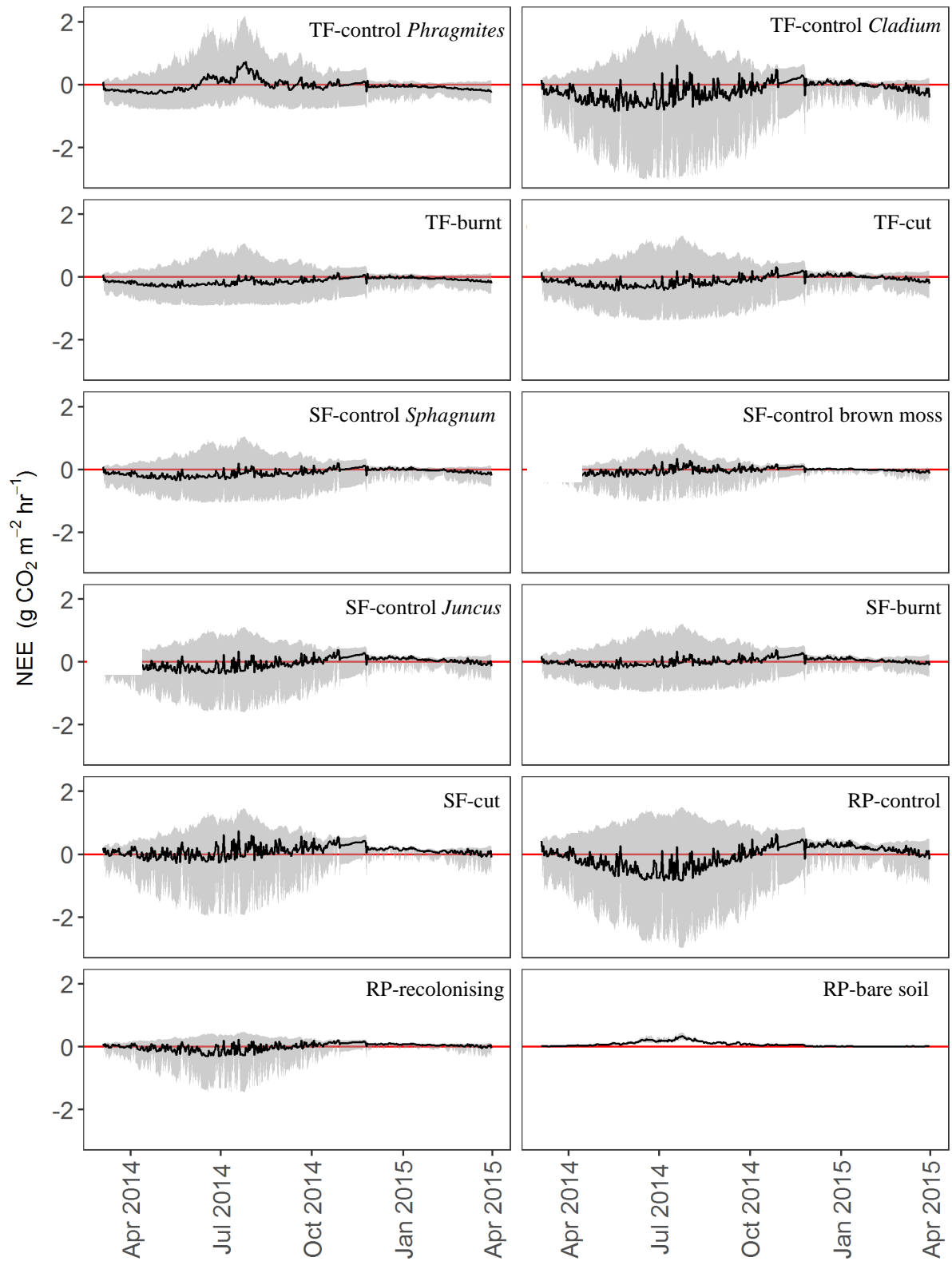
Figure 2.4. Mean (solid line), minimum and maximum (shading) daily modelled NEE fluxes for each vegetation type.
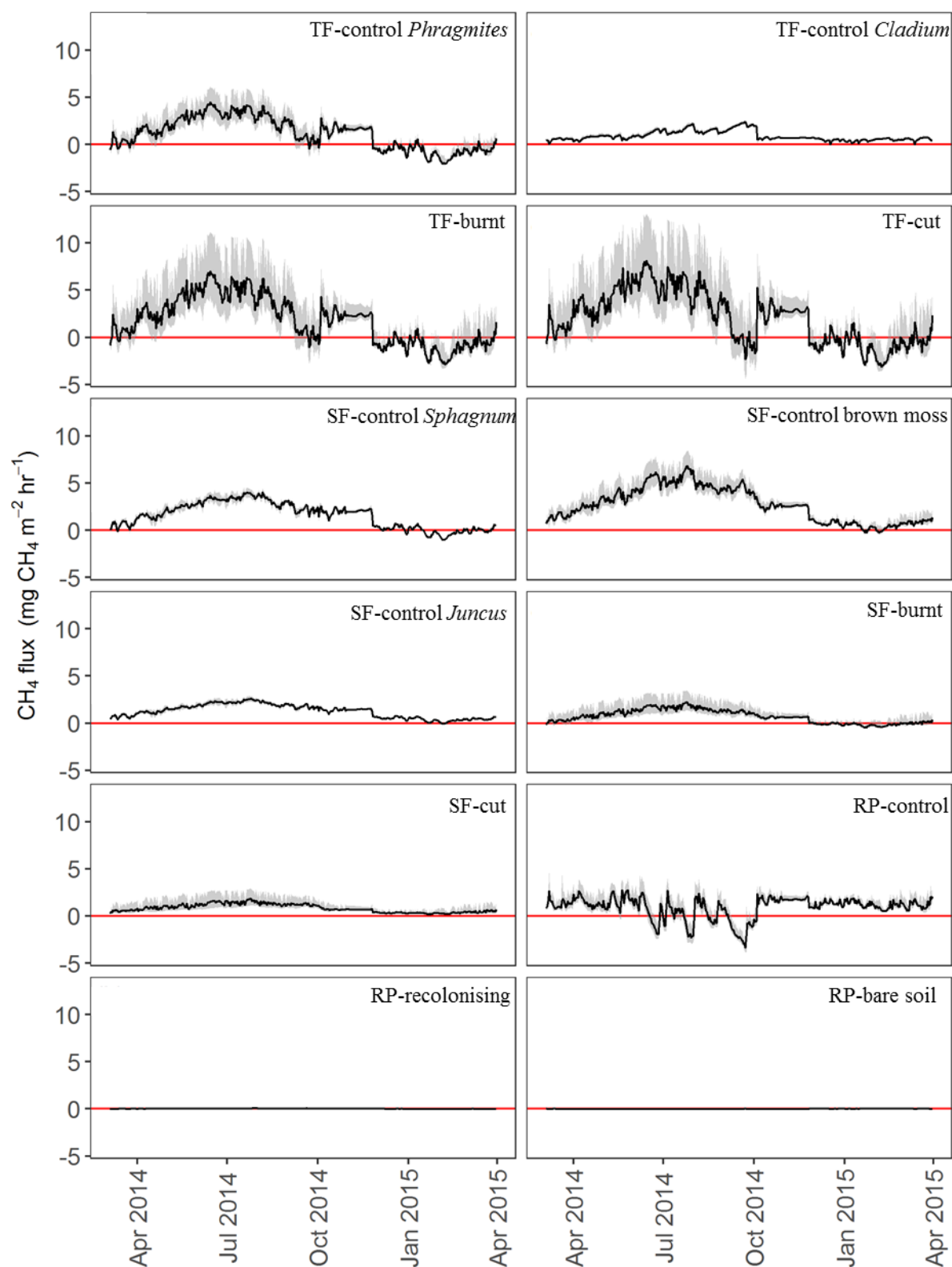
Figure 2.5. Mean (solid line), minimum and maximum (shading) daily modelled CH₄ fluxes for each vegetation type.

*Table 2.2. Annual budget of measurement plots for R_{ECO}, GPP, NEE, CH$_4$, and net GHG exchange using the 2007 IPCC standards with a radiative forcing of 25 for CH$_4$ and a time horizon of 100 years (Forster et al., 2007). ± 1 Standard error is shown in brackets. * denotes the annual CH$_4$ budget was calculated using interpolation rather than modelling*

Treatment	R_{ECO} (t C ha^{-1} yr^{-1})	GPP (t C ha^{-1} yr^{-1})	NEE (t C ha^{-1} yr^{-1})	CH$_4$ (t C ha^{-1} yr^{-1})	Net GHG (t CO$_2$-eq ha^{-1} yr^{-1})
Tall Fen					
TF Control	12.26 (1.33)	-14.04 (3.54)	-1.78 (0.49)	0.12 (0.04)	-2.50 (1.10)
TF *Phragmites* *	11.89 (0.95)	-12.63 (2.16)	-0.75 (0.14)	0.140 (0.03)	1.94 (0.59)
TF *Cladium* *	13.37 (0.97)	-18.27 (3.39)	-4.90 (0.98)	0.064 (0.02)	-15.81 (5.08)
TF burnt	7.33 (0.46)	-10.38 (1.97)	-3.05 (0.61)	0.115 (0.001)	-7.36 (1.59)
TF cut grassland	10.52 (0.53)	-13.19 (1.98)	-2.66 (0.42)	0.133 (0.01)	-5.34 (1.02)
Short Fen					
SF control	7.14 (0.95)	-8.44 (2.84)	-1.30 (0.47)	0.119 (0.20)	-0.78 (0.31)
SF *Sphagnum*	9.94 (0.93)	-10.83 (2.08)	-0.89 (0.19)	0.081 (0.007)	-0.55 (0.13)
SF brown moss	4.37 (0.30)	-4.96 (0.96)	-0.59 (0.12)	0.176 (0.01)	3.72 (0.80)
SF *Juncus*	7.11 (0.47)	-9.53 (1.88)	-2.42 (0.50)	0.101 (0.01)	-5.50 (1.35)
SF burnt	10.55 (0.77)	-10.27 (1.87)	0.28 (0.05)	0.046 (0.004)	2.56 (0.55)
SF cut grassland	12.41 (0.90)	-10.07 (1.73)	2.35 (0.44)	0.055 (0.005)	10.42 (2.20)
Restored Pasture					
RP control	17.34 (1.26)	-19.39 (3.99)	-2.05 (0.45)	0.056 (0.002)	-5.65 (1.25)
RP recolonised	4.71 (0.47)	-4.73 (0.93)	-0.02 (0.004)	0.001 (0.0003)	-0.03 (0.01)
RP bare soil *	1.84 (0.52)	0.00	1.84 (0.52)	-0.001 (0.0002)	6.71 (2.52)

2.3.5. Net GHG balance (in CO_2-equivalents)

CO_2 exchange dominated the net GHG balance in the majority of the study plots, with the overall GHG balance of the TF-control and intervention plots being negative (i.e. a net cooling effect), due to the large net CO_2 uptake (Table 2.2, Fig. 2.6). Although the TF-control plot had a net cooling effect when the *Phragmites* and *Cladium* fluxes were area weighted, the *Phragmites* vegetation (which covered 75 % of the TF-control plot) had a positive GHG balance (1.94 t CO_2-eq ha^{-1} yr^{-1}), dominated by the high CH_4 emissions observed. Similarly, within the SF-control plot, the C balance of the brown moss hollows was dominated by high CH_4 fluxes, which together with low CO_2 exchange resulted in a net warming effect (Table 2.2). Again, following the area weighting of the different vegetation types within the SF-control plot, a negative GHG balance was observed due to the larger net cooling effect of the *Juncus* and *Sphagnum* hummock collars (Fig. 2.6). Both SF intervention plots had an overall net warming effect, with a GHG balance of 2.56 and 10.42 t CO_2-eq ha^{-1} yr^{-1} for the burnt and cut plot respectively. The net GHG balance of the restored pasture varied from a net cooling effect in the control plot (-5.65 t CO_2-eq ha^{-1} yr^{-1}) to neutral for the recolonising plot (-0.03 t CO_2-eq ha^{-1} yr^{-1}) to a net warming effect in the bare soil plot (6.71 t CO_2-eq ha^{-1} yr^{-1}).
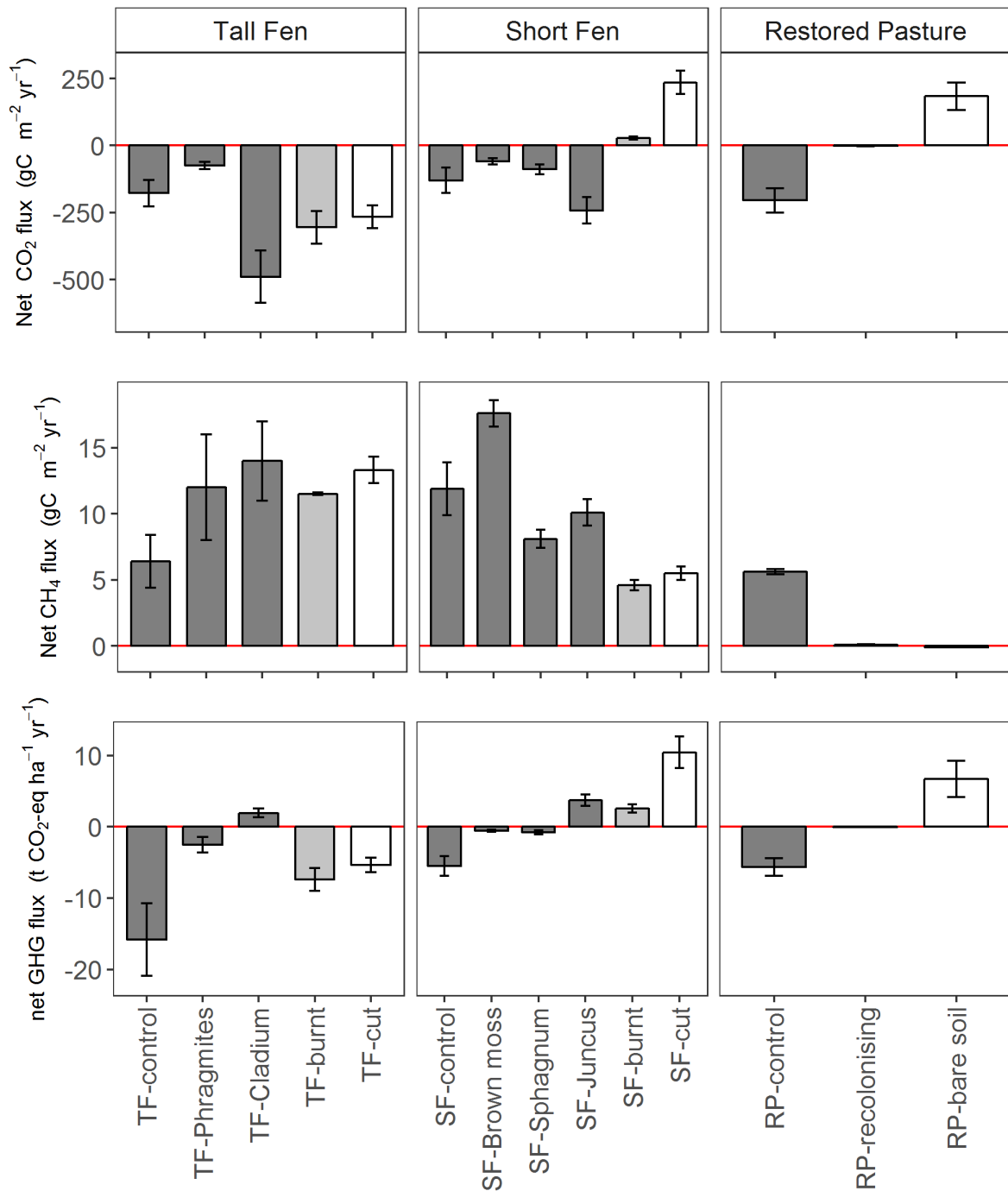
Figure 2.6. Annual (a) CO_2 (NEE); (b) CH_4; (c) and net GHG flux for each measurement plot at the tall fen (TF), short fen (SF) and restored pasture (RP). Error bars represent ± 1 model standard error.

2.3.6. Correlations between annual GHG emissions and site parameters

Annual CO_2 fluxes from vegetation plots showed weak to moderate correlation with surface peat characteristics (Fig. 2.7), with the strongest correlation between NEE and EC (rho = 0.67, p = 0.06) where CO_2 emissions increased with EC. On the other hand, annual CH_4 emissions demonstrated stronger correlations with peat characteristics, with strong negative correlations with pH (rho = 0.67, p = 0.07) and EC (rho = -0.88, p = 0.003). Annual NEE showed a moderate decrease (i.e. greater net CO_2 uptake) as the water table approached the surface, although this was not statistically significant (Fig. 2.8). R_{ECO} also showed a moderate correlation with water table, with an increase in CO_2 emissions observed as the water table height decreased (Fig. 2.8). Although this correlation was not statistically significant when all vegetation groups were considered, the omission of the RP-bare soil plot from this analysis resulted in a strong statistically significant correlation (rho = -0.745, p = 0.01). Similarly, the correlation between CH_4 emissions and water table depth increased when the RP intervention plots were omitted from the analysis, with an increase in CH_4 emissions observed as the water table level increased (rho = 0.770, p = 0.01). Annual CH_4 emissions did not correlate well with annual GPP and R_{ECO} fluxes (Fig. 2.8), but significantly with annual NEE (rho = -0.7, p = 0.04) with higher CH_4 emissions where net CO_2 uptake was greater.
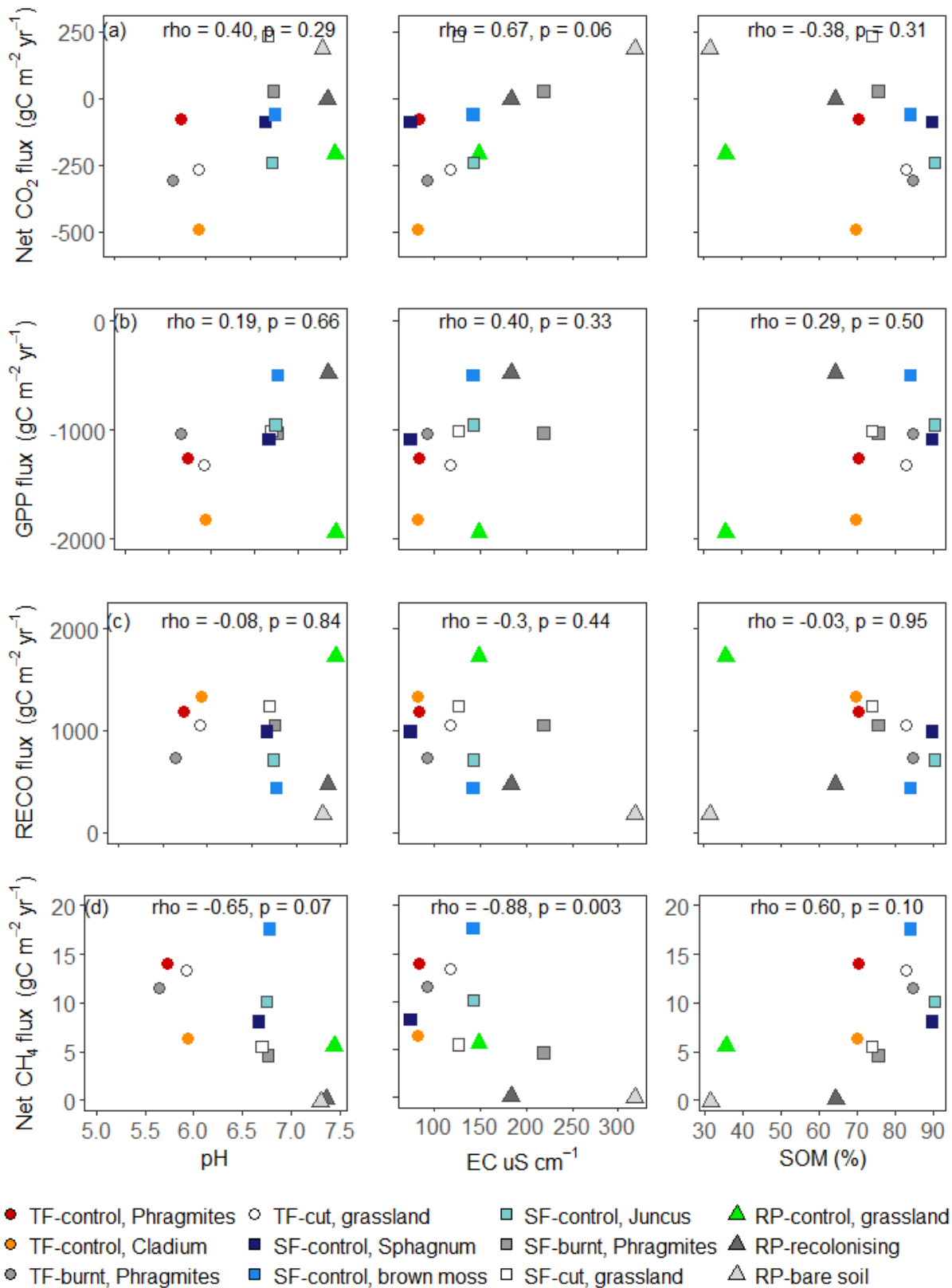
Figure 2.7. Scatter plots of annual (a) NEE; (b) GPP; (c) R_{ECO}; and (d) CH_4 fluxes and peat characteristics; pH, electrical conductivity (EC) and organic matter content (SOM) for all vegetation groups in the measurement plots. Spearman's rank correlation co-efficient (rho) and significance value (p) are shown.
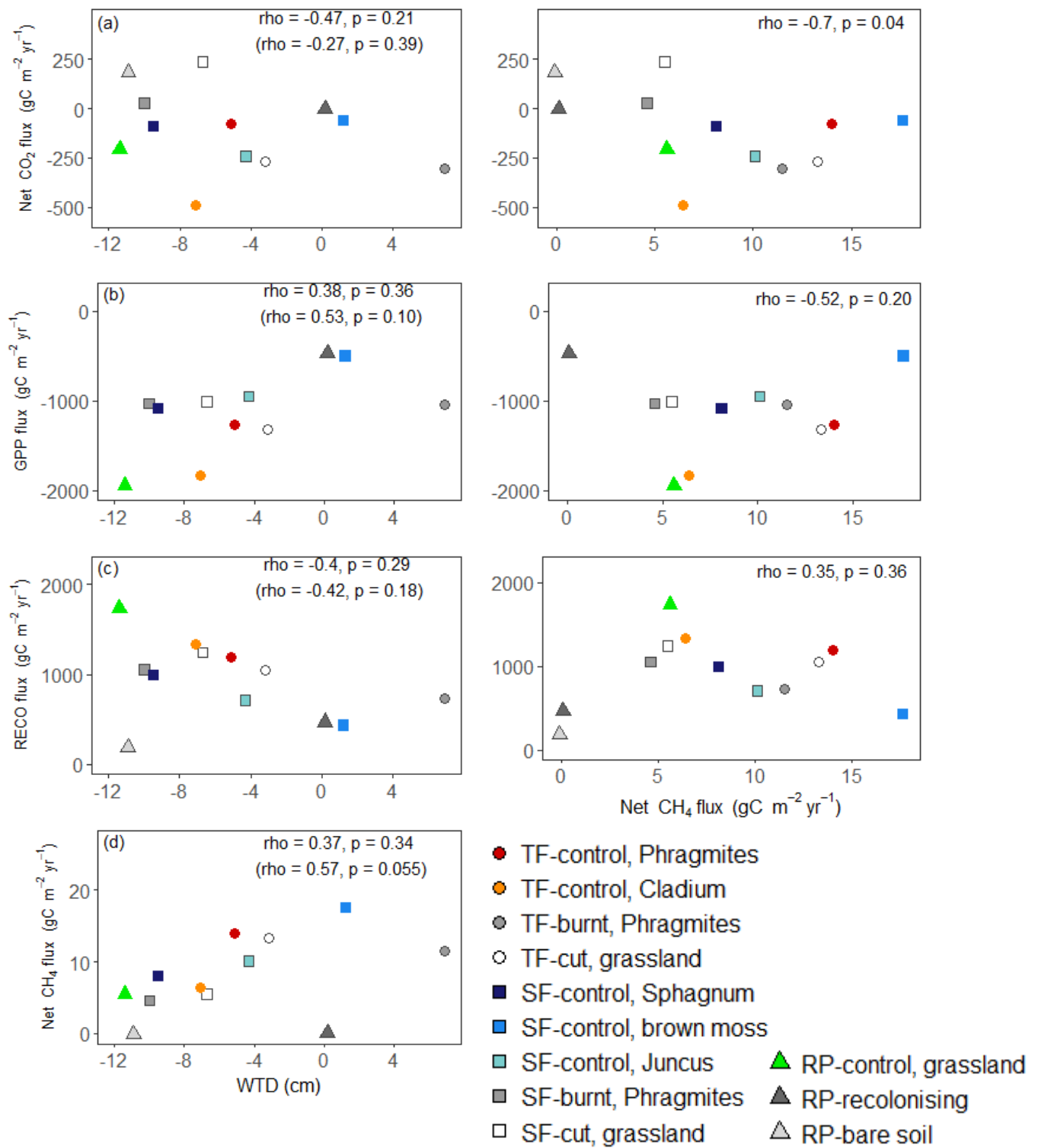
Figure 2.8. Scatter plots of annual (a) NEE; (b) GPP; (c) R_{ECO}; and (d) CH_4 fluxes and mean annual water table depth (WTD) for all vegetation types in measurement plots. Spearman's rank correlation co-efficient (rho) and significance value (p). Rho and p in brackets are for vegetation groups.

2.4. Discussion

2.4.1. Model performance and limitations

A common limitation of modelling annual fluxes from chamber measurements is the bias towards daytime measurements, which leads to not capturing the full diurnal temperature and PAR range. In this study all flux measurements were carried out between 10:00 – 16:00 and consequently modelling could have benefitted from diurnal campaigns or the use of partial shading of chambers to create a range of PAR conditions (e.g. Elsgaard et al., 2012; Gatis et al., 2016). More frequent field measurements would have also enabled higher flexibility when modelling with regards to GHG responses to changing environmental conditions, however given the total number of collars measured across Cors Erddreiniog during each campaign (n=40) and the logistical demands of chamber based approaches to GHG measurement, it was not possible to intensify sampling frequency without sacrificing spatial extent or replication. The difficulty of capturing fine scale temporal dynamics of gas fluxes can be overcome using eddy covariance techniques which can provide near continuous coverage at the landscape scale and have been successfully deployed in various peatland systems (e.g. Veenendal et al., 2007; Hendriks et al., 2010). The integration of fluxes at the landscape scale by eddy covariance methods, however, means it is a challenge to spatially disaggregate data and therefore systems are required to be homogenous and under flat terrain to reduce uncertainty (Schrier-Uijl et al., 2010). Thus, for this study, chamber measurements were the most appropriate technique to capture the small scale variability in GHG fluxes from heterogeneous vegetation cover as well as fluxes from discrete measurement plots.

The use of temperature as the sole explanatory variable for modelling R_{ECO} was used to ensure comparability between measurement plots. Significant relationships between R_{ECO} and WTD or soil moisture have been found in studies (e.g. Ruitta et al., 2007; Juszczak et al., 2013), however, WTD only improved model fits at the RP and SF-burnt plots. The inclusion of water table has been found not to significantly improve R_{ECO} models fits in drained peatland studies where the water table is too low (e.g. Elsgaard et al., 2012 and Beetz et al., 2013) as well as where the water table is stable and fluctuations are too small to improve model fits (e.g. Petersen et al., 2012). Whilst spring and summer R_{ECO} fluxes were well captured by R_{ECO} models in this study, the decrease in modelled flux rates during autumn lagged that observed in the measured values. Overestimations of R_{ECO} have also been reported by Görres et al. (2014) and Berglund et al. (2010) who found exponential R_{ECO} models driven solely by temperature tended to overestimate R_{ECO} at air temperatures of below 10 °C. In the current study, campaigns

conducted when air temperatures were below 10 °C coincided with water table levels recorded within 15 cm of the peat surface, which could explain the stronger decrease in R_{ECO} than that modelled based on temperature alone.

Similarly, GPP models did not fully capture the seasonality of GPP fluxes, underestimating peak productivity in the summer. Many previous studies have based GPP models solely on the light response curve (e.g. Beetz et al., 2013; Beyer and Hoper, 2015; Minke et al., 2016); however, these more intensive field measurements were able to develop separate light response curves for each campaign, extrapolating parameters between campaigns. This approach captures seasonal changes in parameters, but does not determine the environmental variables that control this seasonal variation. The use of temperature as a proxy for seasonal variation significantly improved GPP model fits indicating the long term variation in light response throughout the year was captured.

The modelling of annual CH_4 fluxes is uncommon in the literature, with many studies interpolating between measurement campaigns (e.g. Beetz et al., 2013; Beyer et al., 2015) due to frequently poor fits between observed CH_4 fluxes and measured environmental variables. Linear interpolation of CH_4 fluxes can result in overestimations in annual budgets due to high summer observations (Schrier-Uijl et al., 2008), though the impact of this is reduced as the time between interpolated values is reduced. Modelled annual CH_4 fluxes are generally based on an exponential relationship with temperature (e.g. Hendriks et al., 2007; Schrier-Uijl et al., 2010; Minke et al., 2016); however, the addition of WTD and PAR significantly improved model fits in this study. The lack of satisfactory CH_4 model fit in the RP-bare soil plot was likely due to the very small fluxes measured from which no significant correlation with environmental variables could be established. The unsatisfactory CH_4 model fits at the TF-control plot may be explained by the high variability in fluxes.

2.4.2. Response to vegetation removal

Vegetation removal is an important first step in restoring degraded peatlands in order to remove successive vegetation and promote the colonisation of peat forming vegetation and rare fen species. The way in which vegetation is removed is often site specific; depending on the type of vegetation to be removed as well as historical practice. In this study, plots subjected to the same vegetation removal technique were subsequently dominated by similar vegetation but showed opposing annual C balances, with the SF intervention plots both acting as net C

sources, whilst those within the TF were net C sinks. Furthermore, there was no statistically significant treatment effect between control and intervention plots at either the TF or SF, though differences were observed between specific vegetation types suggesting that the heterogeneity observed in CO_2 and CH_4 fluxes was driven by plot conditions rather than management.

No studies could be found within the literature regarding biomass removal through burning on degraded fens. Both the TF-burnt and SF-burnt plots were dominated by *Phragmites*, which is often a rapid coloniser where water tables are high (Havens et al., 2003). Studies measuring GHG fluxes from *Phragmites* dominated stands report annual CO_2 balances of -528 g C m^{-2} yr^{-1} to +68 ±94 g C m^{-2} yr^{-1} (Minke et al., 2016; Van den Berg et al., 2016; Gunther et al., 2015), with the NEE of the SF-burnt and *Phragmites* dominated collars in the TF-control plot at the upper end of this range (+28 and -75 g C m^{-2} yr^{-1} respectively). The annual NEE of the TF-burnt plot was also within this range, but was towards the lower end (-305 g C m^{-2} yr^{-1}), acting as one of the strongest net CO_2 sinks across the whole study area. The high net CO_2 uptake at the TF-burnt plot was driven by low respiration; the annual R_{ECO} was 30 to 40 % lower than that observed from the other *Phragmites* collars. This can be explained by the inundated conditions which persisted at the TF-burnt plot throughout the majority of the year which would have suppressed heterotrophic respiration, a finding similar to that of Minke et al. (2016) where an annual CO_2 net uptake of -528 and -329 g C m^{-2} yr^{-1} was observed at a *Phragmites-Carex* site in a rewetted mesotrophic fen. In contrast, the SF-burnt and TF-control plots experienced summer drawdown of the water table almost two months earlier that the TF-burnt plot, allowing greater aeration of the peat and thus greater R_{ECO}. In the SF-burnt plot this led to the system being a small net CO_2 source over the study period, whilst the *Phragmites* collars in the TF-control plot were a small net sink.

Annual CH_4 emissions from the *Phragmites* dominated collars in this study ranged from 5 to 14 g C m^{-2} yr^{-1}, in line with Gunther et al. (2015) who reported CH_4 emissions of 1 to 11 ±2 g C m^{-2} yr^{-1} from a rewetted temperate fen in north-eastern Germany. The presence of aerenchymatous tissue in *Phragmites* enables CH_4 to bypass the oxygenated water column or aerated peat, thus increasing the amount of CH_4 released to the atmosphere. CH_4 emissions from the TF-burnt plot and the *Phragmites* collars of the TF-control plot were similar despite the water table being slightly higher at the TF-burnt plot, highlighting the chimney-like effect of the *Phragmites* culms in bypassing the oxidation zone. The annual CH_4 balance of the SF-

burnt plot was less than half of that from the *Phragmites* dominated collars in the TF plots, being at the lower end of values reported in the literature. Soil temperature exerts a considerable control on CH_4 production, with an increase in microbial-mediated CH_4 production as temperatures increase (Lai, 2009). At the SF-burnt plot, the warmer temperatures observed from April to October were associated with low water tables (10 to 30 cm below the surface), meaning that although CH_4 production during this time may have been similar to that in the TF plots, there was a greater opportunity for CH_4 oxidation at the SF-burnt plot compared to the TF plots where the water tables were within 10 cm of the surface on all but one sampling occasion. Furthermore, the *Phragmites* coverage in the SF-burnt plot was less dense compared to the TF plots, potentially limiting the amount of plant-mediated transport in the SF-burnt plot compared to the TF plots. Although some studies suggest that less dense coverage of *Phragmites* might increase CH_4 emissions due to the prevalence of ebullition, which is a faster transport mechanism compared to internal plant transport (Kankaala et al., 2004; van den Berg et al., 2016), the low summer water table at the SF-burnt plot means bubbles of CH_4 are released further down the soil profile and therefore more likely to undergo CH_4 oxidation. This potential limit on ebullition as a transport mechanism of CH_4 to the surface, together with a lower density of *Phragmites* compared to the TF-burnt and TF-control plots may explain why annual emissions are so low from SF-burnt.

Peat soils under grassland management in Europe predominately show a net loss of C, with a recent synthesis of studies in Germany reporting a net C loss of 6.8 ±3.7 t C ha^{-1} yr^{-1} from nutrient rich shallow drained peatlands (Tiemeyer et al., 2016). The net loss of CO_2 from the SF-cut plots is in line with studies on shallow-drained grasslands on peat (e.g. Leiber-Sauheitl et al., 2014; Renou-Wilson et al., 2014; Poyda et al., 2016), with annual NEE values at the lower end of reported values. This may be due to the annual water table being relatively close to the surface (-6.7 cm) whereas shallow-drained peatlands can describe systems with mean annual water tables up to 30 cm below the surface (IPCC, 2014). The SF-cut plot had the highest R_{ECO} of the SF, with emissions being significantly higher than those from the SF-control plot ($p \leq 0.05$), driven by the low summer water table as well as the earlier development of vegetation which reached peak GPP in June rather than July, resulting in greater substrate availability for respiration. In contrast, the TF-cut plot was a net CO_2 sink, with a greater GPP and smaller R_{ECO} compared to the SF-cut plot. The TF-cut plot had a higher mean annual water table which was at or above the surface for 145 days compared to the SF-cut plot where water levels were at this height for just 32 days, indicating why aerobic decomposition was limited

at the former. The mean annual water table depth of -3.2 cm (range -21.5 to +3.1 cm) at the TF-cut plot is more in line with rewetted grasslands on peat rather than those under shallow-drainage, which may contribute to why the net CO_2 balance is the opposite to many grassland on peat studies (Tiemeyer et al., 2016). Furthermore, the relative hydrological isolation of the TF site from the base rich inputs at the east of Cors Erddreiniog may result in less minerotrophic conditions that those typically observed in fens and the often fertilised grasslands on peat (e.g. Beyer et al., 2015). Indeed the annual GPP and R_{ECO} values observed from the TF-cut are similar to those from nutrient poor shallow-drained grasslands reported by Renou-Wilson et al. (2014) and extensively grazed rewetted grasslands by Beetz et al. (2013).

CH_4 emissions from grasslands on peat are generally small, due to the low water table allowing greater CH_4 oxidation and the lack of plant mediated transport where shunt species are not present (Petersen et al., 2012; Henneberg et al., 2015). This was observed at the SF-cut plot, where annual CH_4 emissions were low (5.5 g C m^{-2} yr^{-1}). In contrast, CH_4 fluxes from the TF-cut plot were relatively high (13 g C m^{-2} yr^{-1}), with maximum measured fluxes of 18 mg C m^{-2} hr^{-1}. There was little difference in vegetation between the cut plots, however the water table dynamics differed, with the summer water table at the SF-cut plot ranging between -20 and -5 cm, whereas at the TF-cut plot the water table was closer to the surface (-11 to -2 cm). Consequently, the high emissions from the TF-cut plot were likely driven by high CH_4 production due to warm temperatures during the summer, coinciding with water tables within a few cm of the surface and so there was little opportunity for oxidation to occur. When corresponding water table levels were observed in the SF-cut plot, temperatures were much lower therefore CH_4 production would have been smaller and so a smaller CH_4 flux at the surface was observed.

2.4.3. Response to topsoil removal

Topsoil removal seeks to reduce the nutrient legacy of degraded fens which have become enriched from agricultural use or high nutrient floodwater, in order to successfully restore biodiversity. Following topsoil removal, the bare soil acted as a net source of C, driven by heterotrophic respiration and lack of productivity due to the absence of vegetation. As colonisation of the exposed peat occurs, primary productivity and autotrophic respiration begin; increasing SOM turnover. The recolonising plots in this study had been recovering for approximately two years before measurements commenced, and annual flux estimates indicate productivity had begun to dominate over R_{ECO}, resulting in a (modest) net sequestration of

atmospheric CO_2. In addition to a reduction in CO_2 loss following the colonisation of bare soil, the presence of calcareous rich-fen indicator species such as *Schoenus nigricans* (in the RP-recolonising collars) and *Palustriella commtata* (within the wider area) indicate the desired alkaline fen vegetation was developing (Leonard et al., 2016). The RP-control plot was found to be a moderate net sink of CO_2, contrary to many previous studies of drained grasslands on peat, which generally show significant CO_2 emissions (e.g. Tiemeyer et al., 2016). The water level in the RP-control plot was relatively high due to the close proximity to areas reprofiled during topsoil removal; mean annual water table was 12 cm below the surface, whereas managed grasslands tend to be 20 to 30 cm below the surface. Consequently, R_{ECO} may have been limited by a shallow aerobic zone within the soil profile, enabling GPP to dominate CO_2 exchange. Furthermore, the rapid greening up of the RP-control plot early in the growing season resulted in a persistent net uptake in CO_2 from March to September 2014. Although a moderate net C uptake was observed at the RP-control plot, the grazing management across the site, which was not quantified during this study, could significantly impact the net ecosystem C balance (NECB). The grassland vegetation at the RP-control plot is amongst the best grazing land throughout the fen, which together with the year round subsurface water table, may mean it is preferentially grazed, resulting in low litter delivery to the peat, thus caution needs to be taken when interpreting this annual net C flux.

CH_4 fluxes from the bare soil plot were negligible, in line with low fluxes measured from bare peat in extraction sites (e.g. Waddington et al., 2002; Vaneslow-Algan et al., 2015), eroded upland bogs (Cooper et al., 2014) and agricultural mesocosm experiments (e.g. Harpenslager et al., 2015). The absence of vegetation severely limits substrate availability for methanogenesis, which relies upon plant derived litter and root exudates (Whiting and Chanton, 1993), thus the limitation in energy input would have contributed towards the near zero fluxes at the RP-bare soil plot. CH_4 fluxes from the RP-recolonising plot were similar to those from the RP-bare soil plot and surprisingly low, given the water table position at the surface throughout the year and the presence of vegetation which could provide substrate for methanogenesis (Wilson et al., 2009). A similar finding was observed by Bhullar et al. (2014) where CH_4 fluxes remained low 5 years after topsoil removal due to reduced concentrations of labile carbon in the substrate. The lag in substrate recovery following topsoil removal (or in cutover peatlands) is likely dependent upon the species composition of colonising vegetation, with higher CH_4 emissions expected from more productive peatland communities (Waddington and Day, 2007; Wilson et al., 2009). Furthermore, the topsoil removal undertaken at the RP

would have removed a large part of the soil microbial community, with the development of the new microbial community in the RP intervention plots a slow process (e.g. Croft et al., 2001). It is expected that CH_4 emissions will increase with time from the RP-recolonising plot as vegetation and microbial populations develop, though there is little consensus within the literature on whether emissions from rewetted cutover peatlands can reach or exceed levels seen in pristine peatlands (Tuitilla et al., 2000; Vaneslow-Algan et al., 2015).

2.4.4. Annual flux drivers and implications for restoration

The principal aim of this study was to investigate the effect of restoration interventions in CO_2 and CH_4 fluxes from a lowland fen, however the spatial heterogeneity in gas fluxes appears to be at a smaller scale than the treatment plots established in this study. Within the TF-control plot the *Cladium* dominated collars were a considerably stronger net CO_2 sink compared to the *Phragmites* dominated collars, driven by high productivity consistent with its evergreen properties enabling year round photosynthesis (Saltmarsh et al., 2006). In addition to this, the absence of aerenchymatous tissue in *Cladium* means there is no internal plant transportation of CH_4, therefore fluxes were considerably lower than those from the *Phragmites* collars. Similarly, within the SF control plot, the lack of plant mediated transport and lower water tables at the *Sphagnum* hummocks resulted in low CH_4 fluxes compared to the *Juncus* collars where the shunt species dominated, and the brown moss hollows which were inundated for nine months of the year. The high CH_4 fluxes from the *Juncus* collars in this study are comparable with those reported by Shafer et al. (2012) where average fluxes of up to 3.3 mg CH_4 m^{-2} hr^{-1} were measured from *Juncus* dominated stands in Danish permanent grasslands on peat soils. The *Juncus* collars were also the strongest net CO_2 sink of the SF-control plot, with a relatively shallow water table potentially limiting aerobic R_{ECO}, enabling GPP to dominate the CO_2 balance.

Vegetation composition can be used as a predictor for GHG fluxes, such as where specific plant functional traits are well associated with fluxes e.g. species with aerenchymatous tissue are associated with high CH_4 fluxes (Whalen, 2005; Couwenberg et al., 2009). However, the fine scale spatial heterogeneity and interaction with environmental conditions means that vegetation cover is not always a reliable indicator of GHG fluxes, such as in this study where three groups of *Phragmites* collars ranged from CO_2 sources to moderate CO_2 sinks. Water level is considered the principal driver of annual GHG emissions at the landscape level, with relationships between annual CO_2 fluxes and mean annual water table described by a number

of studies. Renou-Wilson et al. (2016) found a significant reduction in NEE (i.e. smaller net CO_2 sink) as water levels increased at the plot level within shallow-drained and rewetted grasslands on peat soil in Ireland. Although a similar pattern was observed in this study considering the vegetation groups across the entire fen, water level was found to correlate best with annual R_{ECO}, though the RP- bare soil plot did not conform as a low annual R_{ECO} flux was observed despite a mean annual water table of -11 cm. The removal of nutrient enriched topsoil is an extreme restoration measure and the subsequent bare surface represents a temporary state within the overall restoration process; demonstrated by the recolonisation and subsequent CO_2 uptake at the RP-recolonising plot within two years of intervention. The removal from the RP-bare soil plot from this part of the analysis enables the relationship between water table and annual R_{ECO} to be assessed for established vegetation; the remaining vegetation groups across the fen represent a substantial range of the ecology found across semi-natural and conservation managed fens in the UK. A significant reduction in annual R_{ECO} with rising water tables was found when the RP- bare soil plot was omitted from analysis (Fig. 2.7), concurrent with findings in the literature that aerobic respiration becomes limited where water levels are close to the surface (e.g. Tuittila et al., 1999, Wilson et al., 2016b). This water table control on R_{ECO} in part drives the differences observed in NEE, which although not significantly correlated to water table, fit well with other studies reported from UK peatlands which encompass a larger range of management and water table (e.g. Evans et al., 2017).

A mean annual water level ±10 cm of the surface is considered the optimum for mitigating GHG emissions (Couwenberg et al., 2011). This study found vegetation groups acted as net CO_2 sinks where the water table was within 6 cm of the surface; a finding in accordance with Renou-Wilson et al. (2016) who observed net CO_2 uptake at grazed rewetted grasslands in Ireland when the mean annual water table was within 5 cm of the surface. Although a water table just below the surface is recommended where peatlands are to be considered a GHG mitigation tool, high water tables can result in high CH_4 fluxes which can potentially result in a positive GHG balance, offsetting CO_2 'savings' as observed in the *Phragmites* collars in the TF-control plot and brown moss hollows in the SF-control plot. High variability was observed in NEE (-205 to + 234 g C m^{-2} yr^{-1}) where annual water tables were between 6 and 12 cm below the surface; around the depth that is considered optimum for reducing C emissions (Couwenberg et al., 2011). Of the plots which acted as net CO_2 sources within this water table range (RP-bare soil, SF-burnt, SF-cut), the RP-bare soil plot can be expected to regain its CO_2 sink function with vegetation development over time, as observed in the RP-recolonising plot.

Both SF intervention plots acted as CO_2 sources whilst the corresponding TF intervention plots, which had similar vegetation communities but higher water tables, acted as net CO_2 sinks, suggesting that emissions might be reduced if water levels were closer to the surface.

The highest annual CH_4 fluxes in this study were found where water levels were within 6 cm of the surface; agreeing with the consensus in the literature that as water tables rise towards the surface from a depth of approximately 20 cm, CH_4 emissions also increase (Tiemeyer et al., 2016; Poyda et al., 2016). Water table was strongly correlated with annual CH_4 fluxes (rho = 0.574, $p = 0.055$), with the relationship increasing when the RP intervention plots were omitted from analysis (rho = 0.77, $p = 0.01$). Given the large scale disturbance associated with topsoil removal and the unknown timescale of microbial community recovery, the CH_4 fluxes from the RP intervention plots are unlikely to fully represent the abiotic conditions currently present, justifying their removal from this analysis. The pattern between annual CH_4 emissions and WTD from the remaining vegetation groups is similar to that reported from other UK lowland peatlands (Evans et al., 2017; Renou-Wilson et al., 2016a) as well as fitting with European studies (Poyda et al., 2016). Although rewetting of peatlands aims for a water level just below the surface, the effect of inundation on CH_4 flux behaviour is less well understood, with vegetation composition rather than water table position considered the principal driver of gas exchange under inundated conditions (Blain et al., 2014; Minke et al., 2016). Only one plot had a mean annual water table above the surface (TF-burnt; +7 cm) and although CH_4 emissions were high (11.5 g C m^{-2} yr^{-1}) they were similar to those from plots with water tables approximately 4 cm below the surface (SF-control; TF-hand cut). Whilst the dominance of aerenchymatous vegetation at the TF-burnt plot may explain the high CH_4 fluxes observed, because no other plot was inundated throughout the study period the relative importance of plant mediated transport compared to water table position in controlling CH_4 emissions under flooded conditions could not be determined. The TF-burnt plot also had one of the most negative GHG balances across the study area, suggesting that under shallow inundation net CO_2 uptake can counterbalance CH_4 emissions providing there is colonisation by highly productive vegetation, though longer term monitoring would be required to assess the effects of interannual variability on the GHG balance.

The restoration of degraded peatlands typically involves rewetting to create conditions where peat accumulation can re-establish, together with the associated biodiversity and ecosystem services (Bonn et al., 2016). The impacts of rewetting peatlands on GHG fluxes is widely

documented in the literature, however, the restoration process can involve a wide range of management options such as vegetation removal or improving the quality of hydrological inputs, which will affect the GHG balance of peatlands in different ways. Consequently, caution is needed when considering the effect of 'restoration' on peatland GHG dynamics, as the rewetting of a peatland will result in a substantially different GHG response compared to a change in soil nutrient status or plant community. Vegetation removal is widespread within UK peatland restoration as a method of removing undesirable vegetation which has established through either natural succession or intentional planting, and is often complemented with extensive grazing to prevent future succession (Lunt et al., 2010). However, very little information is available on the success of this technique in promoting desirable vegetation, with no evidence on the effects of GHG dynamics. The vegetation removal in this study had no direct impact on the hydrology or soil characteristics and thus a limited effect on GHG fluxes was observed within the TF and SF. However, the change in vegetation resulting from intervention would have had some influence in GHG dynamics, for example the colonisation of *Phragmites* following burning provided a pathway for CH_4 transportation, contributing to the high CH_4 emissions observed. On the other hand, the topsoil removal in the restored pasture not only removed vegetation, but also raised the water table and reduced the soil nutrient status, resulting in a significant effect on GHG fluxes. Overall, this study suggests that a change in water level is required for a change in GHG fluxes to be observed. Further, land use management, such as vegetation removal, does not necessarily change the C balance within the short term, though shifts in vegetation composition may change GHG dynamics in the longer term.

2.5. Conclusion

The results presented in this chapter show the GHG flux response to restoration interventions across a lowland fen is complex, and vegetation removal can influence GHG flux dynamics despite not affecting the water level. In this study the method of biomass removal determined the subsequent vegetation community, with both the tall and short fen exhibiting *Phragmites* dominated stands following burning, and grassland vegetation following cutting. Despite similar vegetation, the net C balance of the intervention plots differed, with those in the short fen acting as net C sources, and the tall fen as net C sinks, likely due to the higher water tables in the tall fen intervention plots limiting R_{ECO}. Whilst this study supports the idea that water tables just below the surface enable peatlands to be used as a climate mitigation tool, high variability in fluxes was observed, which is common within peatlands which can be C sources

in single years but function as long term sinks. The removal of eutrophic peat, whilst an extreme restoration measure, appears to have been successful in creating conditions suitable for the development of calcareous fen vegetation. Although bare soil following topsoil removal was a net C source, this is likely to be a temporary effect given the small net C sink observed within two years where recolonisation had occurred. Whilst there is the potential that CH_4 emissions will increase as methanogenic communities develop, the reprofiling and revegetation at the restored pasture may initiate the development of small scale microtopography typical of lowland fens, with a range of GHG fluxes such as those observed at the short fen control plot. Peatland restoration can involve a wide range of interventions, with rewetting the most common technique used in drained peatlands, however vegetation management and nutrient removal are also key in restoring abandoned lowland fens in the UK where dereliction and nutrient enrichment have occurred. Whilst this study agrees with the literature that water table is the key driver of GHG fluxes and important to restore the C sink function of peatlands, the findings highlight the importance of land use management causing shifts in vegetation community composition which can affect GHG flux dynamics in the long term.

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Chapter 3

CO_2 and CH_4 fluxes from relict peat cutting microforms following rewetting in a cutover lowland raised bog

Abstract. The removal of peat for energy production and horticulture has affected nearly all lowland raised bogs in England and Wales, contributing to a 54 % reduction in extent within Wales. Peat cutting, either by hand or mechanically, first requires drainage which leads to a net loss of stored carbon (C) as aerobic decomposition accelerates. Abandoned cutover raised bogs may retain semi-natural vegetation, but the lack of hydrological recovery maintains the net loss of C. Rewetting through ditch-blocking and the creation of bund cells is used to re-create anaerobic conditions, aiming to reduce C losses and re-instate the C sink function. This study measured CO_2 and CH_4 fluxes across an abandoned hand-cut lowland raised bog which has been rewetted using the bunding technique. Closed-chamber measurements were conducted over 18 months from rewetted and non-rewetted treatments, encompassing microforms (cutover troughs and uncut buffer ridges between cuttings) which were created during historical peat cutting practices. CH_4 fluxes were higher under wetter conditions, with significantly greater emissions from troughs compared to ridges, and from rewetted microforms compared to non-rewetted microforms. CH_4 measurements from the same microforms did not show a significant increase following rewetting which was attributed to the total inundation of vegetation and a lack of aerenchymatous species. Productivity and net CO_2 uptake was significantly greater in the ridges compared to the troughs following rewetting, linked to the enhanced productivity of bryophytes on ridges and submergence of vegetation in the troughs with rewetting. Where water tables remained low, CO_2 fluxes from corresponding microforms did not appear to differ with the rewetting treatments. The bund itself was a hotspot for CH_4 emissions where *Juncus* spp. were present, from which the greatest CO_2 exchange was also measured, with fluxes from the remaining area of the bund similar to those from the ridge microforms. Rewetting using the bunding technique appeared successful in raising the water table and creating a mosaic of higher drier ridges and lower, partially inundated areas analogous to the hummock-hollow microtopography of natural raised bogs, though it took two years for the water table to be maintained at the surface throughout the summer. The higher CH_4 fluxes following rewetting and associated with aerenchymatous vegetation on the bund is a key consideration for restoration projects as this has the potential to counterbalance the reductions in CO_2 emissions observed. Therefore, it is recommended that long term monitoring of fluxes in the years following rewetting is undertaken to assess development.

3.1. Introduction

Pristine peatlands naturally act as a net sink of CO_2 and long term store of carbon (C) as a result of a high water table that impedes the rate of belowground decomposition relative to the rate of ecosystem primary production. Despite the importance of peatlands in the C cycle, globally 80 million hectares of peatland have been destroyed through extensive drainage for agriculture, forestry and peat extraction (Joosten and Clarke, 2002). Cutover peatlands, i.e. those used for peat extraction, are estimated to account for 10 % of all drained peatlands outside of the tropics and are predominately found within the Nordic countries and Eastern Europe (Joosten and Clarke, 2002). Peat extraction within the UK is relatively small-scale; ~7500 ha of peatland are currently under commercial extraction (Bain and Goodyer, 2016), though cutting of peat by hand for domestic fuel provision was historically widespread on the margins of lowland raised bogs. The drainage and removal of surface vegetation in hand-cut peatlands severely impacts ecosystem functioning by terminating the accumulation of peat, directly destroying the peat body (Soini et al., 2010) and altering the soil structure (Hughes et al., 2007). In contrast to the homogenous flat cutting fields found under industrial peat extraction (Lindsay et al., 2014), during peat cutting by hand, alternate strips are removed leaving buffers that result in a microtopography of lowered cuttings and raised buffer strips across the landscape. The lowering of the water table leads to an efflux of CO_2 through increased aeration of the peat which leads to greater aerobic decomposition of organic matter, whilst the removal of vegetation eliminates CO_2 uptake through photosynthesis (Vaneslow-Algan et al., 2015; Waddington et al., 2002). Terrestrial CH_4 emissions are usually decreased by drainage due to increased oxidation in the overlying aerobic layer, although drainage ditches have been found to be hotspots for CH_4 release due to high algal cover (Sundh et al., 2000) or through the influence of intensive agriculture on ditch biogeochemistry (Peacock et al., 2017). The cutting of peat banks into the margins of raised bogs (including the lagg fen) can also enhance drainage, leading to subsidence of the main dome. Although peat cutting by hand in the UK was historically extensive, the cumulative effect over time has resulted in the loss of the wet lagg fen margin from nearly all UK lowland raised bogs (Lindsay et al., 2014). Within Wales, the extent of lowland raised bogs is estimated to have been reduced from 4000 ha to 1840 ha (Blackstock et al., 2010), with the majority affected by hand-cutting due to the relative accessibility of large peat reserves in comparison to upland blanket bogs (Slater, 1983).

Following the cessation of peat cutting by hand, a substantial peat layer may remain and if water tables stay low, this can act as a large persistent source of CO_2. In order to restore a bare

peat layer to a functioning peatland that is sequestering C by peat accumulation plant colonisation is required, however this is often hindered by the harsh conditions of bare peat surfaces, large water table fluctuations and the lack of a viable propagule bank (Money and Wheeler, 1999). Raising the water table through ditch blocking is a common first step in restoring drained peatlands (Wilson et al., 2013). Ditch blocking is established by the construction of multiple damns along the entire ditch length to shed water laterally and raise the water table over a large area (Grand-Clement et al., 2015). The development of anaerobic conditions during rewetting results in a reduction of CO_2 emissions as aerobic decomposition becomes inhibited, whilst CH_4 emissions generally increase as the zone of CH_4 production is moved closer to the surface and there is less opportunity for oxidation (Waddington and Day, 2007; Vaneslow-Algan et al., 2015). Rewetting can also be achieved through the construction of low lying bunds where the peatland has a low surface slope. A shallow linear trench is dug to remove the surface layer of degraded peat which is then infilled with saturated peat from an adjacent 'borrow pit' to a height of ~ 20 cm above the ground surface. The bund is then covered either with the previously removed surface layer of vegetation or brash to prevent the saturated peat from drying out. The linking of bunds to create three or four-sided cells results in both precipitation and runoff being captured within the defined cells, buffering the water table against the impact of summer drought (Money and Wheeler, 1999). The saturated peat which is brought to the surface to create the bund may provide an increase in labile substrate for decomposition, enhancing CO_2 emissions. This could be of particular importance where the cells take time to fill with water following the bund construction, as there will be a short time period where the bund becomes partially aerated, potentially creating a hotspot of CO_2 emissions.

Where bare soil surfaces exist, rewetting alone is rarely sufficient to trigger the spontaneous colonisation of peat forming vegetation. The application of *Sphagnum* spp. fragments and straw mulching have been shown to be effective at initiating plant development (Qunity and Rochefort, 2003; Järveoja et al., 2015), providing a substrate for colonisation and improving soil moisture conditions. Rewetting using bund cells has also been found to enhance *Sphagnum* spp. establishment with extensive colonisation observed within three years where cells are relatively small (~20 m by 20 m) and the water table is within 10 cm of the surface (Lunt et al., 2010). The re-establishment of vegetation has an effect on the ecosystem C balance through C inputs via photosynthesis and C losses via autotrophic respiration. Providing mire vegetation is established in cutover peatlands it is likely that the ecosystem will start to sequester C shortly

after restoration, however, an increased supply of substrates to belowground methanogenic microorganisms and high water tables may result in increased CH_4 emissions. In the case when aerenchymatous vegetation develops aerenchyma may act as a conduit for CH_4 passage to the atmosphere.

Studies of the greenhouse gas response of industrially cutover peatlands to restoration is increasing in the literature, particularly given the legislative obligations for abandoned cutover sites to be restored (e.g. in Germany; Höper et al., 2008). Whilst the impact of peat cutting by hand is less than that at the industrial scale, the extent of impacts are widespread throughout the UK, with the majority of lowland raised bogs impacted in some way. In the 2014 UK Peat restoration and management Compendium, 51 projects of lowland peatland restoration were reported, with the majority focusing on restoring the ecological and hydrological functioning of peatlands (Defra, 2008). Rewetting is a key part of the peatland restoration process and the use of low-lying bunds to achieve this is becoming more frequent in UK lowland raised bog restoration (e.g. Low Moss in East Ayrshire; BogLIFE project in Cumbria; Marches Mosses Boglife project). The construction of bunds introduces spatial heterogeneity throughout the landscape by creating raised features and associated pools. Together with the lowered cuttings and raised buffer strips created during peat cutting, the variation in surface height results in differences in water table level and vegetation, which in turn creates spatial heterogeneity in GHG flux dynamics. The cuttings and buffer strips are analogous to the hummock-hollow complexes observed in natural peatlands, however no studies could be found describing GHG fluxes from anthropogenically generated microtopography in cutover peatlands or from peatlands rewetted using bund cells.

This study aims to investigate how the heterogeneity of microforms created during hand-cutting of peat and the effects of restoration by rewetting using low-lying bunds impact upon GHG fluxes. The hypotheses of this study are: (i) lowered cuttings (referred to as troughs) will be larger sources of CH_4 compared to buffer strips (referred to as ridges) due to the water table being closer to the surface; (ii) ridges will have greater CO_2 emissions compared to troughs due to the lower water table; (iii) the bund (constructed for rewetting) will be a larger source of CO_2 and CH_4 compared to the ridges and troughs; and (iv) following rewetting CO_2 emissions will decrease, whilst CH_4 emissions will increase.

3.2. Method

3.2.1. Study area

The study was focused on Cors Fochno (4°1′W 52°30′N), one of the largest lowland raised bogs in the UK. The bog is situated on the west coast of Wales and experiences a maritime temperate climate with an minimum annual air temperature of 6.7 °C, maximum of 13.5 °C, and annual precipitation of 1075 mm (Gogerddan meteorological station, 4km from Cors Fochno, 31m above sea level). Approximately 60 % of the original bog area was drained during the 19th century for agriculture and peat cutting, with the latter focused around the southern and western margins.

A restoration area was identified in the south-western region of Cors Fochno, known as Pant-y-Dwn (Fig. 3.1a), which is recognised as a degraded raised bog still capable of natural regeneration (Annex I habitat 7120; EU Habitats Directive 97/62/EC). Restoration activities included the establishment of a low lying bund to rewet the area in two stages. In February 2015, the bund and cells were established over a length of ~ 200 m, and in February 2016 the bund was extended to restore the entire Pant-y-Dwn area (Fig. 3.1b). The process of constructing the bund involved removing the surface vegetation layer to expose the underlying saturated peat. The peat was then excavated and piled up to create a bund of approximately 30 cm high and 50 cm wide, which was then overlain with the removed surface layer and vegetation.

The first treatment was established following the bund construction in February 2015 and contained three microforms; (a) troughs (lowered past peat cuttings); (b) ridges (buffer strips where peat was not cut); and (c) the bund itself. This treatment area will be referred to as rewetted-2015 (RW-15, Table 3.1). The second treatment, an area of Pant-y-Dwn that was not initially rewetted, included two microforms: (a) troughs; and (b) ridges. In February 2016, the Pant-y-Dwn bund was extended and rewetted the second treatment area, therefore this treatment area will be referred to as rewetted-2016 (RW-16, Table 3.1), with reference to measurements taken before rewetting and after rewetting.

Triplicate soil samples were collected from the five identified microforms across Pant-y-Dwn in November 2016 using a 15 cm length soil cores and soil characterisation (Table 3.1) was carried out using the methods outlined in Ch. 2 (section 2.2.2.).

Figure 3.1a: Location map of Pant-y-Dwn on the south-western margin of Cors Fochno, UK. (Ordnance Survey). Inset: Location of Cors Fochno in the UK.

Figure 3.1b: Site map of Pant-y-Dwn with microform measurement locations identified. The bund extent in 2015 (dashed line) representing treatment RW-15 and 2016 (dotted line)
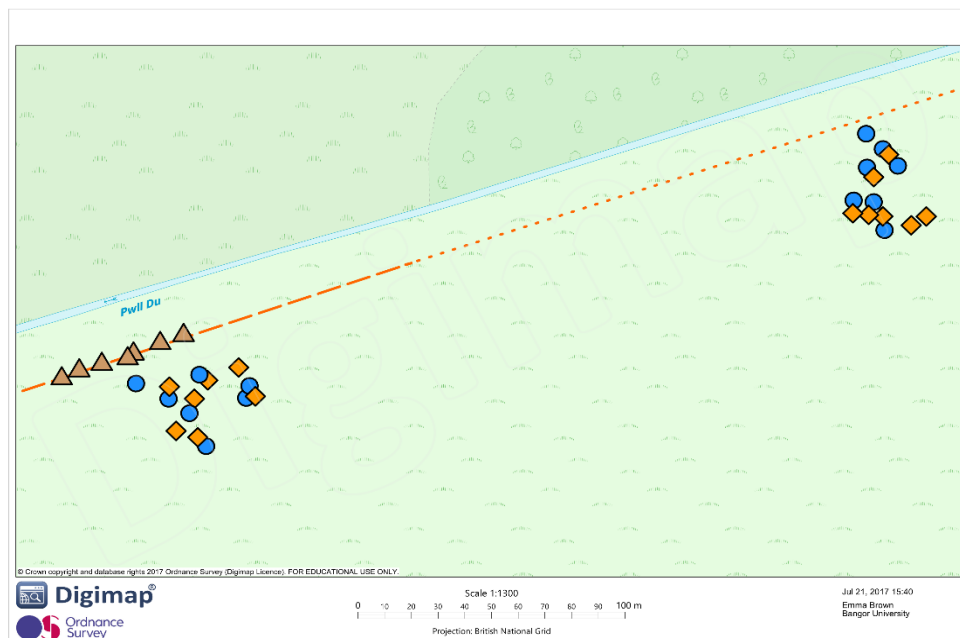
representing RW-16. Collar locations for the bund are represented by triangles, ridges by squares and troughs by circles.

Table 3.1. Description of treatments: microform and mean soil properties (electrical conductivity (EC), soil moisture and organic matter content) and mean growing season (June to September) WTD with ± 1 standard error of the mean shown in brackets.

Treatment	Microform (no. collars)	pH	EC ($\mu S\ cm^{-1}$)	Soil moisture (%)	Organic matter (%)	Mean growing season WTD (mm) 2015	2016
Rewetted-2015	Bund (7)	4.91 (0.14)	42.2 (4.74)	90.40 (1.26)	88.04 (4.98)	-71	-31
	Ridge (7)	4.55 (0.01)	47.9 (4.03)	87.84 (0.99)	91.10 (1.74)	-62	7
	Trough (7)	4.67 (0.01)	38.9 (1.35)	92.42 (0.62)	91.42 (5.01)	-31	63
Rewetted-2016	Ridge (7)	4.24 (0.08)	51.1 (1.96)	83.46 (0.94)	96.90 (0.24)	-197	14
	Trough (7)	4.48 (0.05)	41.3 (2.71)	92.06 (0.36)	96.92 (0.36)	-140	99

3.2.2. Gas flux measurements

CO_2 and CH_4 fluxes were measured approximately every three weeks from July 2015 to October 2016, with 19 measurement campaigns in total using a closed chamber 21 cm diameter and 50 cm tall. Prior to the measurements commencing seven 21 cm diameter collars were installed into the peat of each microform to a depth of 5 cm (35 collars across both restoration treatments and five microforms in total as shown in Fig. 3.1b). Both CO_2 and CH_4 were measured using a Los Gatos Ultraportable GHG Analyser (Los Gatos Research Inc., USA) and Perspex chambers equipped with a pressure equalisation balloon and a fan to ensure thorough mixing of the chamber headspace (Appendix B, Fig. B1). Temperature and relative humidity of the chamber headspace were logged every 10 seconds using a Hobo Data logger (Onset Computer Corporation, USA). Soil temperature at a depth of 6 cm, air pressure and

photosynthetically active radiation (PAR) at the top of the chamber were measured immediately following the placement of the chamber on to a collar. WTD was ascertained during each campaign using a dipwell installed within each microform.

Each chamber placement lasted for approximately four minutes, net ecosystem exchange (NEE) was measured using a transparent chamber and ecosystem respiration (R_{ECO}) using a chamber covered in with an opaque shroud. All chamber measurements were conducted between 10 am and 5 pm.

3.2.3. Flux calculation

CO_2 and CH_4 fluxes were calculated based on the linear change in headspace gas concentration over time using Eq. 3.1, adapted from Denmead, 2008.

$$F_{GHG} = \frac{\partial c}{\partial t} \cdot \frac{V}{A} \qquad [3.1]$$

where F_{GHG} is the calculated flux in gGHG m^{-2} hr^{-1}, ($\partial c / \partial t$) is the change in gas concentration over time, V the volume of the chamber in m^3, and A the area of the chamber in m^2. Fluxes were accepted if the r^2 of the slope ($\partial c / \partial t$) was greater than 0.7 and p value was ≤ 0.05, however small but non-zero fluxes (i.e. p ≤ 0.05) were retained if the r^2 was less than 0.7 to avoid a bias towards large fluxes (following the approach used in Ch. 2; section 2.2.4). Negative fluxes denote the uptake of gas by the ecosystem and positive fluxes a loss to the atmosphere. Gross primary productivity (GPP) was calculated as the difference between NEE and R_{ECO}.

3.2.4. Statistical analysis

All statistical analysis was performed using the R statistical package (R Core Team, 2016). Correlations between CO_2 or CH_4 fluxes and environmental variables (e.g. soil temperature, WTD and PAR) were analysed using the Kendall Tau statistic. The effect of microtopography and rewetting on CO_2 and CH_4 fluxes were investigated using a linear mixed effect model using the package 'nlme' (Pinheiro et al., 2016). CO_2 and CH_4 fluxes were considered response variables, with environmental variables including soil temperature, site, microform and rewetting status as fixed effects and measurement location (collar ID) as a random effect to take into consideration repeated measures. To meet the assumptions of this analysis, i.e. homoscedasticity and normality of residuals, the natural logarithm of CH_4 flux was taken as the response variable. The CO_2 flux data did not meet the requirements of equal variance or

normality of residuals, even after transformation, thus a non-parametric repeated measured Friedman test was used to test for differences between measured R_{ECO} and GPP fluxes taken at different microforms and restoration treatments.

3.3. Results

3.3.1. Environmental conditions

The mean annual air temperature for 2015 and 2016 was 10.3 °C and 10.5 °C respectively, in line with the long term average. The soil temperature observed over the study period demonstrated a clear seasonal pattern, with a minimum and maximum temperature of 3.1 and 23.6 °C respectively (Fig. 3.2a). The mean soil temperature measured between June and September was higher in 2016 compared to 2015 (17.5 °C and 16.6 °C respectively). Soil temperature was similar within treatments (Fig. 3.2a), showing no significant difference between microform (Appendix B, Table B1). Soil temperatures within the RW-15 treatment were significantly higher than those within the RW-16 treatment ($\chi 2 = 4.27$, $p < 0.001$), with a mean measured soil temperature of 14.3 and 13.0 °C respectively (Fig. 3.2a).

The soil pH was consistent across the study area, ranging from 4.24 to 4.91 (Table 3.1). Electrical conductivity (EC) differed between microforms; EC in the troughs and bund ranged between 38.9 ± 1.4 and 42.4 ± 4.7 µS cm^{-1} whilst ridge EC was higher, ranging between 47.9 ± 4.0 and 51.1 ± 2.0 µS cm^{-1}. The organic matter (OM) content of the soils varied by treatment area rather than by microform with the bund having the lowest OM content (88 ± 5%), followed by the ridges and troughs in RW-15 (~ 91 ± 2.4 %) and then the RW-16 microforms (~ 97 ± 0.2 %).

The lowest water tables were measured during 2015, when a summer downdraw of up to -31 cm was observed (Fig. 3.2b), compared to only -7 cm in the summer of 2016. Water tables rose towards the surface during the autumn in all microforms, with many of the troughs becoming inundated over the winter (Table B2). The water levels remained high in 2016, with the majority of the troughs in both treatments inundated throughout the remainder of the study. Inundation was less common in the ridge microforms in 2016 compared to the troughs, only occurring in four collars, though water levels remained within 5 cm of the surface. The water level at the bund remained below the surface throughout the entire study period (Fig. 3.2b).
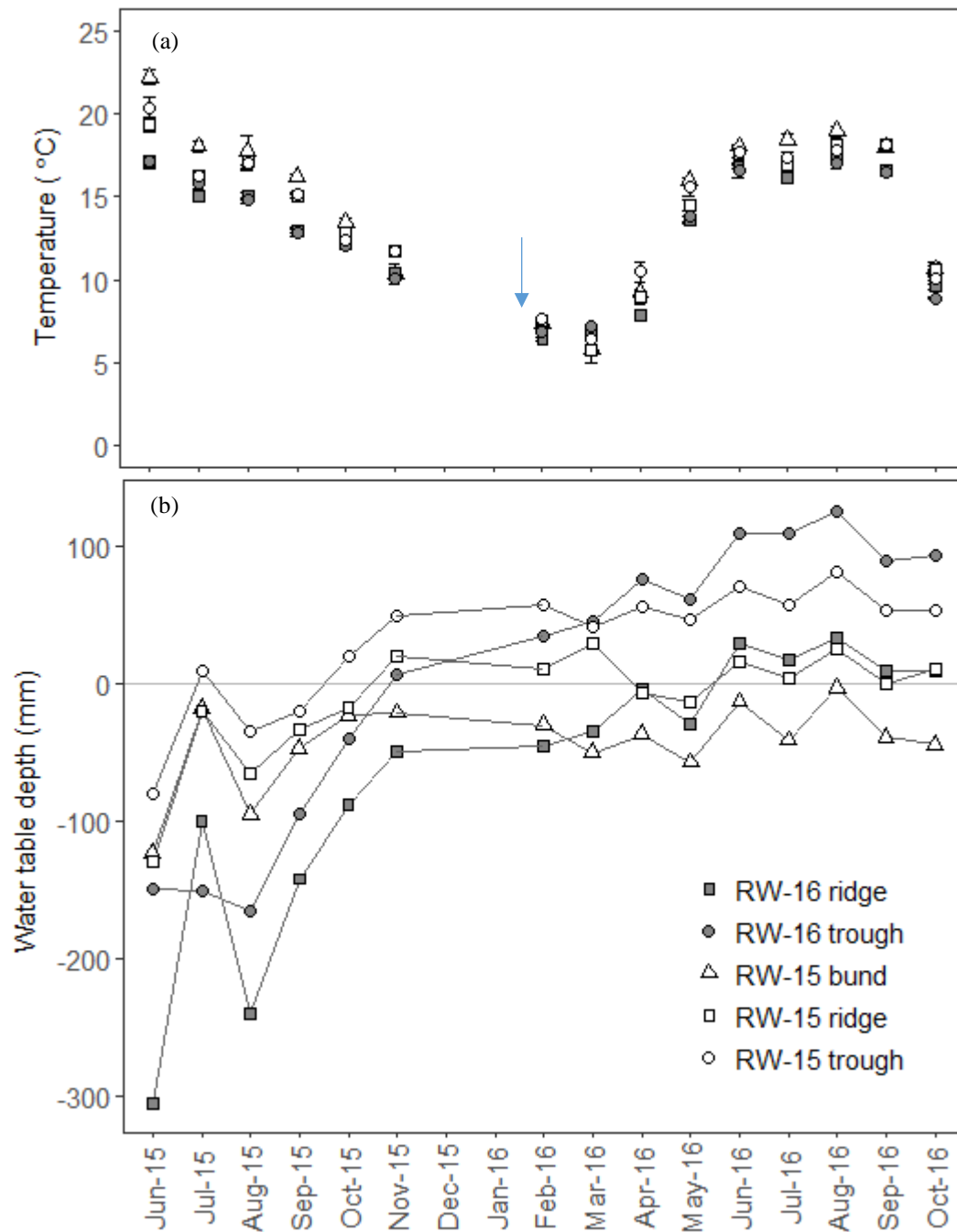
Figure 3.2. (a) Mean measured soil temperature and (b) WTD for each microform (bund, ridge and trough) within the RW-15 and RW-16 treatments. The arrow indicates the timing of bund extension to rewet RW-16 (February 2016).

3.3.2. Carbon dioxide

Measured daytime NEE fluxes were negative between June and September suggesting an instantaneous net uptake of CO_2 across all microforms (Fig. 3.3a,b), with the exception of June

2015 and the RW-16 trough microforms during 2016, when net emissions of CO_2 were measured. The CO_2 emissions measured in June 2015 were driven by high R_{ECO} fluxes, and coincided with the highest soil temperatures and lowest water table levels measured. From November 2015 to March 2016 the ridges and troughs were near neutral with regards to CO_2 balance, with fluxes ranging from -0.05 gC m^{-2} hr^{-1} to 0.06 gC m^{-2} hr^{-1}. A similar winter range in NEE fluxes was measured from the bund, with the exception of collars dominated by vegetation (*Juncus* spp.) from which a net uptake of CO_2 was measured throughout the study period (Fig. 3.4).

3.3.2.1. CO_2 fluxes and microforms

CO_2 uptake was consistently greater from the ridge microforms compared to the troughs in both treatments (Fig. 3.3c,d), though differences in NEE between the ridges and troughs were negligible in 2015 and spring 2016 (Fig. 3.3a,b). The ridges rewetted during RW-16 acted as significantly greater net CO_2 sinks compared to the troughs in 2016 (p = 0.037, Table 3.2a); NEE measured between May and September 2016 ranged from -331 to 0.74 mg C m^{-2} hr^{-1} in the ridges and -120 to 80.5 mg C m^{-2} hr^{-1} in the troughs. R_{ECO} fluxes were similar between the two microforms (p = 0.976), however productivity was significantly greater in the ridges (p=0.037), where the maximum uptake measured was -410 mg C m^{-2} hr^{-1}; over double the maximum measured in the troughs (-180 mg C m^{-2} hr^{-1}). The ridges remained persistent net CO_2 sinks following rewetting in RW-16, however a net emission of CO_2 was observed in 15 out of 38 RW-16 trough collar measurements from May to October 2016. Within the RW-15 treatment mean R_{ECO} fluxes measured between June and September were twice as high in the ridges (99.8 mg C m^{-2} hr^{-1}) compared to the troughs (49.1 mg C m^{-2} hr^{-1}) (Fig. 3.3e), though this was not statistically significant (p = 0.109). CO_2 uptake was consistently greater, though not significantly, from the ridges compared to the troughs within the RW-15 treatment (p = 0.109, Fig. 3.3c), with no statistically significant difference in NEE between microforms (p = 0.637, Fig. 3.3a). The bund was a significantly greater net CO_2 sink compared to the RW-16 ridges (p = 0.002), RW-16 troughs (p < 0.001) and the RW-15 troughs (p = 0.002), though no significant difference was found between CO_2 fluxes from the bund and the RW-15 ridges (p = 0.142).
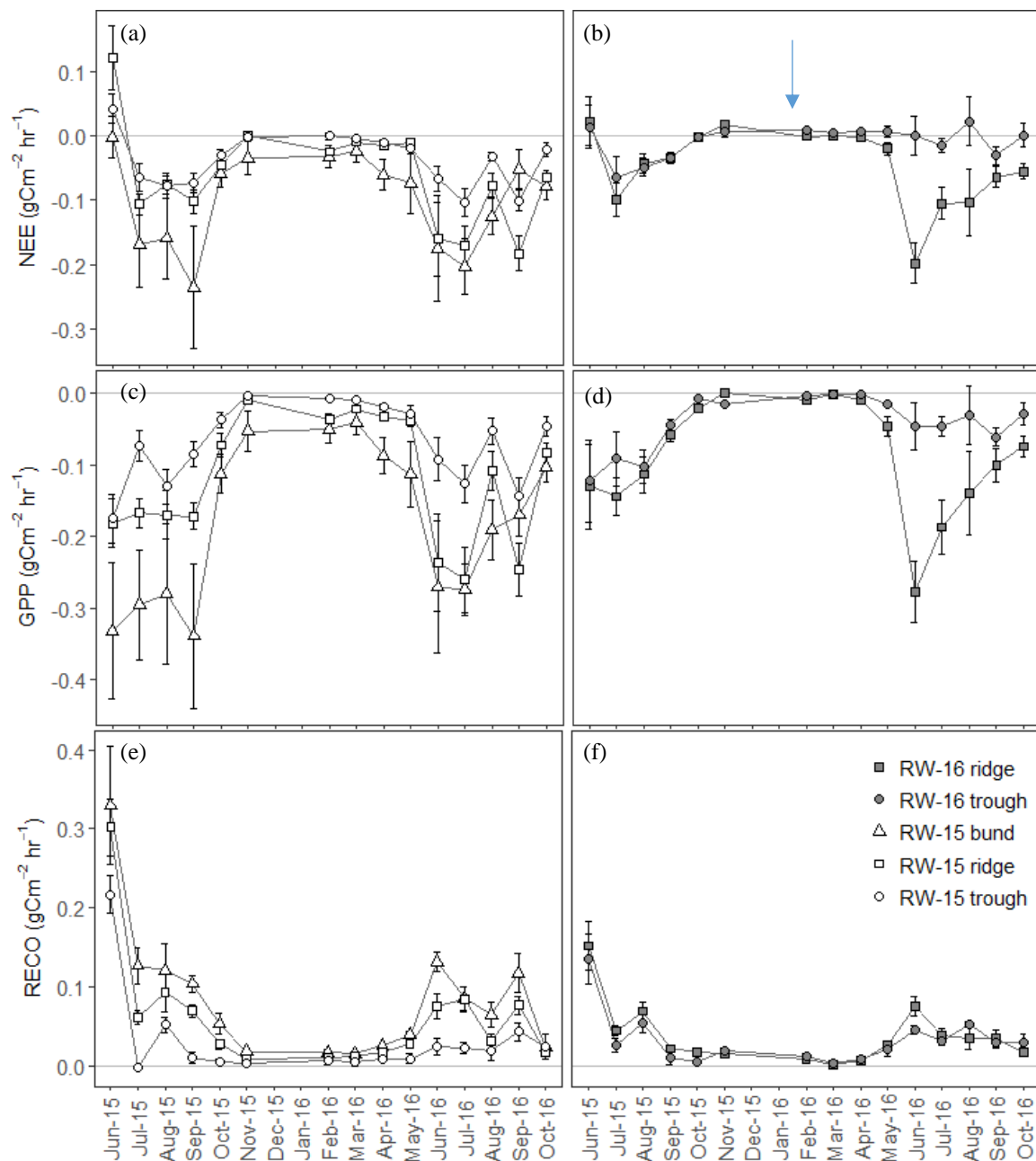
Figure 3.3. Average monthly measured CO_2 fluxes; net ecosystem exchange (a, b); gross primary productivity (c, d) and ecosystem respiration (e, f) for the RW-15 treatment (a, c, e) and the RW-16 treatment (b, d, f). Error bars represent ± 1 SE around the mean. The arrow indicates the timing of bund extension to rewet RW-16 (February 2016).

3.3.2.2. CO_2 and rewetting

Establishment of the bund in two stages provided an opportunity to compare CO_2 fluxes from rewetted (RW-15) and non-rewetted (RW-16) microforms using measurements taken between June and November 2015. CO_2 uptake in the rewetted ridges was consistently greater compared to the non-rewetted ridges (Fig. 3.4b); mean summer GPP fluxes were -173 and -133 mg C m^{-2} hr^{-1} from the rewetted and non-rewetted ridges respectively. The difference between rewetted and non-rewetted troughs was minimal with GPP fluxes ranging from -14.5 to -295 mg C m^{-2} hr^{-1} and -13.6 to -433 mg C m^{-2} hr^{-1} in the rewetted and non-rewetted treatment respectively. R_{ECO} and NEE fluxes were similar from rewetted and non-rewetted microforms (Fig. 3.4a,c), and no statistically significant treatment effect was observed on CO_2 fluxes from corresponding microforms in 2015 (Table 3.2a). Following the second rewetting treatment in February 2016, CO_2 fluxes from the ridges were similar between treatments (measurements right of the arrow in Fig. 3.4) with NEE fluxes ranging between -388 to 7.1 mg C m^{-2} hr^{-1} from the RW-15 ridges and -331 to 9.3 mg C m^{-2} hr^{-1} from the RW-16 ridges. In contrast, the CO_2 fluxes from the troughs differed between the two treatments, though not statistically significant (NEE p = 0.23, R_{ECO} p = 0.98, GPP p = 0.38), with some troughs within the RW-16 treatment acting as net CO_2 sources resulting in a mean CO_2 balance of -11 mg C m^{-2} hr^{-1} between June and September, whilst the RW-15 troughs remained net CO_2 sinks, with a mean NEE of -84 mg C m^{-2} hr^{-1} during this period.

Comparison of measured CO_2 fluxes from collars sampled during the same months (June to October) before (2015 i.e. non-rewetted) and after (2016 i.e. rewetted) the second rewetting treatment showed the ridges to be a greater net CO_2 sink following rewetting; with a maximum net CO_2 uptake of -331 mg C m^{-2} hr^{-1} after rewetting compared to -218 mg C m^{-2} hr^{-1} before rewetting (Fig. 3.5), though this was not statistically significant (p = 0.204). The troughs showed the opposite response to the ridges and acted as smaller net CO_2 sinks following rewetting (NEE range of -120 to 80.5 mg C m^{-2} hr^{-1}) compared to before (NEE range -218 to 85.4 mg C m^{-2} hr^{-1}) though again this was not statistically significant (p = 0.761). Similar findings were observed for the GPP and R_{ECO} fluxes, with no statistically significant effect of rewetting on either ridge or trough microforms (Table 3.2b, Fig. 3.5).
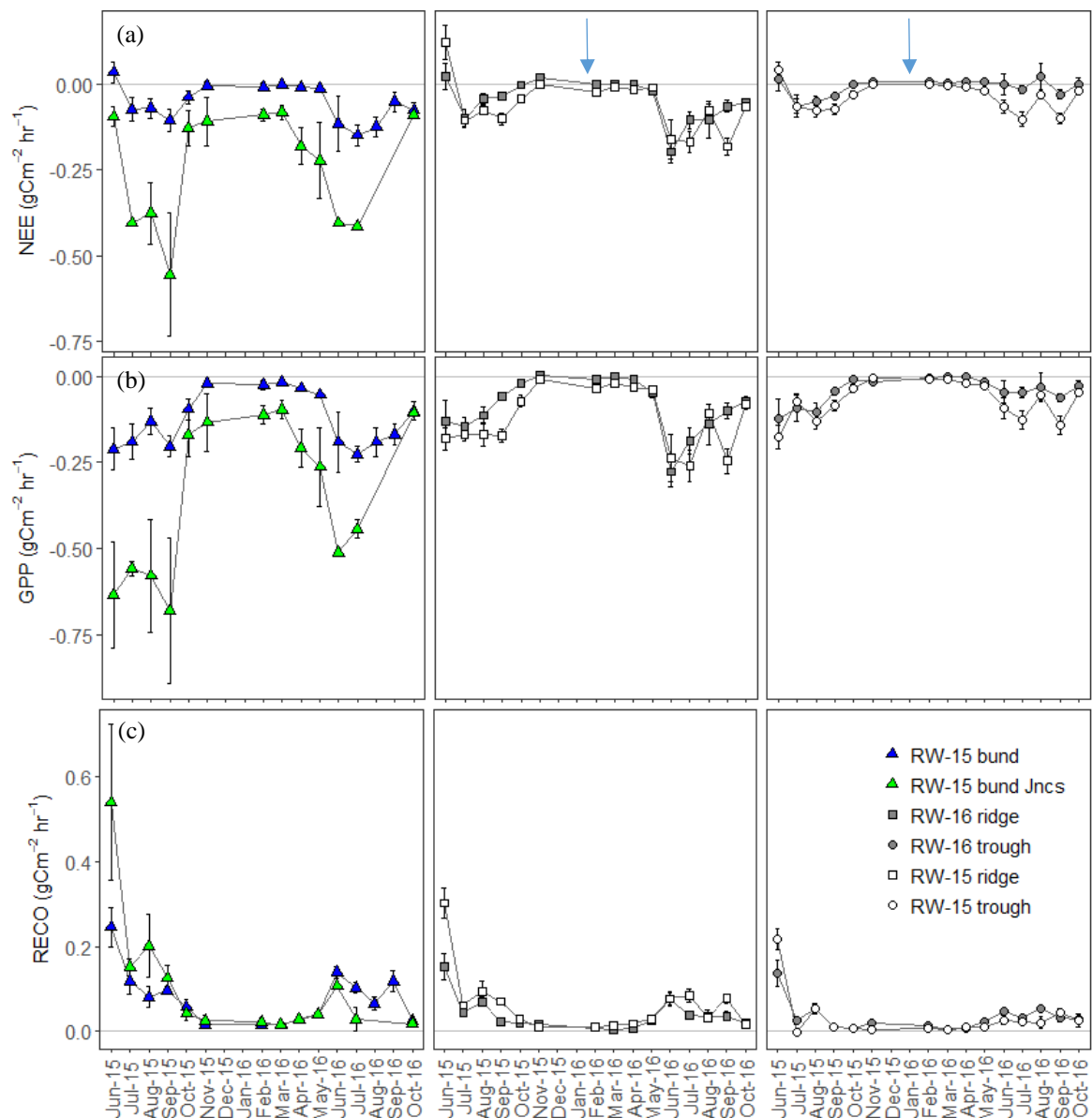
Figure. 3.4. Average monthly measured CO₂ fluxes; (a) net ecosystem exchange; (b) gross primary productivity and (c) ecosystem respiration for the bund (collars with and without Juncus spp.), ridges and trough. Error bars represent ± 1 SE around the mean. The arrow indicates the timing of bund extension to rewet RW-16 (February 2016).

Table 3.2a. Friedman Test summary statistics for comparison of CO_2 fluxes. Significant p-values are shown in bold (p < 0.05).

	NEE 2015	R_{ECO} 2015	GPP 2015	NEE 2016	R_{ECO} 2016	GPP 2016	NEE	R_{ECO}	GPP
Test-score χ^2	3.29	3.65	3.65	4.92	3.43	5.07	5.89	4.97	6.24
P value	**0.009**	**0.002**	**0.002**	**< 0.001**	**0.006**	**< 0.001**	**< 0.001**	**< 0.001**	**< 0.001**
Post-hoc test results (p values)									
RW-16_ridge – bund	**0.009**	0.183	**0.009**	0.226	**0.006**	0.166	**0.002**	**0.001**	**0.001**
RW-16_trough – bund	**0.009**	**0.029**	**0.002**	**< 0.001**	**0.037**	**< 0.001**	**< 0.001**	**0.001**	**< 0.001**
RW-15_ridge – bund	0.262	0.705	0.705	0.665	0.166	0.665	0.142	0.082	0.346
RW-15_trough - bund	0.123	**0.002**	**0.048**	**0.037**	**0.006**	**0.009**	**0.002**	**< 0.001**	**< 0.001**
RW-16_trough – RW-16_ridge	1	0.95	0.996	**0.037**	0.976	**0.037**	0.182	1	0.109
RW-15_ridge – RW-16_ridge	0.705	0.892	0.262	0.946	0.756	0.899	0.637	0.562	0.284
RW-15_trough – RW-16_ridge	0.892	0.588	0.982	0.946	1	0.835	1	0.888	0.991
RW-15_ridge – RW-16_trough	0.705	0.47	0.123	**0.003**	0.976	**0.002**	**0.003**	0.562	**< 0.001**
RW-15_trough – RW-16_trough	0.892	0.95	0.892	0.226	0.976	0.38	0.182	0.888	0.284
RW-15_trough – RW-15_ridge	0.996	0.122	0.588	0.569	0.756	0.297	0.637	0.109	0.109

Table 3.2b. Friedman Test summary statistic for comparison of CO_2 fluxes from collars measured before and after RW-16 (i.e. non-rewetted (2015) and rewetted (2016)). Significant p-values are shown in bold (p < 0.05).

	NEE	R_{ECO}	GPP
χ^2	3.18	1.96	2.94
P value	**0.008**	0.203	**0.017**
Post-hoc test results (p values)			
ridge 2016 - ridge 2015	0.204		0.316
trough 2015 - ridge 2015	0.995		0.611
trough 2016 - ridge 2015	0.611		0.761
trough 2015 - ridge 2016	0.122		**0.017**
trough 2016 - ridge 2016	**0.008**		**0.035**
trough 2016 - trough 2015	0.761		0.995

Figure 3.5. Measured CO₂ fluxes (a) GPP; (b) R_ECO; (c) NEE from the same collar and month before rewetting (2015) and after rewetting (2016) for each microform. Months included are June – October. The red line represents the 1:1 line.

3.3.2.3. Controls on CO₂ fluxes

Ecosystem respiration was significantly correlated with soil temperature across all microforms, with an increase in R_{ECO} observed as soil temperature increased (Table 3.3, Fig. 3.6a). Similarly a significant increase in productivity (i.e. more negative GPP) was found with an increase in soil temperature. The relationship between water table and R_{ECO} varied between the two years of the study, with a significant decrease in R_{ECO} observed in all microforms as the water table rose towards the surface in 2015 (Table 3.3, Fig. 3.6b). The opposite response was observed in 2016, with an increase in R_{ECO} observed as the water table increased and rose above the surface, a correlation which was statistically significant in the bund and RW-16 microforms (Table 3.3). The correlation between GPP and water table showed a significant increase in productivity (i.e. more negative GPP) with a lowering of the water table in all microforms in 2015. However there was no significant correlation between GPP and water table observed in 2016, except in the RW-16 ridges were a lowering of the water table was associated with a decrease in productivity (i.e. less negative GPP). The correlation between GPP and PAR was significant in all microforms in 2015 only, with productivity increasing as PAR increased (Table 3.3, Fig. 3.6e).

Table 3.3. Kendall Tau Correlation between CO₂ fluxes (GPP and R_{ECO}) and environmental parameters (soil temperature, WTD and PAR). Statistically non-significant correlations (p ≥ 0.05) are denoted by ^{NS}.

Treatment	Soil Temperature		WTD		PAR	
	2015	2016	2015	2016	2015	2016
Gross Primary Productivity						
RW-15 Bund	-0.397	-0.36	0.291	-0.071^{NS}	-0.384	-0.075^{NS}
RW-15 ridge	-0.446	-0.62	0.499	0.058^{NS}	-0.458	0.078^{NS}
RW-15 trough	-0.589	-0.54	0.656	-0.005^{NS}	-0.635	-0.069^{NS}
RW-16 ridge	-0.561	-0.64	0.549	-0.298	-0.607	-0.081^{NS}
RW-16 trough	-0.382	-0.38	0.404	-0.032^{NS}	-0.327	0.047^{NS}
Ecosystem Respiration						
RW-15 Bund	0.652	0.541	-0.453	0.18		
RW-15 ridge	0.626	0.559	-0.701	-0.145^{NS}		
RW-15 trough	0.455	0.409	-0.55	0.002^{NS}		
RW-16 ridge	0.514	0.461	-0.531	0.193		
RW-16 trough	0.336	0.385	-0.281	0.328		
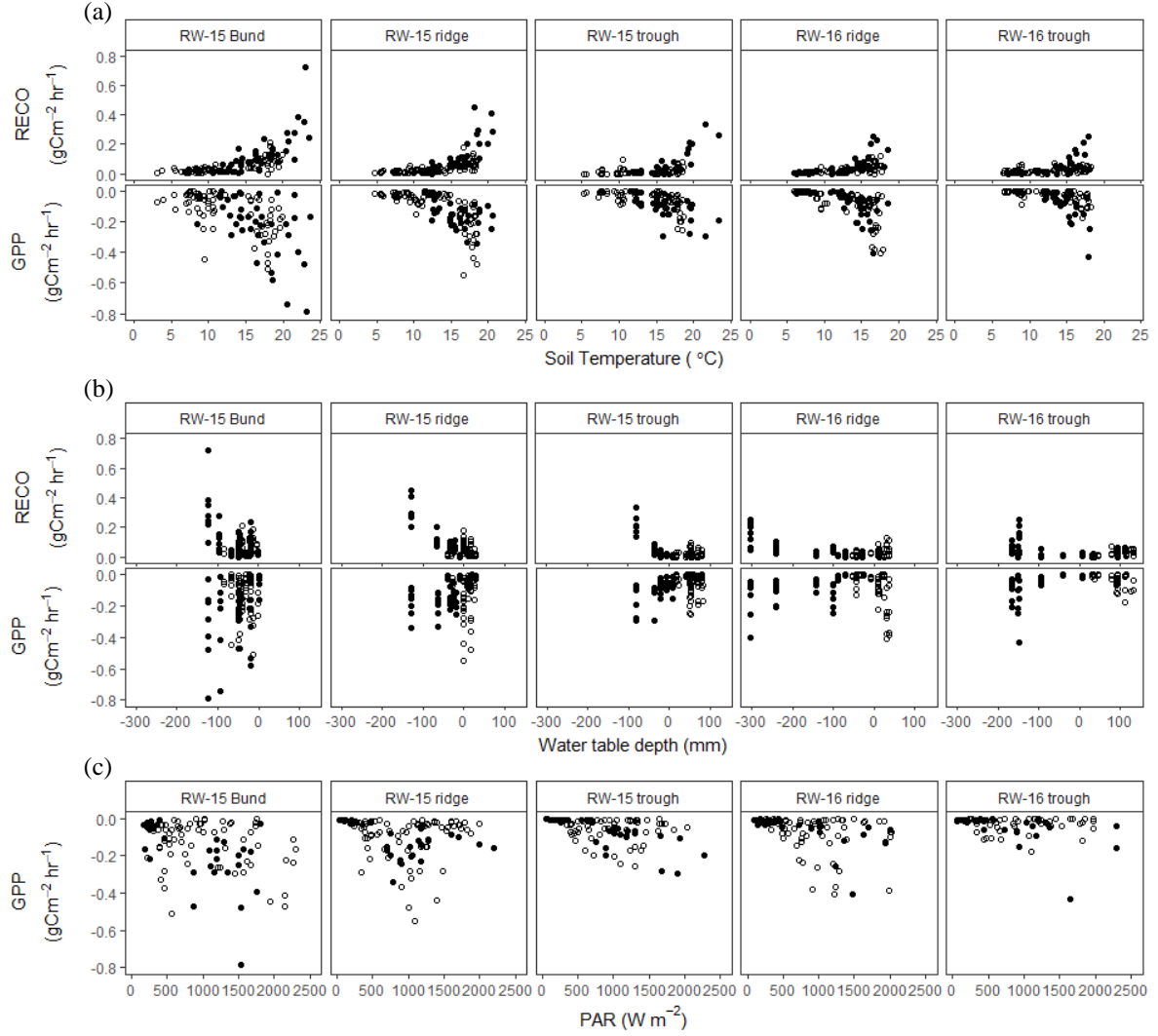
Figure 3.6. Relationship between R$_{ECO}$ or GPP flux and environmental variables for each microform; (a) soil temperature; (b) water table; (c) PAR. Filled circles represent measurements collected in 2015, open circles 2016.

3.3.3. CH4 fluxes

CH4 fluxes showed a seasonal pattern throughout the study period, with the highest CH4 fluxes occurring between June and September and the lowest fluxes during the winter (Fig. 3.7). All microforms acted as net CH4 sources throughout the study period with high variability observed, particularly within the RW-15 treatment.
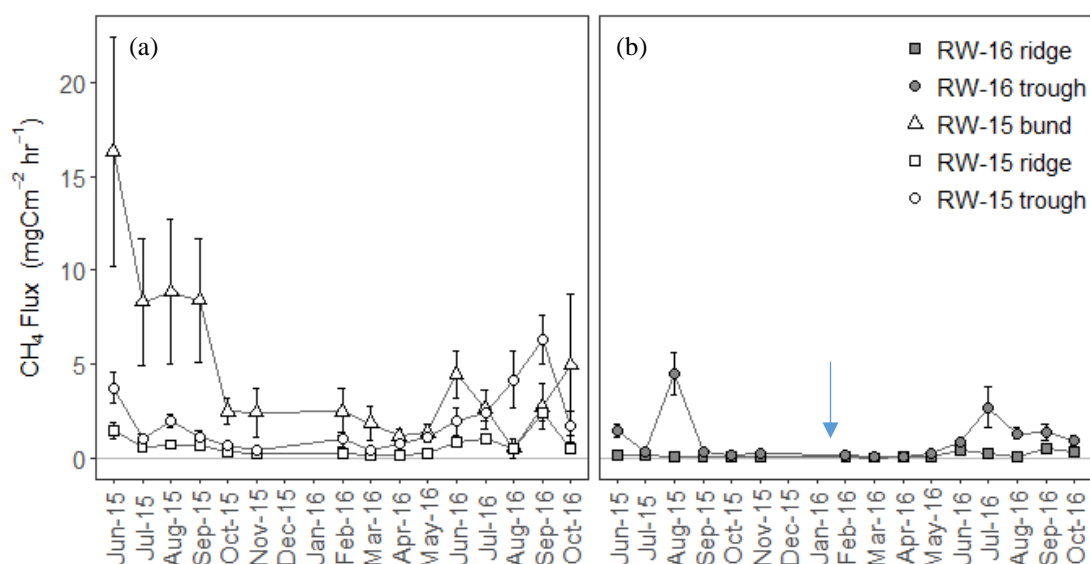
Figure 3.7. Mean monthly measured CH4 fluxes for each microform in the (a) RW-15 treatment and (b) RW-16 treatment. Error bars represent ± 1 SE around the mean. The arrow indicates the timing of bund extension to rewet RW-16 (February 2016).

3.3.3.1. CH4 fluxes and microforms

The results of the mixed effects analysis investigating the effect of microform on CH4 fluxes is shown in Table 3.4. Soil temperature, microform, treatment and year were found to be significant fixed effects, with significant interactions found between soil temperature and treatment as well as between microform and year. An increase in CH4 fluxes was found as soil temperature increased and the highest fluxes were measured from the bund, followed by the trough and then the ridges. The interaction between microform and year indicates that the difference in CH4 fluxes between the bund and ridges, as well as the bund and troughs, was greater in 2016 compared to 2015 (Fig. B3b). The interaction between soil temperature and treatment indicate that the increase in CH4 fluxes with soil temperature was greater in the RW-16 treatment compared to the RW-15 treatment (Fig. B3a), although CH4 fluxes were higher within the RW-15 treatment.

Table 3.4. Mixed effect model summary statistics for comparing microforms (all data). log(CH₄) *is the response variable, Tsoil (6 cm soil temperature), Treatment (RW-15 or RW-16), MicroT* *(microform) and Year (2015 or 2016) are the fixed effects and collar is a random effect. *denotes* *an interaction between variables. Significant p-values are shown in bold (p < 0.05).*

Equation:	$\log(CH_4) \sim T_{soil} * Treatment + \ MicroT * Year + (1|Collar)$				
	Value	Std.Error	DF	t-value	p-value
(Intercept)	-4.291	0.501	977	-8.56	**< 0.001**
Tsoil	0.255	0.015	977	17.42	**< 0.001**
Treatment(RW-15)	2.472	0.397	31	6.23	**< 0.001**
MicroT(ridge)	-1.496	0.441	31	-3.39	**0.002**
MicroT(trough)	-0.364	0.443	31	-0.82	0.418
Year(2016)	-0.372	0.161	977	-2.31	**0.021**
Tsoil * Treatment(RW-15)	-0.087	0.018	977	-4.92	**< 0.001**
MicroT(ridge) * Year(2016)	0.630	0.191	977	3.30	**0.001**
MicroT(trough) * Year(2016)	0.763	0.198	977	3.84	**< 0.001**

3.3.3.2. CH₄ fluxes and rewetting

The mixed effect analysis using only 2015 data (i.e. comparing rewetted and non-rewetted microforms) found treatment to be a significant fixed effect, with CH₄ fluxes significantly higher from rewetted microforms compared to the non-rewetted microforms (Table 3.5, Fig. 3.8). Average CH₄ emissions during 2015 from the rewetted microforms were 0.59 and 1.43 mg C m^{-2} hr^{-1} for the ridges and troughs respectively, whereas fluxes from the non-rewetted microforms were 0.08 and 1.02 mgC m^{-2} hr^{-1} for the ridges and troughs respectively. The significant interaction between treatment and water table indicates that as the water table increased towards the surface, there was a smaller treatment effect, i.e. a smaller difference in CH₄ fluxes between RW-15 and RW-16 treatments (Fig. B4a). Similarly the significant interaction between microform and water table suggests that a smaller difference was observed in CH₄ fluxes from the ridges and troughs as the water table rose (Fig. B4b).

*Table 3.5. Mixed effect model summary statistics for comparing rewetted and non-rewetted treatments (2015 data only). log(CH₄) is the response variable, Tsoil (6 cm soil temperature, Treatment (RW-15 or RW-16), MicroT (microform) and WT (water table) are the fixed effects and collar is a random effect. * denotes an interaction between the two variables.*

Equation:	$\log(CH_4) \sim T_{soil} + Treatment * MicroT * WT + (1|Collar)$				
	Value	Std.Error	DF	t-value	p-value
(Intercept)	-3.434	0.471	282	-7.29	**< 0.001**
Tsoil	0.044	0.034	282	1.28	0.203
Treatment(RW-15)	1.823	0.399	24	4.57	**< 0.001**
MicroT(trough)	0.537	0.417	24	1.29	0.210
WT	-0.002	0.001	282	-1.69	0.092
Treatment(RW-15) * MicroT(trough)	0.371	0.557	24	0.67	0.512
Treatment(RW-15) * WT	-0.008	0.002	282	-3.52	**< 0.001**
MicroT(trough) * WT	-0.012	0.002	282	-6.41	**< 0.001**
Treatment(RW-15) * MicroT(trough) * WT	0.008	0.003	282	2.43	**0.016**
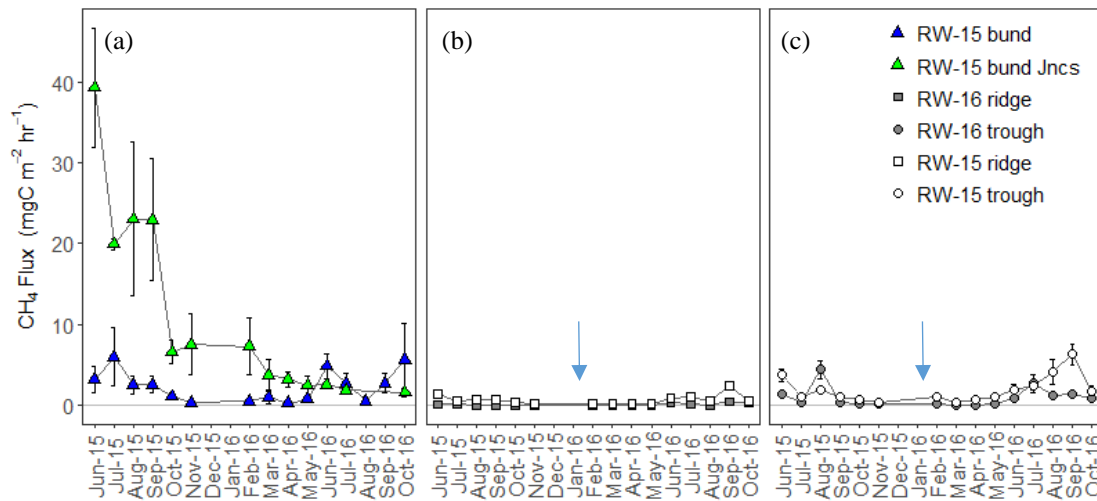
Figure 3.8. Mean monthly measured CH₄ fluxes for the bund (collars with and without Juncus spp.), ridges and troughs. Error bars represent ± 1 SE around the mean. The arrow indicates the timing of bund extension to rewet RW-16 (February 2016).

The mixed effect analysis using only data from the RW-16 treatment (i.e. comparing CH_4 fluxes before and after rewetting) revealed rewetting status to be a significant fixed effect, with CH_4 fluxes significantly greater after rewetting (Table 3.6). A significant increase in CH_4 fluxes was also found with an increase in soil temperature, with a significant interaction between soil temperature and microform indicating that the increase in CH_4 fluxes with soil temperature was greater in the troughs compared to the ridge microforms (Fig. B5). Fig. 3.9 compares measured CH_4 fluxes from collars sampled during the same months (June to October) in both 2015 and 2016. CH_4 fluxes measured from the ridges were a consistently higher following rewetting but fluxes were very low fluxes in both years (<1.3 mg C m^{-2} hr^{-1}). The range in CH_4 fluxes from the troughs was similar in both years; ranging from -0.08 to 12.8 mg C m^{-2} hr^{-1} before rewetting and -0.7 to 19.5 mg C m^{-2} hr^{-1} following rewetting, however an increase was observed in the mean CH_4 flux (from 0.8 to 1.1 mg C m^{-2} hr^{-1}) with rewetting.

*Table 3.6. Mixed effect model summary statistics for comparing before and after rewetting treatments (RW-16 data only). log(CH₄) is the response variable, Tsoil (6 cm soil temperature), MicroT (microform) and RW (rewetting status; before or after) are the fixed effects and collar is a random effect. * denotes an interaction between the two variables.*

Equation:	$\log(CH_4) \sim T_{soil} * MicroT + RW + (1|Collar)$				
	Value	Std.Error	DF	t-value	p-value
(Intercept)	-5.060	0.354	380	-14.28	**< 0.001**
Tsoil	0.211	0.021	380	9.98	**< 0.001**
MicroT(trough)	0.300	0.506	12	0.59	0.565
RW(before)	-0.342	0.124	380	-2.77	**0.006**
Tsoil * MicroT(trough)	0.091	0.030	380	3.04	**0.003**

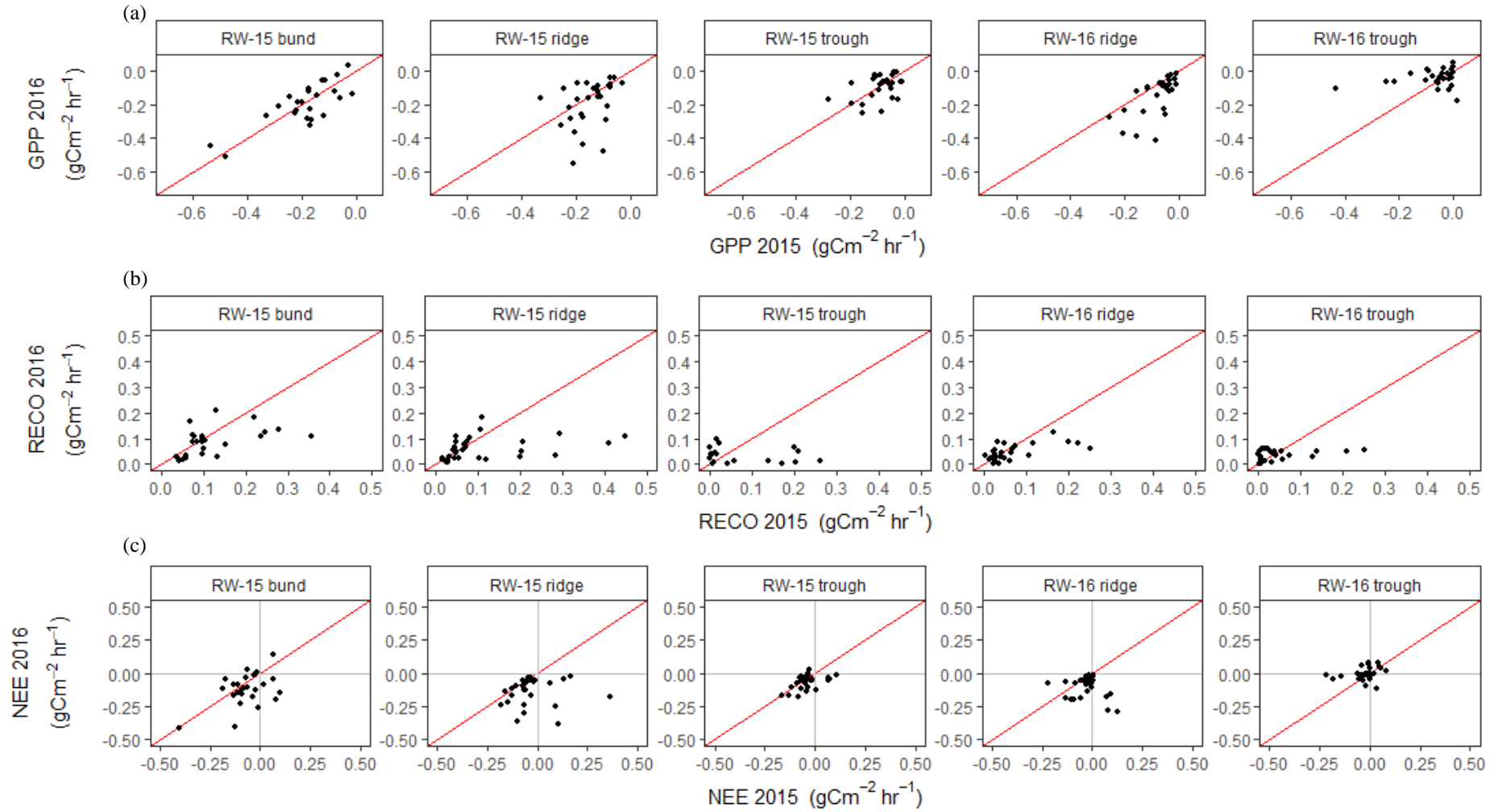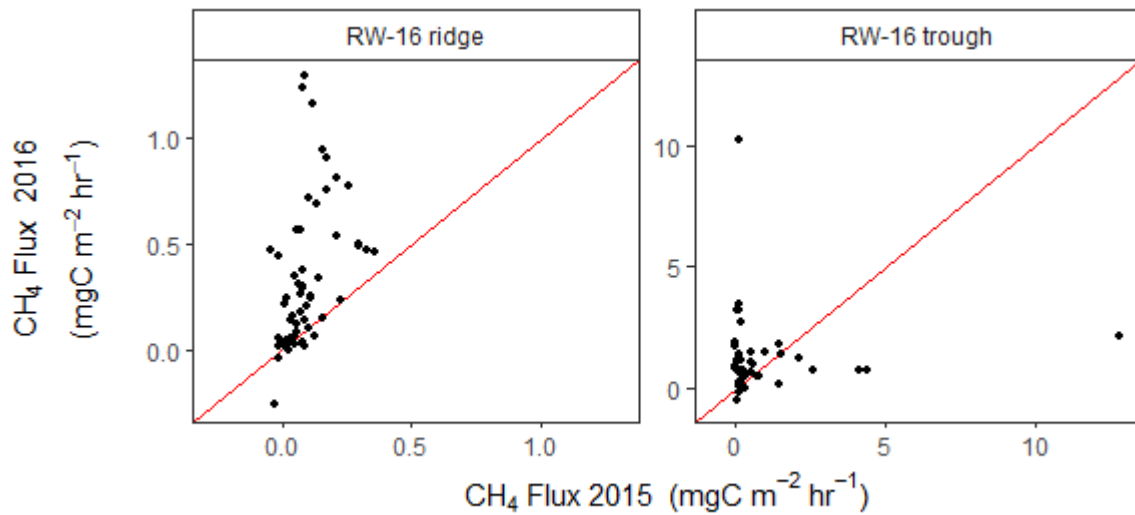*Figure 3.9. Measured CH$_4$ fluxes from the same collar and month before rewetting (2015) and after rewetting (2016) for the ridge and trough microforms within the RW-16 treatment. Months included are June – October. The solid line represents the 1:1 line*

3.3.3.3. Controls on CH$_4$ fluxes

CH$_4$ fluxes were significantly correlated with soil temperature within all microforms, with an increase in CH$_4$ emissions observed with increasing temperatures (Fig. 3.10a). The correlation between soil temperature and CH$_4$ flux was stronger in the microforms within the RW-15 treatment compared to the RW-16, although following rewetting in 2016 (RW-16 treatment) correlations between CH$_4$ flux and soil temperature were similar across the treatments (Table 3.7). As with CO$_2$ fluxes, the correlation between CH$_4$ fluxes and WTD differed between the two years of the study, with CH$_4$ fluxes increasing as the water table lowered in 2015 (significant at all microforms except for RW-16 ridge) (Fig. 3.10b). In 2016 a weak positive correlation was found between WTD and CH$_4$ emissions which was only significant within the RW-16 microforms. CH$_4$ emissions were significantly correlated with GPP and R$_{ECO}$ fluxes throughout both years of the study with greater R$_{ECO}$ and productivity (i.e. more negative GPP) associated with higher CH$_4$ fluxes (Table 3.7).

Table 3.7. Kendall Tau Correlation between CH₄ fluxes and environmental parameters (soil temperature, WTD and PAR) and CO₂ fluxes (GPP, R_{ECO} and NEE). Statistically non-significant correlations (p > 0.05) are denoted by ^{NS}.

Treatment	Soil temperature		WTD		PAR	
	2015	2016	2015	2016	2015	2016
RW-15 bund	0.237	0.124	-0.208	0.112[NS]	0.179 [NS]	-0.178
RW-15 ridge	0.531	0.536	-0.6	-0.029 [NS]	0.477	-0.113 [NS]
RW-15 trough	0.417	0.445	-0.479	0.095 [NS]	0.607	0.056 [NS]
RW-16 ridge	0.171	0.394	-0.139 [NS]	0.179	0.11 4 [NS]	-0.026 [NS]
RW-16 trough	0.278	0.491	-0.389	0.183	0.197 [NS]	-0.005 [NS]

	GPP		R_{ECO}		NEE	
	2015	2016	2015	2016	2015	2016
RW-15 bund	-0.524	-0.428	0.382	0.288	-0.409	-0.341
RW-15 ridge	-0.444	-0.618	0.563	0.548	-0.186	-0.578
RW-15 trough	-0.444	-0.453	0.457	0.549	-0.06 [NS]	-0.314
RW-16 ridge	-0.158	-0.453	0.175	0.229	-0.12 [NS]	-0.427
RW-16 trough	-0.237	-0.291	0.136 [NS]	0.353	-0.131 [NS]	-0.141
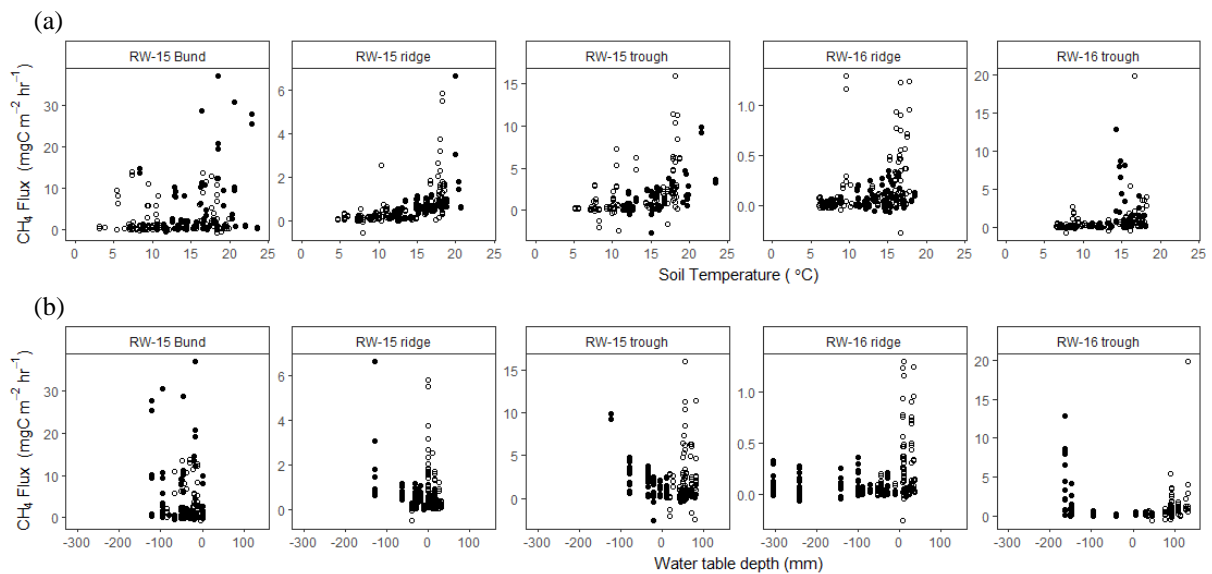
Figure.3.10. Relationship between CH₄ fluxes and (a) soil temperature and; (b) WTD for each microform. Filled circles represent measurements collected in 2015, open circles 2016. Note the different y-axis scales.

3.4. Discussion

3.4.1. The effect of microtopography on GHG fluxes

In natural peatlands, the surface microtopography is generated through different rates of peat accumulation and decomposition, and as a result the surrounding hydrology, vegetation and soil chemistry become altered reinforcing the microsite differences (Waddington and Roulet, 2000; Bubier et al., 2006). All of these factors will in turn influence CO_2 and CH_4 exchange with higher, drier areas generally sequestering more CO_2 than lower, wetter areas, whilst CH_4 emissions are greater from the latter (Lai, 2009). Due to the extraction of peat in strips during hand cutting, a microtopography similar to that of natural peatlands can be formed that consist of ridges and troughs differing in their soil moisture status. Troughs in the current study had higher water tables compared to the ridges, creating conditions that favour the production of CH_4 over ridges where soil moisture was lower which favoured CO_2 sequestration. CH_4 fluxes measured from the troughs were on average five times greater than fluxes from the ridge microforms, confirming the original hypothesis and agreeing with the consensus in the literature. Laine et al., (2007) found CH_4 fluxes to be 70 % higher from hollows (a.k.a. troughs) compared to hummocks (a.k.a. ridges) in an Irish blanket bog, whilst Waddington and Roulet (1996) also found ridges to be significantly lower CH_4 sources in a Swedish raised bog. A mechanistic explanation for the differences is that troughs typically have a higher water table compared to ridges resulting in reduced CH_4 oxidation in the thin overlying aerobic peat layer (Bubier et al., 1995 in Tuittila et al., 2000). Consequently a greater proportion of the CH_4 produced is released to the atmosphere compared to the ridge microforms where oxidation rates are higher. Additionally, soil temperature is generally higher at the water table interface when it is closer to the surface, further promoting CH_4 production rates in troughs, hollows or pools (Lai, 2009).

The establishment of anaerobic conditions is necessary for CH_4 production, following which vegetation type is a key control on CH_4 emissions influencing both the substrate quality and supply to microbial populations as well as potentially facilitating gas transportation. In the work presented here, many of the trough microforms were colonised by bryophytes, however, some of the collars within the RW-15 treatment area, particularly where shallow inundation occurred, contained the aerenchymatous species *E.vaginatum* and *P.australis*. Aerenchymatous tissue has large spaces between cells that can act as a conduit for gas exchange, enabling CH_4 to bypass the aerobic soil layer thus avoiding oxidation by

methanotrophic communities (Strom et al., 2005). Several field studies have shown high CH_4 emissions can be explained by the presence of vegetation known to contain aerenchymatous tissue, for example Green and Baird (2012) found CH_4 fluxes from cores dominated by *E.anugustifolium* to be almost six times greater than fluxes from *Sphagnum* spp. dominated cores. Similarly Tuittila et al. (2000) reported mean fluxes of ~ 1.5 mg C m^{-2} hr^{-1} from tussocks with *E.vaginatum* coverage of > 95 % compared to intertussocks with *E.vaginatum* coverage of < 1 %, where CH_4 fluxes were approximately -0.15 mg C m^{-2} hr^{-1}. CH_4 fluxes reported in the literature for aerenchyma species range from 0 to 12 mg C m^{-2} hr^{-1} (e.g. Beyer and Hoper, 2015; Frenzel and Karofel, 2000; Green and Baird, 2012; Tuittila et al., 2000; Wilson et al., 2013), with CH_4 fluxes derived from aerenhcyma vegetation within the trough microforms of this study within this range (0.1 to 11 mg C m^{-2} hr^{-1}). The influence of vegetation type on CH_4 fluxes was also observed from the bund as mean CH_4 fluxes associated with *Juncus* spp. were five times higher than parts of the bund where *Juncus* spp. were not present (10.3 ± 1.8 and 2.1 ± 0.41 mg C m^{-2} hr^{-1} respectively). Both the mean and range (0.2 to 53 mg C m^{-2} hr^{-1}) of CH_4 fluxes from *Juncus* spp. dominated collars in this study are considerably greater than CH_4 fluxes measured from *Juncus-Sphagnum* microsites in a rewetted cutover bog in Ireland (up to 4 mg CH_4 m^{-2} hr^{-1}; Wilson et al., 2013). The lower CH_4 fluxes reported by Wilson et al. (2013) has been linked to aerenchymatous plants facilitating O_2 transportation from the atmosphere to the roots, increasing the oxygenation of the rhizosphere and thus reducing CH_4 production (Roura-Carol and Freeman, 1999; Fritz et al., 2011). The high CH_4 fluxes from *Juncus* spp. in this study may also be linked to the large input of labile organic matter associated with the establishment of the bund; i.e. the saturated peat brought to the surface during the bund construction.

CO_2 fluxes were expected to be greater from the ridge microforms compared to the troughs, however there was no statistically significant difference in CO_2 fluxes (GPP, R_{ECO} or NEE) from the ridges and troughs within the RW-15 treatment. The variation in water table typically drives the differences observed in CO_2 fluxes across microtopographic gradients (Strack and Waddington, 2007), however in this study there was little difference in WTD between the RW-15 ridges and troughs; a maximum difference of 59 mm was recorded in April 2016. Consequently, it is likely that R_{ECO} and GPP would have been limited (or not limited) to the same degree within both microforms and thus no significant difference was observed. A similar situation was observed within the non-rewetted microforms (RW-16 treatment area) during 2015 with no significant difference in CO_2 fluxes observed, coinciding with a small difference

in WTD. However, following the second rewetting (RW-16), the ridge microforms acted as significantly greater net CO_2 sinks compared to the troughs, which can be attributed to the significantly greater productivity measured within the ridges. The greater productivity within the RW-16 ridges during 2016 coincided with a WTD within 2 cm of the surface, potentially creating optimum conditions for bryophyte productivity. For example, greater productivity under higher water table levels has been observed in hummocks dominated by *Sphagnum fuscum* (Bridgham et al. 2008) and high hummocks in a boreal bog, linked to enhanced bryophyte activity (Pelletier et al. 2011). Differences in productivity between the ridge and trough microforms within the RW-16 treatment may have been further enhanced by the inundation of approximately half of the trough collars following rewetting (51 out of 98 collar measurements from February to October 2016). The inundated areas had little or no vegetation above the surface which would have led to photosynthesis being substantially reduced, and resulted in a net CO_2 emission being observed in 13 out of 32 trough measurements between May and September. The bund acted as a significantly greater CO_2 sink compared to the trough microforms and the RW-16 ridges, with the largest uptake of CO_2 measured from the bund where *Juncus* spp. were present. The consistently lower water table, together with the dominance of vascular plants (particularly collars containing *Juncus* spp.) favours higher productivity and therefore greater CO_2 fluxes. Furthermore, the way the bund was constructed brought saturated peat to the surface to form the bund which is then covered with the removed (degraded) surface vegetation. Although this bund peat layer is not drained, it does not remain fully saturated and therefore a large supply of labile substrate becomes available for decomposition, resulting in high R_{ECO} fluxes.

3.4.2. The effect of rewetting on GHG fluxes

The circumstances of this study led to two opportunities to study the effect of rewetting on the GHG fluxes of a cutover raised bog; the comparison of rewetted (RW-15) and non-rewetted (RW-16) microforms using measurements from June to November 2015 and the comparison of microforms before and after the rewetting as a result of the bund extension (RW-16 treatment). During 2015, CH_4 fluxes from the rewetted microforms were significantly higher than those from the non-rewetted microforms, with treatment a significant fixed effect in the mixed effects analysis. The water table was on average 11 cm closer to the surface in the rewetted microforms compared to the non-rewetted microforms from June to September 2015, agreeing with findings across the literature that higher CH_4 fluxes are associated with higher

water tables due to the thinner aerobic peat layer reducing the opportunity for CH_4 oxidation to occur (Blodau et al., 2004; Lai, 2009). The mixed effects analysis also revealed a decreasing treatment effect as the water table approached the surface (Fig. B4); CH_4 fluxes from the two treatments were similar when water tables were closer the surface suggesting the difference between treatments was driven by the low water tables experienced in the non-rewetted (RW-16) microforms. Additionally, there was a higher coverage of aerenchymatous vegetation found within the rewetted troughs compared to the non-rewetted troughs which may have further contributed to higher CH_4 fluxes from this treatment. Despite CH_4 fluxes from the rewetted microforms being significantly higher than fluxes from the non-rewetted treatment, within each microform, a significant negative correlation was found between CH_4 flux and WTD in 2015, with the exception of the non-rewetted ridges (RW-16) where the correlation was not significant. Although water table controls the amount of CH_4 oxidation occurring, temperature controls the rate of CH_4 production by microbial populations (Segers, 1998). During 2015, water tables ± 2 cm of the surface (October and November) occurred when soil temperatures were at their lowest (mean soil temperature of 11.8 °C), consequently CH_4 production may have been limited compared to the summer when soil temperatures were approximately 6 °C warmer and the water table still within 10 cm of the surface. Similar seasonal patterns in CH_4 fluxes have been observed by Wilson et al. (2013) in a rewetted cutover bog and Laine et al. (2007) in a pristine lowland blanket bog, driven by the changes in soil temperature and leaf area. CH_4 production is also controlled by substrate availability; fresh labile plant litter inputs and root exudates result in greater rates of CH_4 production (Waddington and Day, 2007; Minke et al., 2016). Although GPP did not differ significantly between the rewetted and non-rewetted microforms in 2015, productivity was consistently greater in the former, as well as there being a stronger correlation between CH_4 fluxes and GPP fluxes, suggesting the substrate availability within the RW-15 treatment may have contributed towards higher CH_4 fluxes.

CH_4 fluxes were significantly higher following rewetting within the RW-16 treatment, with the increase in CH_4 fluxes most pronounced within the ridge microforms. Before rewetting the WTD within the ridges was on average 20 cm below the surface between June and September, however following rewetting the mean WTD for the same months was within 2 cm above the surface, with 3 out of 26 measurement collars inundated. The high water table under rewetted conditions would have substantially reduced CH_4 oxidation, with little or no aerobic layer within the peat to support this process. Although CH_4 fluxes from the ridges increased with rewetting, the range of fluxes observed (-0.26 to 1.30 mg C m^{-2} hr^{-1}) was smaller than that

observed in the troughs (up to 19.5 mg C m^{-2} hr^{-1}) and the ridges rewetted for a year longer (RW-15 treatment; -0.53 to 5.8 mg C m^{-2} hr^{-1}). The recovery of methanogenic populations with rewetting following drought events has been shown to increase with drying intensity with higher concentrations of electron acceptors following prolonged drought conditions delaying the onset of CH_4 production (Estop-Aragonés and Blodau, 2012). However, the extent to which methanogenesis is inhibited by alternative electron acceptors is unclear, particularly the time scale of methanogen recovery following rewetting (Knorr and Blodau, 2009). A mesocosm experiment by Estop-Aragones et al. (2016) found it took 200 days for CH_4 fluxes to reach pre-drying levels following a dry period of 100 days where water levels dropped to 21 to 45 cm below the surface. Therefore, it might be expected that CH_4 fluxes from the ridges rewetted within the RW-16 treatment will increase with time as the methanogenic population grows, however due to the long term drainage of the site, the time scale of this recovery is unknown.

It was hypothesised that CO_2 fluxes from rewetted microforms would be lower than those from non-rewetted microforms due to anoxic conditions inhibiting aerobic respiration. However, no statistically significant difference in CO_2 fluxes was found between the rewetted and non-rewetted microforms during 2015 (RW-15 treatment). Whilst a decrease in R_{ECO} with rising water tables has been widely reported (e.g. Pelletier et al., 2011; Strack and Zuback, 2013, Jarveoja et al., 2016; Wilson et al., 2016a), it has also been observed that once the water table is below the surface, R_{ECO} fluxes can remain stable despite further water table downdraw (e.g. Lafleur et al., 2005; Juszczak et al., 2013; Lee et al., 2017). It has been suggested that the variability in water tables deep below the surface has little effect on the soil moisture conditions in the upper peat layers where respiration potential is greatest and therefore no response in R_{ECO} is observed (Lafleur et al., 2005). However, in the current study the water table typically fluctuated within 16 cm of the surface, with water tables below this only measured on two occasions. An explanation for observing no difference in CO_2 fluxes between the rewetted and non-rewetted microforms could be the degree of desiccation of the upper peat layers limiting R_{ECO} in the non-rewetted microforms but not the rewetted microforms. The prolonged drainage of the study site meant that the surface layer within the non-rewetted microforms (especially the ridges) was likely highly desiccated, resulting in a reduction in R_{ECO} within the upper peat layer which was not offset by increased respiration from the aerated deeper peat layer (e.g. Dimitrov et al., 2010). Conversely, in the rewetted microforms, whilst the water table remained below the surface during 2015, the soil moisture conditions at the surface had likely been improved following the construction of the bund. Therefore R_{ECO} fluxes may be higher due to

greater soil moisture conditions and thus no difference between rewetted and non-rewetted microforms was observed. Historical moisture availability has been identified as the dominant factor in CO_2 exchange in a laboratory experiment using samples from a cutover bog by McNeil and Waddington (2003), with rewetting following drying creating a R_{ECO} pulse. No field studies could be found of CO_2 flux measurements made within the first $1-2$ years of rewetting, thus the applicability of laboratory drying and wetting cycles to the field is unknown. However, R_{ECO} fluxes within the RW-15 treatment microforms were lower after one year of rewetting compared to within the first year, suggesting that if a R_{ECO} pulse is observed as desiccated peat becomes rewetted the effect is short lived if a high water table is established. A similar reasoning may explain why no significant difference was observed in R_{ECO} fluxes before and after rewetting within the RW-16 treatment, despite mean summer R_{ECO} fluxes being at least a third smaller following rewetting (45 % lower for ridges, 39 % for troughs). GPP measurements covered a similar range in the ridges before and after rewetting though the mean CO_2 uptake during the summer after rewetting was notably greater (-0.22 g C m^{-2} hr^{-1} after, -0.13 g C m^{-2} hr^{-1} before). This may be linked to enhanced bryophyte activity as the water table was within a few cm of the surface following rewetting but on average 22 cm below the surface during the summer before rewetting.

3.4.3. Implications for peatland restoration

To restore the carbon sink function of cutover peatlands, there are two main factors to be addressed; firstly the restoration of hydrological conditions comparable to those of pristine peatland systems, and secondly the recolonization of peat-forming vegetation (Qunity and Rochefort, 2003; Wilson et al., 2016a). Colonisation of peatlands by peat forming bryophytes (*Sphagnum* spp.) is often hindered by the large water table fluctuations that are found in drained peatlands. To improve hydrological conditions, an increase in water retention is required to reduce the magnitude of wetting and drying cycles (McNeil and Waddington, 2003; Rochefort, 2000; Wheeler and Shaw, 1995). The construction of low-lying bunds and cells have been shown to be successful in creating stable water levels near the soil surface, enabling bryophyte colonisation, which through positive feedback, reduces water table fluctuations, and enhances the growing conditions for peat forming vegetation (Rochefort, 2000; Lunt et al., 2010). Although the study area here had extensive vegetation cover, the presence of drainage ditches resulted in hydrological conditions unfavourable for peat-forming species such as *Sphagnum* spp., hence restoration intervention was still necessary. Blocking drainage ditches at Pant-y-Dwn was not feasible due to nearby agricultural land, however, the shallow slope of the site

and the high levels of precipitation experienced on the west coast of the UK, were favourable for restoration through rewetting with the bund technique. WTD was higher within the RW-15 treatment following the first-stage of bund construction compared to the non-rewetted area, however, below average precipitation at the beginning of 2015 prevented the water table from reaching the surface. During 2016 the bund construction was completed, and as precipitation was also higher, the observed WTD was also more favourable to restoration. A mixture of inundated areas (troughs) and higher, drier areas (ridges), created a mosaic of habitats analogous to that found in natural peatlands. Despite the lack of statistically significant difference in CO_2 fluxes under rewetted and non-rewetted conditions, R_{ECO} fluxes were at their lowest in 2016 when the water table was near, or at, the surface of the ridge and trough microforms. Higher water tables were also associated with greater vegetation productivity in the ridge microforms, in line with the presence of bryophyte communities which are vulnerable to moisture stress. It is possible that bryophyte productivity was limited during 2015 when precipitation and WTDs were low. The CO_2 flux response to higher WTDs in the current study are in line with the consensus in the literature that CO_2 emissions can be reduced through rewetting (Wilson et al., 2016b). However, net CO_2 emissions observed from the troughs rewetted within the RW-16 treatment also indicate that although water tables need to be close to the surface, inundation which submerges vegetation substantially reduces productivity and therefore is not recommended. Further to this, deep inundation may pose a challenge for colonisation of *Sphagnum* spp. with recolonisation only observed where flooding is shallow and sheltered or where loose floating rafts have formed (Money and Wheeler, 1999; Smolders et al., 2002). Due to the gas flux measurements occurring immediately after rewetting, it is too soon to tell whether the inundation that occurred is too deep for colonisation, or whether *Sphagnum* spp. and productivity will re-establish. The higher CH_4 emissions under rewetted conditions highlights that although the raising the water table may reduce CO_2 emissions, this 'saving' could be counterbalanced by an increase in CH_4 emissions which has a greater radiative forcing. Furthermore, inundation can cause fresh, easily decomposable plant litter to become susceptible to anaerobic decomposition resulting in high methane emissions (Vaneslow-Algan et al., 2015).

The creation of low-lying bunds to aid the rewetting of peatlands is increasingly common in lowland raised bog restoration in the UK, with the colonisation of the cells and bunds by *Sphagnum* spp. observed within three years at Glasson Moss, Cumbria (Natural England). At Pant-y-Dwn, no evidence of bund colonisation by *Sphagnum* spp. was observed during the

study period, however, the shallow flooding and sheltered conditions created by the bund appear favourable for future colonisation, particularly given the available seed bank from the surrounding bog. Indeed, *Eriophorum* spp. had begun to colonise the edges of the cells and bunds, which in addition to being a good companion species for mosses (Ferland and Rochefort, 1997), may impact the C balance due to its ability to act as a conduit for CH_4 emissions. The development of vegetation along the bund and across the site in line with the newly established high water level will be key in determining the CO_2 and CH_4 balance of the site. In turn this will also result in a shift in microbial populations reflecting the soil water conditions, and could result in higher CH_4 emissions (Tuittila et al., 2000).

3.5. Conclusion

The study presented here suggests that the bunding technique of rewetting can successfully raise the water table, though it may take more than one year for the water table to reach the surface if precipitation levels are below average. Rewetting at Pant-y-Dwn, together with the microforms remaining from historical peat cutting at the site, resulted in a heterogeneous microtopography analogous to natural peatlands. It is too soon to tell whether bryophyte colonisation will be successful across the bund and associated cells, however an observed increased productivity of bryophytes in high water table areas suggests that peat accumulation should re-establish, providing deep inundation does not occur. CH_4 emissions were higher under wetter conditions and where aerenchymatous vegetation prevailed, highlighting the potential of a positive net GHG balance if aerenchymatous vegetation colonised the bund preferably over bryophyte communities. This study monitored the immediate response of CO_2 and CH_4 fluxes to rewetting; the initial transition phase that is rarely reported for peatlands under restoration management and is crucial to identify whether a GHG 'pulse' is created during the rewetting process. Considerable uncertainty also surrounds the time required for peatland vegetation and microbial communities to reach equilibrium following rewetting, and therefore the long term monitoring of GHG fluxes is highly recommended to assess the success of rewetting as a climate mitigation tool.

3.6. References

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Chapter 4

Automated GHG flux measurements from a shallow-drained and rewetted grassland on peat

Abstract. Grasslands on peat soils are a large net C source, with losses dominated by CO_2 emissions due to the high rates of soil decomposition under aerobic conditions. In Wales, drainage of lowland raised bogs for agriculture occurred several decades ago, with many now under extensive grazing management and rewetting to reduce the net loss of C. Despite rewetting, substantial drawdown of the water table may still occur during summer which can result in large CO_2 emissions under high temperatures, and therefore the net GHG balance of rewetted grasslands on peat is still surround by uncertainty. This study measured CO_2 and CH_4 fluxes from a drained and adjacent rewetted grassland on peat, on the degraded outskirt of a lowland raised bog in west Wales. Automatic chamber measurements enabled high frequency fluxes to be obtained, whilst capturing the small scale spatial heterogeneity within the grasslands. CO_2 and CH_4 fluxes were measured between July 2015 and October 2016, and annual flux estimates were modelled using environmental drivers. The drained grassland was a small net CO_2 source (~23 g C m^{-2} yr^{-1}), and CH_4 neutral, whilst the rewetted grassland was a large net CO_2 sink (-323 g C m^{-2} yr^{-1}) and CH_4 source (21.7 g C m^{-2} yr^{-1}). Peak productivity between the two grasslands was similar, with differences in NEE driven by high nocturnal R_{ECO} fluxes from the drained grassland. This led to a greater diurnal fluctuation in CO_2 fluxes within the drained grassland compared to the rewetted grassland, highlighting the uncertainty of upscaling daytime only chamber measurements to annual fluxes. The high CH_4 fluxes from the rewetted grassland were associated with high soil temperatures when the water table was within 10 cm of the surface. Despite the high CH_4 fluxes from the rewetted grassland, the large net CO_2 uptake resulted in a negative GHG balance. This study indicates there is potential for rewetting grasslands on peat so the mean annual WTD is within a few cm of the surface, leading to net C gains, whilst the land is still viable for extensive grazing.

4.1. Introduction

Peatlands are a crucial component of the terrestrial carbon store, however their drainage for agriculture, forestry and extraction has led to large carbon losses. Drained organic soils are estimated to account for 10 % of all greenhouse gas (GHG) emissions from the agriculture, forestry and other land use (AFOLU) sectors (Smith et al., 2014), posing a significant GHG source. Drainage increases the aeration of peat soils beyond the acrotelm meaning microbial decomposition of soil organic matter is no longer inhibited by low oxygen availability. Consequently the rate of ecosystem respiration is increased and therefore CO_2 emissions to the atmosphere. The lowering of the water table also moves the zone of CH_4 production, which is at its maximum at the water table boundary, down the soil profile. This enhances the potential for oxidation as CH_4 diffuses through the larger aerated soil layer. Subsequently, CH_4 emissions to the atmosphere are lower from drained soils, though drainage ditches in peatlands have been found to be CH_4 hotspots (Peacock et al., 2017; Hendriks et al., 2007; Schrier-Uijl et al., 2010).

Across England and Wales, large areas of lowland peatland have been drained for agriculture, with an estimated 37 % of lowland peatlands in England converted to improved grassland and at least half of Welsh lowland raised bogs (Blackstock et al., 2010). Globally, drained peatlands are estimated to emit 0.9 $GtCO_2$-eq a^{-1} (Smith et al., 2014), with the UK one of the twenty countries with the largest CO_2 emissions from degraded peatlands (Joosten, 2009). Whilst many drained peatlands remain under arable management, there is growing interest in restoring abandoned agricultural and extraction peatlands to reduce biodiversity loss and promote ecosystem services such as flood mitigation and carbon sequestration (Peh et al., 2014). Subsequent land use management ranges from creating nature reserves (Hendriks et al., 2007; Schrier-Uijl et al., 2010; Peh et al., 2014) to extensive grasslands with low density grazing (Beetz et al., 2013; Beyer and Hoper, 2014; Renou-Wilson et al., 2016). Even without arable production, grasslands on peat soils may be significant carbon sources where soils are deeply drained and eutrophied (Renou-Wilson et al., 2014; Beyer et al., 2015), however water table control together with extensive management can result in smaller carbon losses or net carbon sequestration (Beetz et al., 2013; Görres et al., 2014).

The primary technique in restoring drained peatlands is rewetting, with a rise in water table aiming to re-establish vegetation and soil biota adapted to anaerobic conditions (IPCC, 2014). As the soil becomes saturated, decomposition is limited due to low oxygen availability,

resulting in a decrease in ecosystem respiration (R_{ECO}) and therefore CO_2 released to the atmosphere. Gross primary productivity (GPP) is also expected to decrease, but to a lesser extent. Renou-Wilson et al. (2016) found that with rewetting a smaller net CO_2 source was observed compared to a shallow-drained area of a grassland on peat in Ireland, with net carbon uptake with rewetting when grazing was also excluded. A synthesis of GHG emissions from German grasslands on organic soils by Tiemeyer et al. (2016) also found that in general shallow-drained sites had lower CO_2 emissions than deeply drained sites, although the relationship between CO_2 fluxes and water table depth was highly site specific. Whilst rewetting reduces CO_2 emissions, higher water tables can lead to higher CH_4 fluxes, for example CH_4 emissions from the rewetted grassland on peat measured by Renou-Wilson et al. (2016) were at least four times higher than those from the shallow-drained grassland on peat. CH_4 production relies upon the anaerobic decomposition of organic matter, and although this is a much slower process compared to the aerobic equivalent, CH_4 has a 25 times higher global warming potential compared to CO_2 over a 100 year time span (Forster et al., 2007) and therefore a greater warming effect on the atmosphere. Within the UK, peatland restoration efforts and associated carbon balance measurements have predominately focused on upland bogs due to their greater extent (and typically lower agricultural value) compared to lowland peatlands, with GHG fluxes from the latter only recently being addressed by the Defra project SP1210; Lowland peatland systems in England and Wales – evaluating GHG fluxes and carbon balances (Evans et al., 2017). The two extensive grasslands on peat monitored within the Lowland Peat Project (Tadham Moor in Somerset and Bakers Fen in East Anglia) were both found to have a net warming effect when the net ecosystem carbon balance was considered. Both sites had low water tables, particularly during the summer (at least 50cm below the surface) with the entire peat layer becoming aerated during all three summers measured at Bakers Fen. Although the Lowland Peat Project began to address the research gap with regards to quantifying the C and GHG balance of grasslands on peat within the UK, reliable flux data on drained and rewetted grasslands in the UK are extremely sparse. The quantification of carbon fluxes from peatlands is essential for national GHG reporting to the UN Framework Convention on climate change (IPCC, 2014), however the GHG exchange of drained peat soils varies not only due to fine scale edaphic conditions, but also land use, management practice and intensity (Renou-Wilson et al., 2016), complicating the production of emission factors and comparison across the literature.

The two principal methods of GHG flux measurements are chamber-based methods and micrometeorological methods (e.g. eddy covariance 'flux tower' systems). The latter provides a landscape scale flux estimation on a high temporal frequency whilst the former allows fine spatial scales to be measured, but manual efforts often mean the temporal resolution is considerably reduced. The development of automatic chamber systems allows GHG fluxes to be measured at the fine scale both spatially and temporally, improving confidence in empirical models derived from chamber measurements which are used to calculate annual flux estimates. Although meeting the required power demands is a challenge in remote peatland environments, automatic chamber systems enable short term events to be captured (e.g rainfall; Koskinen et al. 2014) improving our understanding of the controls on biogeochemical cycles. Additionally, the absence of field workers during chamber placement may reduce disturbance effects such as ebullition, increasing the reliability of fluxes measured.

The aim of this study was to investigate the effect of rewetting on CO_2 and CH_4 fluxes from a grassland on peat, compared to an adjacent shallow-drained grassland on peat. The hypotheses of this study are: (1) the shallow-drained site will act as a net source of CO_2 due to higher rates of ecosystem respiration whereas the rewetted sites will act as net sinks of CO_2; (2) the rewetted sites will act as net sources of CH_4, whereas the shallow-drained site will be near neutral or a small sink of CH_4; (3) diurnal fluctuations in net ecosystem exchange will be greater in the shallow-drained site compared to the rewetted site; (4) CH_4 fluxes will increase with soil moisture, whilst CO_2 emissions will decrease as soil moisture increases.

4.2. Methods

4.2.1. Study area

The study area was located on the degraded outskirt of a lowland raised bog, Cors Fochno, on the west coast of Wales (latitude/longitude: 52.509, -4.046). The area experiences a maritime temperate climate with mean annual air temperature of 10.1 °C and annual precipitation of around 1070 mm (Gogerddan met station). The study area has been under conservation management by Natural Resource Wales since the early 1990s, following drainage for agriculture during the 19th century. It is currently managed as a conservation grassland with sheep grazing from April to June (stocking density ~1.24 sheep ha^{-1}), followed by cattle from June to November (stocking density, 0.47 cows ha^{-1}), as well as horses from May – November (maximum of 6).

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Three sites were identified within the study area; two sites which have undergone rewetting through the blocking of surrounding field ditches (grazing compartment 91 = RW-91 and 92 = RW-92) and one site which remains shallow-drained (grazing compartment 78), due to an active drain between the site and an adjacent caravan park (Fig. 4.1). The rewetted sites are a mixture of rush pasture and wet grassland, dominated by *Juncus effusus* and *J. acutiflorus* with locally abundant *J. articulates* and *Carex* species, whilst the shallow-drained site is similar but with a higher dominance of dry grassland (NRW site survey, 2005). Soil characterisation representing the shallow-drained and rewetted site conditions (Table 4.1) was conducted using the same methods as those described in Ch. 2 (section 2.2.2.).

Table 4.1. Site description: mean soil properties and annual water table depth for the shallow-drained and rewetted grasslands. ± 1 standard error (SE) of the mean is shown in brackets.

Site (no. pairs of collars)	pH	EC ($\mu S\ cm^{-1}$)	Moisture content (%)	Organic matter content (%)	Water table (cm)
Shallow-drained (7)	5.3 (0.08)	226 (85.6)	66.7 (1.27)	55.6 (3.13)	-37
Rewetted (14)	5.51 (0.05)	84.8 (7.09)	87.3 (1.44)	85.2 (2.38)	-2/+2
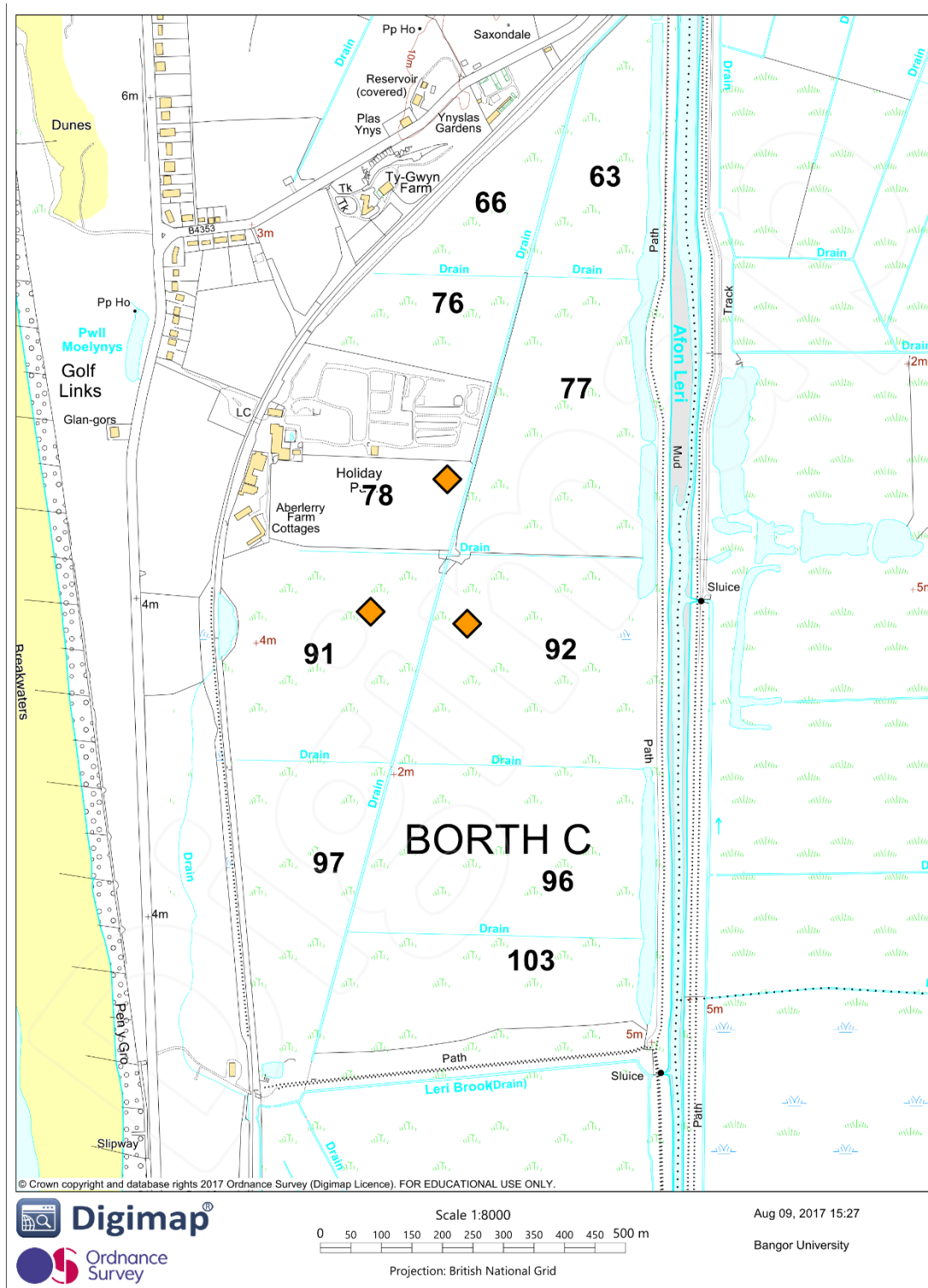
Figure 4.1. Site map of Cors Fochno, with collar locations identified within the shallow-drained site grassland (field compartment 78) and rewetted grassland (field compartments 91 and 92).

4.2.2. Greenhouse gas flux measurements

GHG measurements were carried out from July 2015 to July 2016 in the rewetted sites and from March 2016 to October 2016 in the shallow-drained site using the LI-8100A automated soil gas flux system (LI-COR Biosciences, USA). Soil collars (diameter = 21 cm) were installed to a depth of approximately 5 cm along a 30 m transect which was established in each site perpendicular to the main field ditch. The collars were installed in pairs as the LI-8100A chambers used cannot measure light and dark fluxes over the same collar, thus the transparent (NEE) and opaque (R_{ECO}) chambers were placed over adjacent collars, resulting in 7 measurement locations in each site. Gas fluxes were measured at 1.5 hour intervals for up to 48 hours in each site, with each chamber closure lasting for 2.5 minutes. To ensure flushing of the system between chamber measurements a 30 second pre-purge and post-purge was used. Chambers were positioned such that the collars were not shaded throughout the day, minimising the effect of the chambers being left *in situ* for 48 hours. A Los Gatos Ultra-Portable GHG Analyser (UGGA, Los Gatos Research, USA) was integrated with the LI-8100A system to enable CH_4 fluxes to be measured simultaneously. Inundation of the rewetted sites precluded the deployment of the system from October to April as the water table was more than 20cm above the surface during this time, and so the system would have been partially submerged. Overall, six monthly measurement campaigns were carried out at the rewetted sites (July – September 2015; May 2016 – July 2016) and eight at the shallow-drained site (March 2016 – October 2016).

Each automated chamber was connected to a soil moisture sensor (ECH2O 5cm probe, Decagon Devices, WA, USA) and soil temperature probe (8150-203 soil temperature probe, LI-COR Biosciences), both of which were inserted at 5cm depth. In addition to this, each transparent (NEE) chamber had a PAR sensor (LI-190 Quantum Sensor, LI-COR Biosciences) attached which was installed adjacent to the chamber. Measurements from these probes were taken concurrently during the chamber closure period and averaged to provide a single reading for each chamber closure. Chamber headspace temperature and relative humidity were measured by the LI-8100A.

Figure 4.2. Automatic chamber set up: the clear chamber (for measuring NEE) and opaque chamber (for measuring R_{ECO}) shown in the shallow-drained grassland.

4.2.3. Flux calculation

CO_2 and CH_4 fluxes were calculated based on the linear change in headspace concentration over time using Eq. 4.1;

$$F_{GHG} = \frac{\partial c}{\partial t} \cdot \frac{V}{A} \qquad [4.1]$$

Where F_{GHG} is the calculated flux in gGHG m^{-2} hr^{-1}, $\delta c/\delta t$ is the change in gas concentration over time, V the total volume inside the chamber, aboveground portion of the collar and the gas tubing in m^3 and A the area of the collar in m^2.

Due to the large number of chamber measurements, a standard flux test time for CO_2 data from the LI-8100A was established to enable automatic processing. The standard flux start time was 15 seconds after chamber closure and end time was 135 seconds after chamber closure, giving a flux test time of 120 seconds. To assess whether there was any disturbance (e.g. ebullition) or saturation effects which could affect the flux calculation, the concentration change during the test time was visually assessed and if the standard start and end times were not suitable (e.g. saturation had occurred within 135 seconds of chamber closure), times were manually chosen. CH_4 concentrations from the Los Gatos UGGA were matched by timestamp to the LI-8100A data, with the flux test time manually selected.

Fluxes were accepted if the r^2 of the slope was greater than 0.7 and p value was < 0.05, however if fluxes did not meet this criteria they were visually inspected and retained if small but non-

zero (i.e. p < 0.05). Negative fluxes denote the uptake of gas by the ecosystem and positive fluxes a loss to the atmosphere. GPP was calculated as the difference between NEE and R_{ECO}.

4.2.4. CO₂ flux modelling

4.2.4.1. Gross Primary Productivity

The transparent chambers absorb approximately 7 % of incoming radiation, therefore the PAR data were corrected by a factor of 0.93 prior to modelling. Gross primary productivity was modelled for individual chambers using the Michelis-Menten type relationship with PAR for each measurement campaign using Eq. 4.2;

$$GPP = \frac{GP_{max} \cdot \alpha \cdot PAR}{GP_{max} + \alpha \cdot PAR} \quad [4.2]$$

where GPP is the measured gross primary productivity (gCO_2 m^{-2} hr^{-1}) and PAR the photosynthetically active radiation (W m^{-2}), α is the initial slope of the regression curve (gCO_2 m^{-2} hr^{-1}/W m^{-2}) and GPmax is the limit of GPP when approaching infinite PAR (gCO_2 m^{-2} hr^{-1}).

Values for parameters GPmax and α between measurements dates were calculated using linear interpolation on a daily basis. Due to the lack of measurements throughout the winter and early spring in the rewetted sites, model parameters derived for the end of the growing season (September) were maintained throughout winter (until March 1st) so that interpolation did not artificially increase ecosystem assimilation during winter when vegetation was dormant. Hourly GPP fluxes were calculated using hourly PAR measurements from the nearby automatic weather station (AWS) with the interpolated model parameters.

4.2.4.2. Ecosystem Respiration

R_{ECO} was modelled using the Arrhenius model approach of Lloyd and Taylor (1994) parameterised with soil temperature at a depth of 5 cm (Eq. 4.3) using data from the entire study period to obtain the maximum soil temperature range. Due to the measurement of R_{ECO} by clear (NEE) chamber during the night time, as well as R_{ECO} measured by the opaque chambers, R_{ECO} models were fitted using all the data from the opaque chambers as well as nocturnal CO_2 fluxes from the clear chambers (23:00 – 04:00).

$$R_{ECO} = R_{ref} \, exp\left\{E_0 \left(\frac{1}{T_{ref} - T_0} \frac{1}{T_{soil} - T_0}\right)\right\}, \quad [4.3]$$

where R_{ref} and E_0 are parameters fitted using the 'nls' function in R, T_{ref} is the reference temperature (283.15 K), T_0 is the temperature constant for the start of biological processes (227.13 K) and T_{soil} is the soil temperature at 5cm (in Kelvin). Models were fitted separately for each pair of chambers, with hourly R_{ECO} fluxes calculated from the continuous AWS soil temperature measurements.

4.2.4.3. Annual CO_2 fluxes

NEE was calculated for each chamber pair on an hourly basis as the sum of GPP and R_{ECO} fluxes for each chamber pair. Annual flux estimates for GPP, R_{ECO} and NEE were then produced by summing the hourly values from 1 April 2015 to 31 March 2016.

4.2.5. CH_4 flux modelling

Annual CH_4 fluxes were modelled using an exponential relationship with the soil temperature at 5 cm depth as the sole explanatory variable, as in Eq 4.4 below;

$$CH_4 = t1 * t2^{(T-T_{ref})/10} \quad [4.4]$$

where t1 and t2 were estimated using non-linear least squares regression, T is 5cm soil temperature, T_{ref} is 10 °C. Models were fitted by site, using the measurements from all the collars for the whole study period due to the limited data availability at the two rewetted sites. Hourly CH_4 measurements were modelled using the exponential regression and continuous AWS measurement, then summed to provide annual flux values.

4.2.6. Net ecosystem carbon balance and net GHG flux

The net ecosystem carbon balance (NECB) was derived from combining the modelled annual gaseous CO_2 and CH_4 fluxes. Although the shallow-drained site was cut once a year, biomass was left in-situ and therefore was not included as a loss to the system. Carbon import – export from on-site manure was also assumed to be negligible due to the low stocking densities. To fully assess differences in the site GHG dynamics in this study, CH_4 fluxes were converted to CO_2 equivalent using a forcing factor of 25 over a 100-year horizon (Forster et al., 2007) in accordance with the approach used for the UK GHG Inventory (Brown et al., 2016). The net GHG flux (t CO_2-eq ha^{-1} a^{-1}) was calculated for each chamber pair using chamber specific NEE and site specific CH_4 fluxes. Whilst considering only terrestrial CO_2 and CH_4 fluxes does not represent the full GHG balance, this terminology is used to refer to the net gaseous carbon

exchange expressed in CO_2-equivalents for simplicity, on the assumption that nitrous oxide emissions from these unfertilised sites can be considered negligible (Hendriks et al., 2007).

4.2.7. Statistical analysis

Correlations between gas fluxes and environmental drivers were identified using Kendall's Tau correlation and a t-test or Mann Whitney U test was used to identify whether soil characteristics differed between the shallow-drained and rewetted sites. Transformation of gas flux data did not result in normally distributed data, therefore the Friedman test was used to identify differences in measured GHG fluxes between sites. A one-way ANOVA was performed on annual flux values to determine differences.

Model performance was rated using the thresholds outlined by Hoffmann et al. (2015) following the same approach as chapter 2. Model uncertainty was estimated using the approach outlined by Renou-Wilson et al. (2016) which captures both the random errors from measurement uncertainties and the scatter in the model results. The standard error of the model is represented by a percentage of the mean fluxes as shown in Eq. 4.5, which is then applied to the annual GPP and R_{ECO} balance.

$$E_r = \sqrt{\sum_{i=1}^{n} \frac{(F_{obs} - F_{mod})^2}{(n-1)*n}} \quad [4.5]$$

where E_r is the model standard error, F_{obs} is the measured flux and F_{mod} is the modelled flux and n is the total number of measured fluxes. The uncertainty in annual NEE fluxes was calculated using the law of error propagation as the square root of the sum of the squared standard errors of GPP and R_{ECO}.

4.3. Results

4.3.1. Environmental conditions

During the study period the mean annual air temperature was 11.5 °C (1 April 2015 – 31 March 2016), with total annual rainfall of 1100 mm (Aberporth, lat long: 52.139 -4.570). Soil temperatures showed a distinct seasonal cycle with the highest temperatures recorded in June at all sites (Fig. 4.3a). The highest soil temperatures were recorded at the shallow-drained site, though there was little difference in monthly mean soil temperatures between sites. A typical diurnal cycle in soil temperatures was observed with temperatures peaking in late afternoon to early evening (Fig. 4.3b).

The soil pH across the study area was slightly acidic with values from the rewetted sites being marginally greater than the shallow-drained site (5.5 and 5.3 respectively). The electrical conductivity of the shallow-drained site was significantly higher than that of the rewetted site ($p = 0.017$), whilst the soil organic matter content of the shallow-drained site was significantly lower than the rewetted sites ($p < 0.05$).

The rewetted sites had a mean annual water table within 3 cm of the surface (+2 cm in RW-92 and -2.5 cm in RW-91), whilst the shallow-drained site had a mean annual water level of -37 cm below the surface. The water table levels followed a seasonal pattern in line with precipitation with a maximum summer drawdown to -11 cm (RW-92), -20 cm (RW-91) and -50 cm (SD) during July. The water table was above the surface in both rewetted sites from October to April, whilst the shallow-drained site had a winter water table approximately -20 to -30 cm below the surface.

The soil moisture sensors recorded water contents ranging from -0.6 to 2.1, indicating that the sensors were not always operating correctly. Volumetric moisture content (VMC) measurements were taken every second during chamber closure and where values were measured less than 0 or greater than 1, all VMC measurements from that chamber closure were omitted to provide confidence in the measurements used. This led to the exclusion of 16% of VMC data points (18 % of R_{ECO} and 14 % of NEE VMC measurements). A seasonal pattern was evident in monthly mean VMC at the shallow-drained site, decreasing to a minimum in June 2016 then rising again to levels seen at the start of the study (Fig. 4.3c). The large range in VMC measurements was found across all collars during each of the 48 hour chamber deployments. A similarly large range in VMC measurements was found at the rewetted sites, though the limited autochamber deployment meant a seasonal pattern in VMC could not be identified. The mean monthly VMC was higher at the rewetted sites compared to the shallow-drained site, though values did not exceed 38 % except for site RW-92 in June when a moisture content of 79 % was recorded. The low VMC measurements contradicted the water table measurements which were close to the surface for the majority of the study period, consequently, the VMC measurements were removed from further analysis.
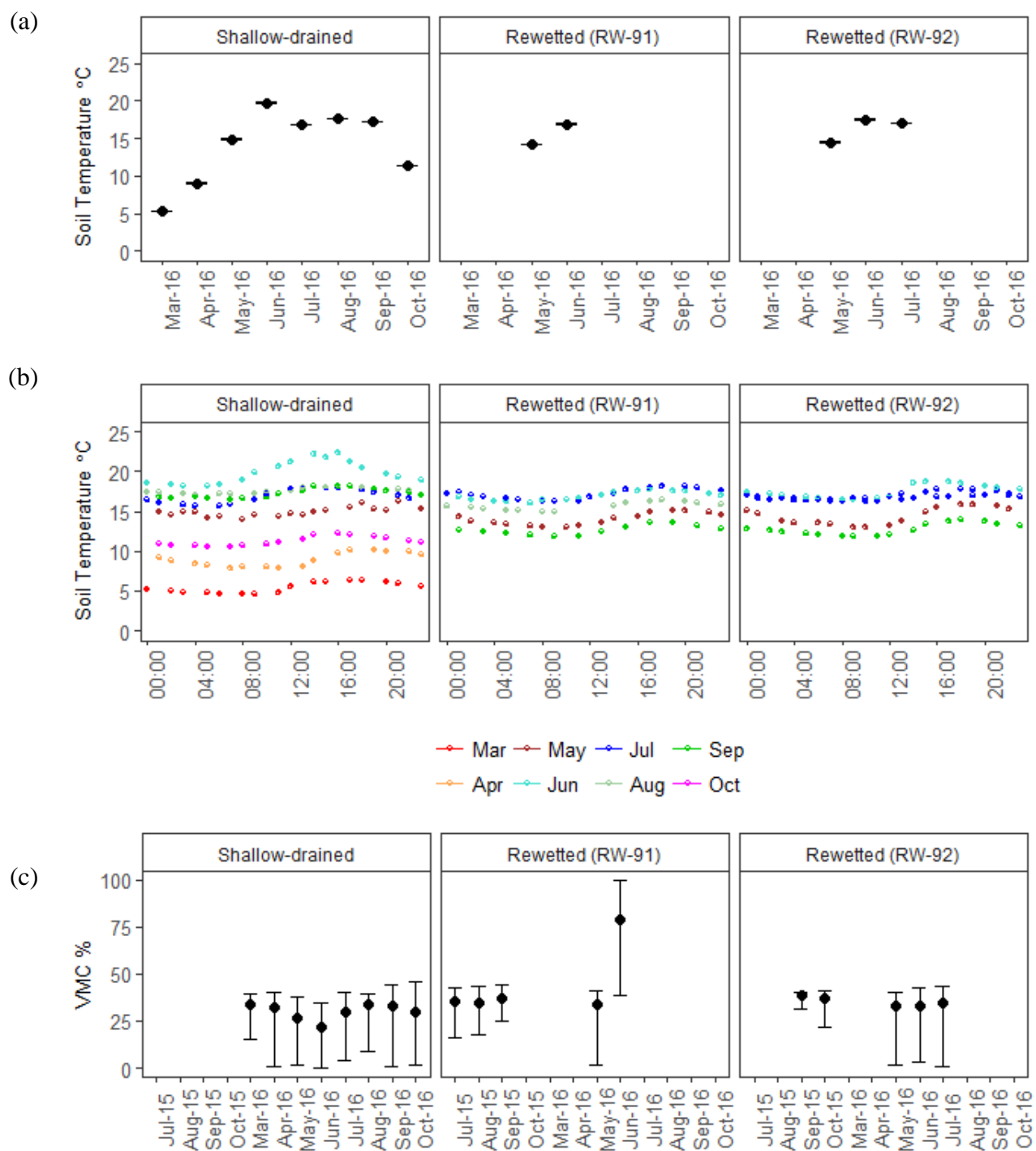
Figure 4.3. Average site (a) monthly soil temperature at 5 cm depth; (b) hourly soil temperature at 5 cm depth; (c) monthly volumetric moisture content. Error bars represent ±1 SE of the mean.

4.3.2. Diurnal CO_2 fluxes

The diurnal cycle of NEE was broadly consistent across the three sites (Fig. 4.4), with net CO_2 sequestration during the day and CO_2 emission at night, however some collars in the shallow-drained site also acted as net sources of CO_2 throughout the day (June 2016; July 2016). NEE at the two rewetted sites had a similar diurnal pattern in terms of magnitude and timing of the daily switch between source and sink. NEE fluxes from the shallow-drained site had greater diurnal fluctuations compared to the rewetted sites, particularly during June 2016 when the highest R_{ECO} values were measured. The difference in NEE between the shallow-drained and rewetted sites was mostly driven by the higher nocturnal CO_2 emissions at the former, however the latter also had slightly greater CO_2 uptake during May and July 2016. Diurnal variations in R_{ECO} were less prevalent, with no obvious diurnal pattern at the rewetted sites (Fig. 4.5). On the other hand, the shallow-drained site generally showed a peak in CO_2 emissions around midnight, a pattern which was more evident from May – July 2016. The full time series for CO_2 fluxes for each measurement campaign is shown in Fig. 4.6 for NEE and Fig. 4.7 for R_{ECO}.
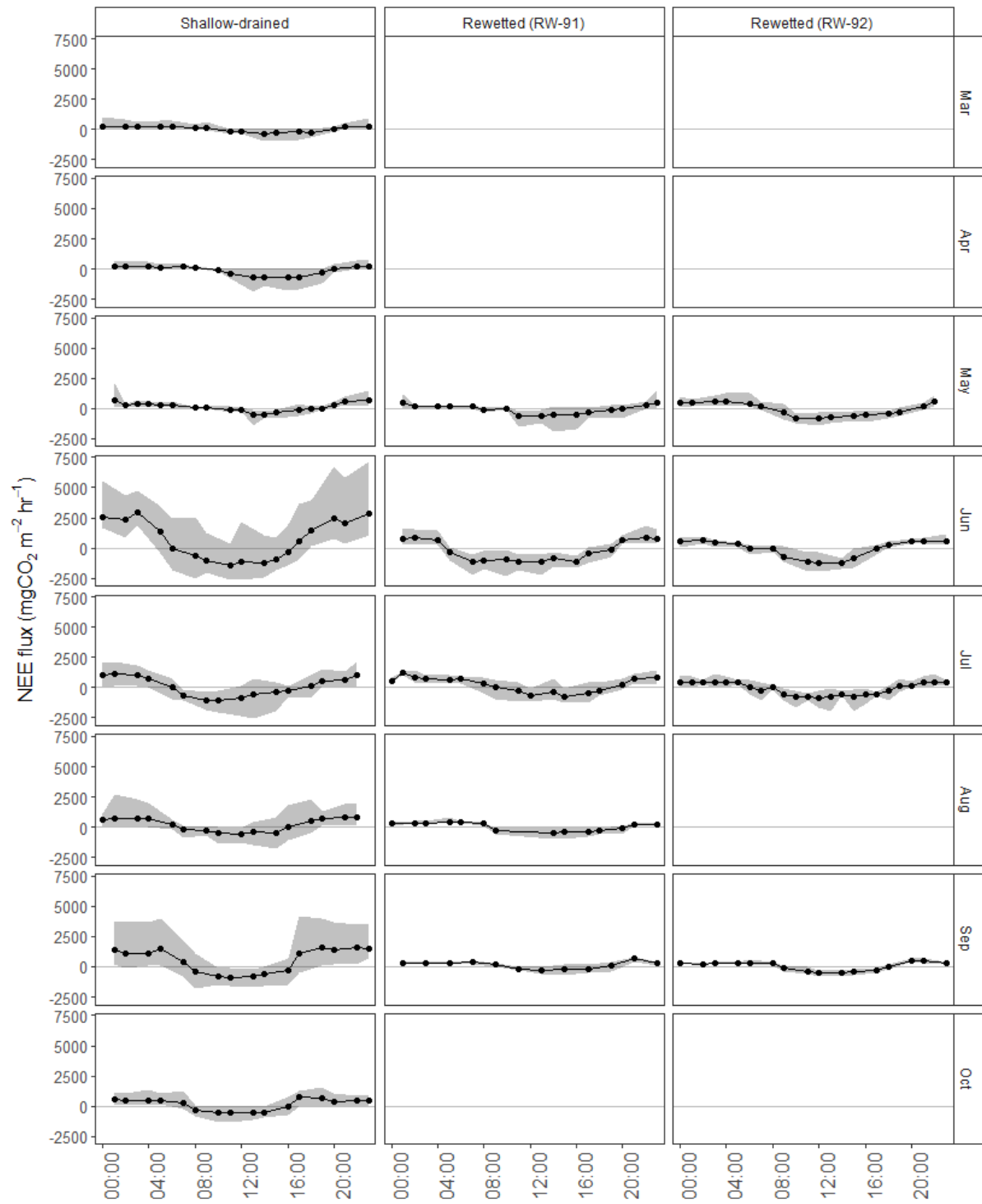
Figure 4.4. Average hourly measured NEE fluxes (points) for each site over 24 hours with minimum and maximum NEE fluxes represented by the shaded area.
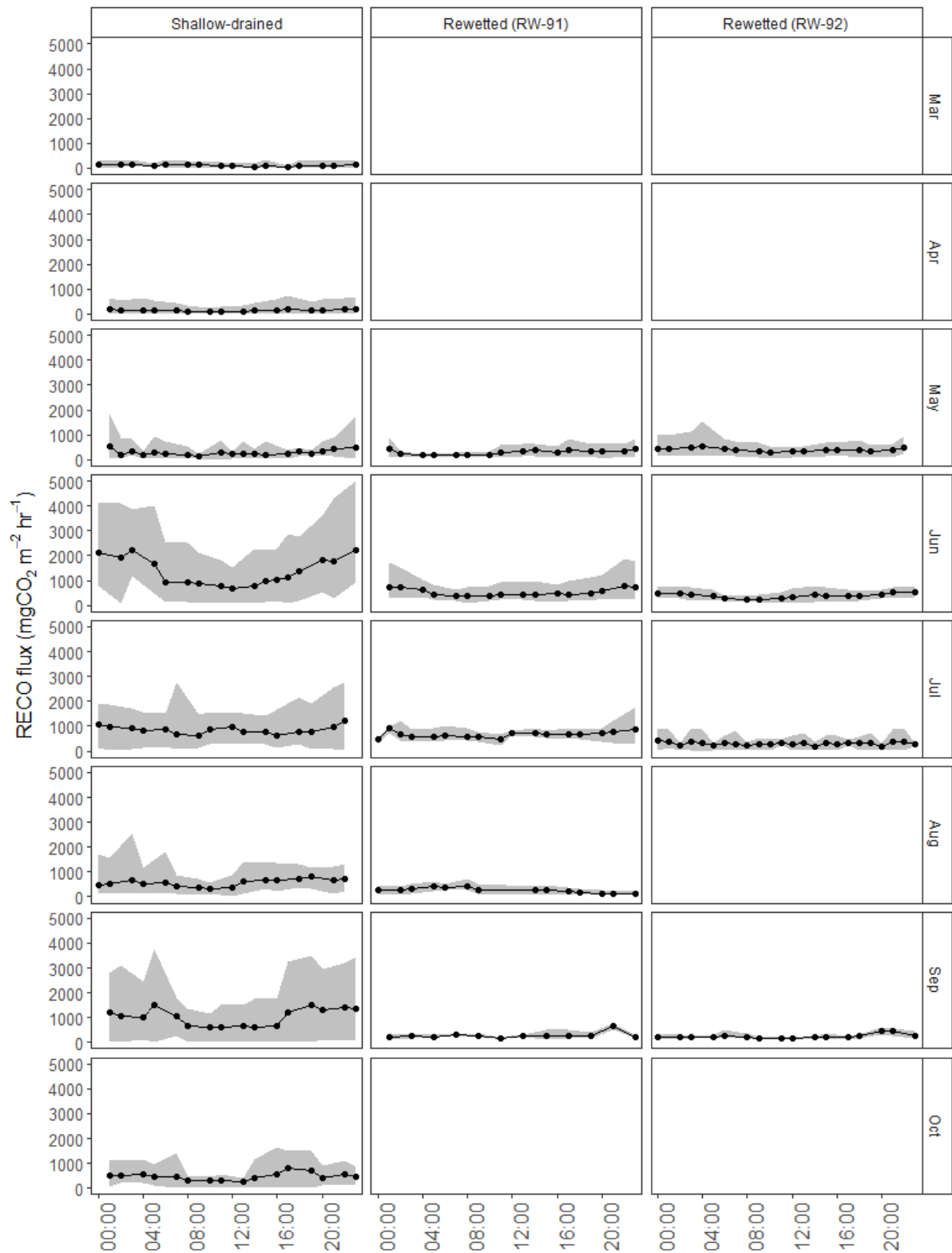
Figure 4.5. Average hourly measured R_{ECO} fluxes (points) for each site over 24 hours with minimum and maximum R_{ECO} fluxes represented by the shaded area. R_{ECO} fluxes here are from opaque chamber measurements only.
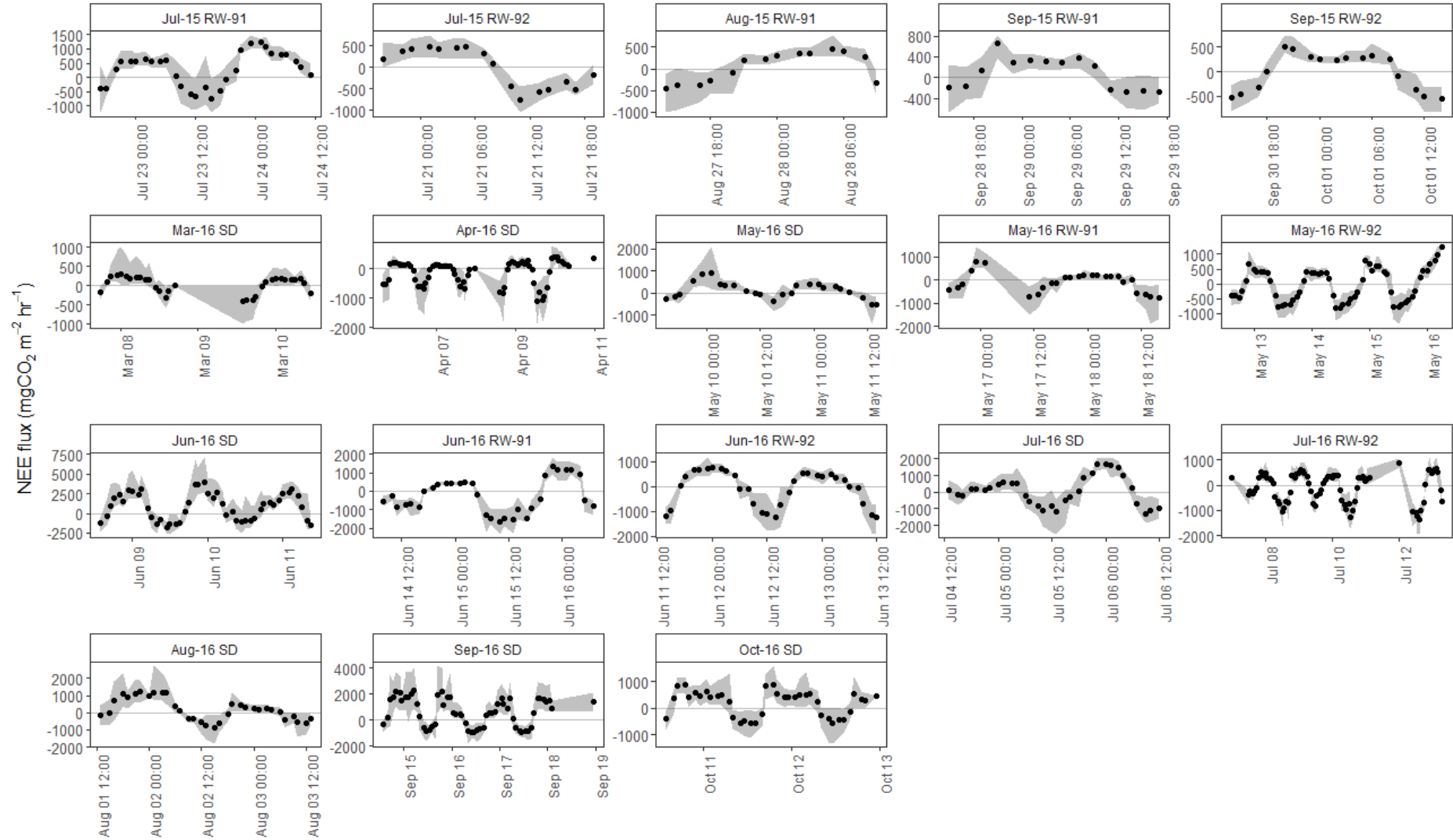
Figure 4.6. Average hourly measured NEE fluxes (points) for each site during each system deployment, with minimum and maximum NEE fluxes represented by the shaded area. Note the different scales between months.
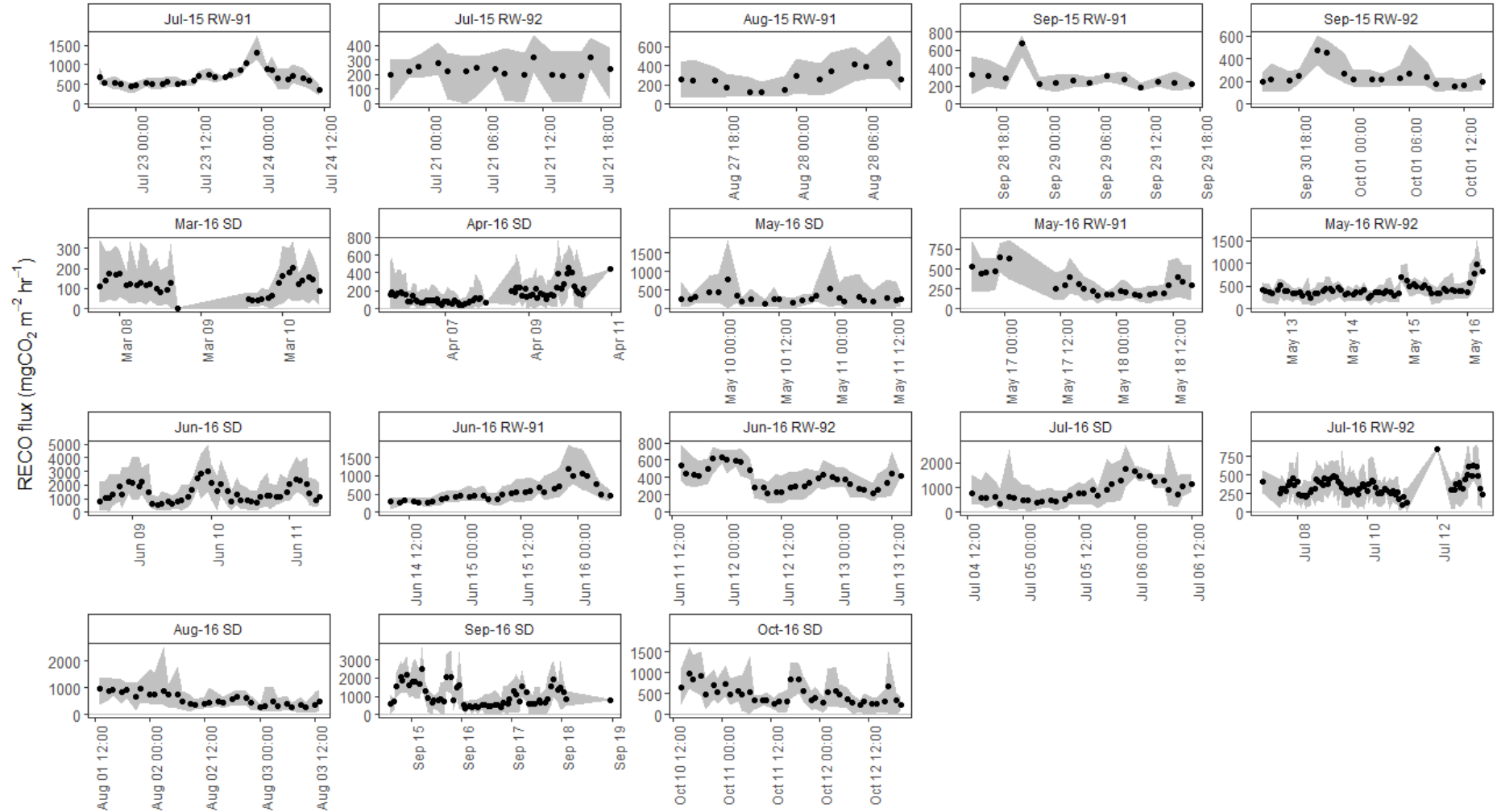
Figure 4.7. Average hourly measured R_ECO fluxes (points) for each site during each system deployment, with minimum and maximum R_ECO fluxes represented by the shaded area. Note the different scales between months.

4.3.3. Gross Primary Productivity

4.3.3.1. GPP flux drivers

GPP was significantly correlated with PAR at all collars within the study, with an expected increase in productivity (i.e. more negative GPP) as light levels increased (Table 4.2). The correlation between GPP and PAR was generally stronger in the rewetted grassland compared to the shallow-drained site. Soil temperature was also significantly correlated with GPP at all collars in the shallow-drained site, and the majority of collars in the rewetted sites. Generally GPP became more negative as soil temperatures increased, indicating a seasonal control on productivity, though two collars in the rewetted grassland (RW-91 collar 1 and RW-92 collar 13) showed a positive correlation with soil temperature ($p = 0.01$).

Table 4.2: Kendall tau correlation coefficients for GPP and soil temperature, and PAR. Non significant results (p > 0.05) shown by NS

Collar ID	GPP Soil temp	GPP PAR
Shallow-drained grassland		
SD.1	-0.242	-0.598
SD.3	-0.135	-0.419
SD.5	-0.315	-0.457
SD.7	-0.378	-0.32
SD.9	-0.318	-0.445
SD.11	-0.449	-0.463
SD.13	-0.415	-0.486
Rewetted grassland (RW-92)		
RW92.1	-0.256	-0.599
RW92.3	-0.058 NS	-0.599
RW92.5	-0.125	-0.473
RW92.7	-0.098 NS	-0.664
RW92.9	-0.15	-0.529
RW92.11	0.028 NS	-0.699
RW92.13	0.139	-0.698

	Rewetted grassland (RW-91)	
RW91.1	0.563	-0.567
RW91.3	-0.009 NS	-0.529
RW91.5	-0.202	-0.477
RW91.7	-0.276	-0.616
RW91.9	-0.329	-0.497
RW91.11	-0.415	-0.426
RW91.13	-0.182	-0.53

4.3.3.2. GPP model parameters and performance

The light response parameters GPmax and α showed seasonal variation with the greatest ecosystem assimilation capacities (represented by GPmax parameter, Fig. 4.8a) from May to June in both the shallow-drained site and rewetted sites, coinciding with increasing light response efficiencies (indicated by the *α* parameter, Fig. 4.8b). Parameter development at the rewetted sites was similar, though the peak in light response efficiency was slightly greater at RW-91 compared to RW-92. The shallow-drained site had a consistently greater light response efficiency however there was greater variability between collars compared to the rewetted sites where model parameters were more consistent (Appendix C, Fig. C1 for individual collar parameters).

(a)

(b)
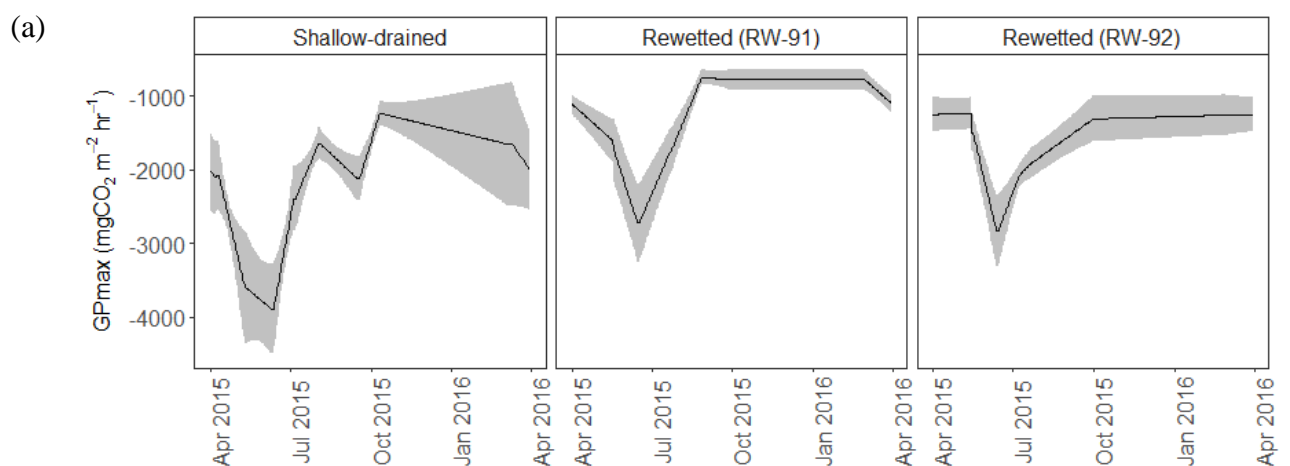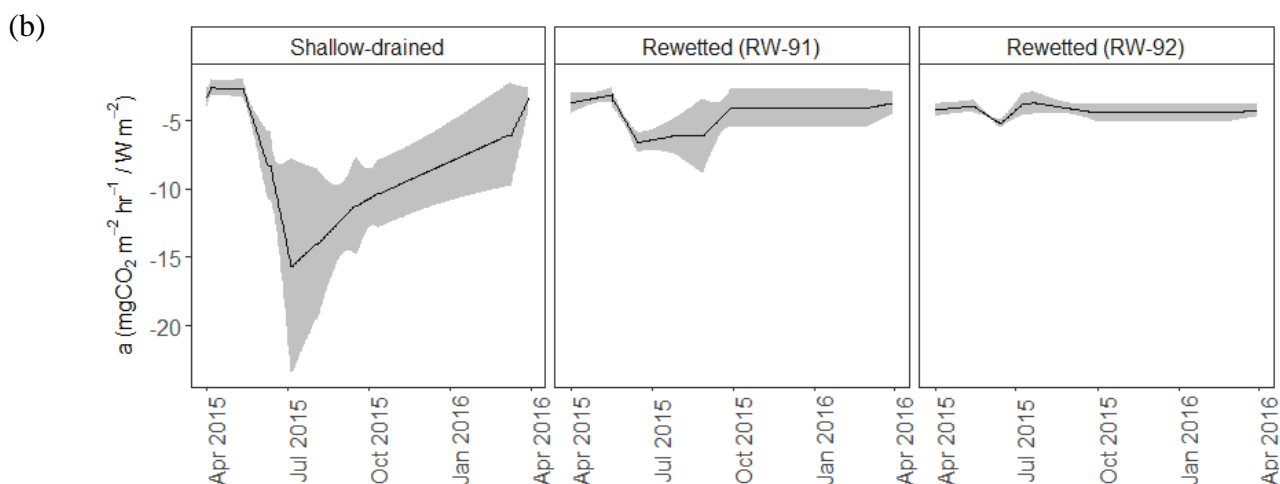
Figure 4.8. Interpolated GPP model parameters. Average site model parameters (a) GPmax and (b) alpha; shown by the solid line for the period of annual flux modelling (1 April 2015 to 28 March 2016). The shaded area represents the site minimum and maximum parameter values.

Comparison of the measured and modelled GPP fluxes generally showed an even scatter around the 1:1 line for each site, though in the shallow-drained site the most negative GPP fluxes were underestimated by the models (Fig. 4.9). GPP model performance, rated using the co-efficient of determination (r^2), was at least satisfactory in 91 % of the models fitted, with 75 % having a good-to-excellent fit (Table 4.3). For collars where satisfactory models could not be attained, model parameters were interpolated between the previous and subsequent measurement dates. All except one of the models which did not achieve a satisfactory fit were from the shallow-drained site (Appendix C, Fig. C2) and were associated with either small ranges in PAR or high variability in GPP at low light levels.
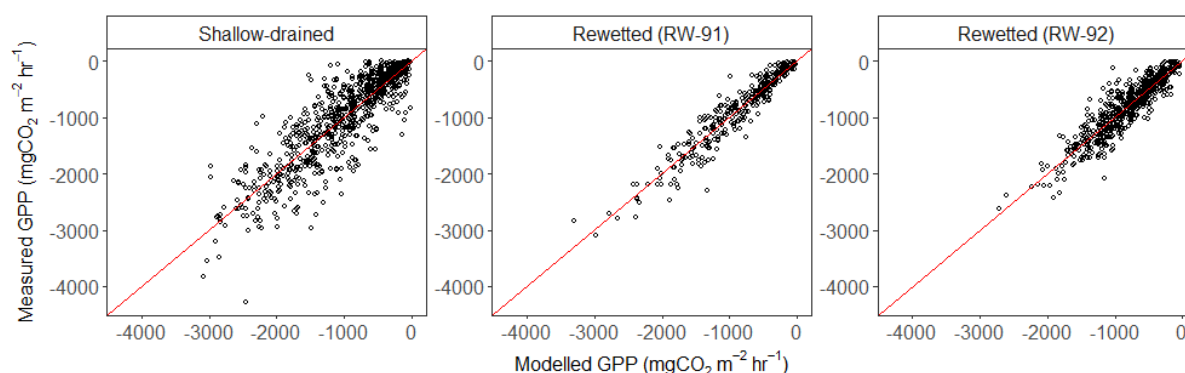
Figure 4.9. GPP flux model validation. Measured v modelled GPP fluxes for each site. The solid line represents the 1:1 line.

Table 4.3. CO_2 model fits. Co-efficient of determination (r^2) for GPP models for individual measurement collars per campaign and R_{ECO} models for individual collars using data from the entire study period. [NS] denotes the model was not satisfactory (Appendix A, Table A2).

Collar ID	GPP Mar	GPP Apr	GPP Apr/May	GPP May	GPP Jun	GPP Jul	GPP Aug	GPP Sep	GPP Oct	R_{ECO}
Shallow-drained grassland										
SD.1	0.86		0.84		0.22[NS]	0.67		0.36	0.87	0.18[NS]
SD.3	0.75		0.57		0.74	0.21[NS]	0.20[NS]	0.38	0.61	0.44
SD.5	0.92			0.51	0.70	0.10[NS]	0.44	0.83	0.77	0.42
SD.7	0.00[NS]		0.67		0.74	0.25	0.35	0.48		0.30
SD.9		0.10[NS]		0.50	0.64	0.35	0.45	0.11[NS]		0.15[NS]
SD.11	0.43			0.58	0.85	0.66	0.85	0.78	0.06[NS]	0.29
SD.13	0.80	0.44		0.56	0.79	0.73	0.32	0.68	0.66	0.26
Rewetted grassland (RW-91)										
RW91.1						0.14[NS]	0.89	0.41		0.39
RW91.3				0.68	0.90	0.82	0.45			0.26
RW91.5					0.90	0.87	0.94			0.37
RW91.7				0.88	0.89		0.71	0.92		0.16[NS]
RW91.9				0.76	0.89		0.79	0.98		0.37
RW91.11				0.70	0.81	0.89	0.39	0.63		0.34
RW91.13				0.56	0.94		0.92	0.83		0.64

Rewetted grassland (RW-92)

RW92.1		0.73	0.85	0.85	0.57
RW92.3	0.53	0.97	0.25	0.97	0.03 [NS]
RW92.5	0.58	0.93	0.74	0.92	0.03 [NS]
RW92.7	0.85	0.97	0.74		0.01 [NS]
RW92.9	0.30	0.52	0.63	0.91	0.01 [NS]
RW92.11	0.59	0.96	0.91	0.62	0.03 [NS]
RW91.13	0.82	0.94	0.70	0.94	0.08 [NS]

4.3.3.3. Annual GPP fluxes

Annual modelled GPP was greatest within the shallow-drained site, with an average CO_2 uptake of -1145 ± 59 gC m^{-2} a^{-1}, whilst CO_2 uptake from the rewetted sites was -849 ± 68 gC m^{-2} a^{-1} and -780 ± 79 gC m^{-2} a^{-1} for RW-91 and RW-92 respectively. Gross primary productivity was generally consistent across collars in the rewetted sites, with the exception of collar RW92.7 which had the strongest CO_2 uptake, and RW91.1 which had the weakest CO_2 uptake (Table 4.4). The high uptake in CO_2 by collar RW92.7 coincides with the stronger light response curve compared to the other collars in the site as shown in Fig. 4.10a (by the red line), which due to the interpolation of model parameters led to a greater ecosystem assimilation capacity and light response efficiency compared to the rest of the site. The small CO_2 uptake modelled for collar RW91.1 may be explained by the poor model fit during July ($r^2 = 0.14$, Fig. C2) coupled with the lack of data earlier in season due to the collar being submerged until May, and equipment failure during the June measurement campaign. In July at collar RW91.1, the linear regressions used to calculate fluxes were generally poor, with those at the highest PAR levels failing to meet the r^2 and significance threshold (flux calculation $r^2 = 0.08$ to 0.26). Consequently, only CO_2 fluxes from a small PAR range (up to 307 µmol m^{-2} s^{-1}) were included in determining the GPP model. Due to the unsatisfactory performance of the July model in RW91.1 and the lack of data earlier in the growing season, a satisfactory light response curve was not achieved until August, and therefore this collar was omitted when calculating the average site CO_2 fluxes.

Similarly, collar RW91.9 had a smaller GPP compared to the other collars within RW-91 with very good model fits (r^2 0.76 to 0.98) though maximum PAR values were relatively low, with only 7 out of 46 measurements over the whole study period reaching PAR > 1000 µmol m^{-2} s^{-1}. Measurements were captured throughout the growing season (May, June, August, September) however the light response curve measured in June was smaller than those from the rest of the site (Fig. 4.10b, red line = RW91.9) with maximum productivity reached at a much lower PAR. Coupled with the lack of data measured in July, this resulted in the modelled GPP annual flux of RW91.9 (-5.74 tC ha^{-1} a^{-1}) being smaller than the rest of the site. Collar RW91.9 was retained in calculating the site average CO_2 balance, though it likely represents the lower end of GPP fluxes for this site.
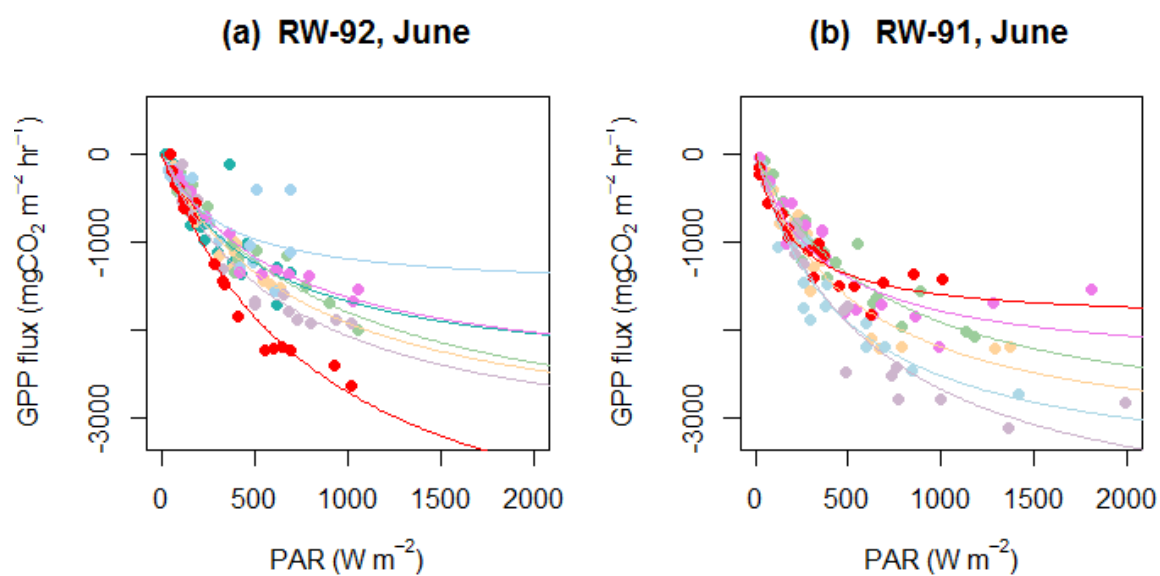
Figure 4.10. Light response curves for GPP in site (a) RW-92 in June, with collar RW92.7 represented by the red line; (b) RW-91 in June, with collar RW91.9 represented by the red line.

*Table 4.4. Annual budget of individual measurement collars and site averages for GPP, R_{ECO}, NEE, CH₄ and net GHG exchange in t CO₂-eq ha⁻¹ a⁻¹ using the 2007 IPCC standards with a radiative forcing of 25 for CH₄ and a time horizon of 100 years (Forster et al., 2007). Model error is shown in brackets. * denotes was omitted from the site average calculations.*

	GPP $gC\,m^{-2}a^{-1}$	R_{ECO} $gC\,m^{-2}a^{-1}$	NEE $gC\,m^{-2}a^{-1}$	CH₄ $gC\,m^{-2}a^{-1}$	Net GHG $tCO_2eq\,ha^{-1}\,a^{-1}$
Shallow-drained grassland					
SD.1 *	-1345 (67)	1419 (53)	74 (5)		
SD.3	-1234 (49)	1252 (52)	17 (1)		
SD.5	-1205 (36)	860 (30)	-345 (16)		
SD.7	-976 (48)	1019 (48)	43 (3)		
SD.9 *	-965 (39)	1032 (76)	67 (6)		
SD.11	-1015 (20)	1199 (53)	183 (9)		
SD.13	-1277 (38)	1494 (69)	218 (12)		
Site average	-1142 (59)	1165 (108)	23.2 (100)	1.19	1.25
Rewetted grassland (RW-91)					
RW91.1 *	-400 (24)	670 (41)	270 (23)		
RW91.3	-872 (35)	576 (26)	-295 (18)		
RW91.5	-901 (27)	411 (18)	-490 (26)		
RW91.7 *	-943 (28)	672 (36)	-272 (17)		
RW91.9	-574 (17)	605 (32)	31 (2)		
RW91.11	-862 (26)	548 (24)	-315 (17)		
RW91.13	-910 (27)	592 (27)	-317 (17)		
Site average	-823 (55)	546 (35)	-277 (85)	20.0	-3.50
Rewetted grassland (RW-92)					
RW92.1	-709 (43)	339 (12)	-370 (26)		
RW92.3 *	-857 (26)	1148 (37)	291 (13)		
RW92.5 *	-868 (17)	983 (23)	115 (4)		
RW92.7 *	-1219 (24)	962 (25)	-257 (8)		
RW92.9 *	-667 (27)	605 (27)	-62 (4)		
RW92.11 *	-792 (16)	592 (19)	-200 (8)		
RW92.13 *	-834 (17)	1176 (41)	342 (14)		
Site average	-709 (43)	339 (12)	-370 (26)	23.3	-5.78

4.3.4. Ecosystem respiration

4.3.4.1. R_{ECO} flux drivers

R_{ECO} fluxes showed weak to moderate correlations with volumetric moisture content (Table 4.5), with a significant increase in R_{ECO} observed as soil moisture decreased in the shallow-drained grassland at all collars except SD.1 and SD.13. A similar relationship was observed in rewetted grassland RW-92, however this was only significant at two collars. The correlation between R_{ECO} and VMC was more varied in site RW-91, with two collars showing a significant increase in R_{ECO} as soil moisture increased (RW91.1 and RW91.11), and two collars showing a significant negative correlation (Table 4.5).

Table 4.5: Kendall tau correlation coefficients for R_{ECO} and soil parameters; 5 cm soil temperature, and VMC. Non significant results ($p > 0.05$) shown by [NS]

Collar ID	R_{ECO} Soil temp	R_{ECO} VMC
Shallow-drained grassland		
SD.1	0.291	0.001 [NS]
SD.3	0.451	-0.074
SD.5	0.496	-0.220
SD.7	0.508	-0.159
SD.9	0.419	-0.194
SD.11	0.528	-0.124
SD.13	0.451	0.120
Rewetted grassland (RW-91)		
RW91.1	0.377	0.367
RW91.3	0.427	-0.235
RW91.5	0.437	-0.417
RW91.7	0.193	0.153 [NS]
RW91.9	0.494	-0.103 [NS]
RW91.11	0.49	0.259
RW91.13	0.487	-0.03 [NS]
Rewetted grassland (RW-92)		

RW92.1	0.578	-0.117 [NS]
RW92.3	-0.104[NS]	-0.173
RW92.5	0.269	-0.031 [NS]
RW92.7	0.123	-0.072 [NS]
RW92.9	-0.054 [NS]	-0.101[NS]
RW92.11	0.225	0.014 [NS]
RW92.13	-0.244	-0.303

R_{ECO} fluxes generally showed a significant positive correlation with soil temperature across the study sites (Table 4.5), however the relationship was weaker in rewetted grassland RW-92 where a significant negative correlation was also found (collar RW92.13). The inundation at the rewetted grassland sites meant that R_{ECO} fluxes were measured over a smaller soil temperature range (10 – 21.4 °C) compared to the shallow-drained grassland (3.1 to 26.7°C). The maximum soil temperatures observed at the shallow-drained site were higher than those in the rewetted site, though both were recorded during the same measurement campaign (June 2016). Consequently, the relationship between R_{ECO} and soil temperature was compared across the same temperature range by binning soil temperature in 1 °C classes (Fig. 4.11). The average (median) R_{ECO} flux was similar within all three sites at soil temperatures from 11 to 15 °C, however as soil temperatures increased from 16 to 20 °C, the increase in R_{ECO} was greater at the shallow-drained site compared to the rewetted sites. Furthermore, the increase in R_{ECO} with soil temperature was slightly greater at the rewetted grassland RW-91 compared to RW-92. Maximum R_{ECO} fluxes were observed at a soil temperature of approximately 19 °C, with fluxes starting to decline above 21 ° in the shallow-drained site. Due to the lower maximum soil temperatures at the rewetted sites, it could not be determined whether R_{ECO} fluxes follow the same pattern. The variability of R_{ECO} fluxes within each soil temperature class was considerably larger in the shallow-drained grassland compared to the rewetted sites (Fig. 4.11), with R_{ECO} fluxes ranging from 0 to 4000 $mgCO_2$ m^{-2} hr^{-1} at all soil temperature classes from 17 to 21 °C.
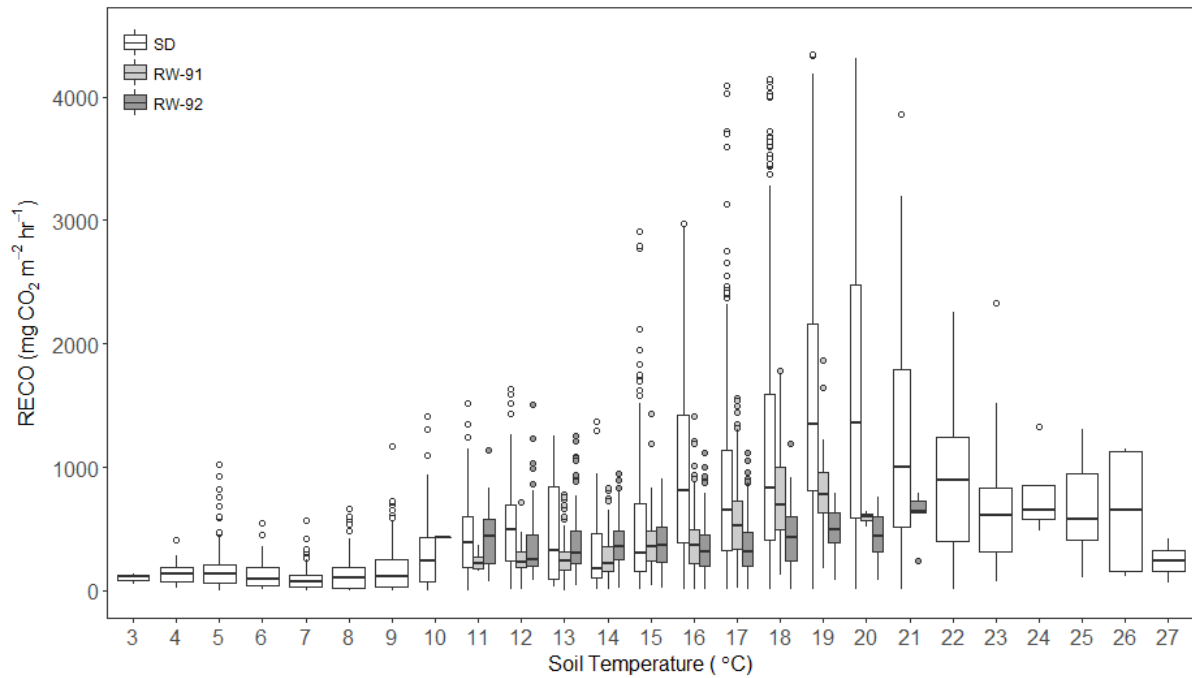
Figure 4.11. Boxplot showing the relationship between observed R_{ECO} fluxes and soil temperature at 5 cm depth during the entire study period. Soil temperature was binned into 1°C classes.

The relationship between R_{ECO} and soil temperature during the day and night was also compared by binning soil temperature in 1 °C classes (Fig. 4.12). Little difference was observed in nocturnal measurements between the opaque and transparent chambers (Appendix C, Fig. C3), indicating there was no bias in nocturnal measurements due to chamber type. In the shallow-drained site, the increase in R_{ECO} with soil temperature during the day and night was similar from 3 to 17 °C, after which the average nocturnal R_{ECO} flux was generally higher compared to the same temperatures during the day. The maximum nocturnal soil temperature was 20 °C, whereas during the day soil temperatures up to 27 °C were recorded, consequently it could not be determined whether R_{ECO} fluxes declined at higher soil temperatures as observed during the day time. The range of R_{ECO} fluxes observed in each soil temperature class were similar during the day and night in all three sites. The increase in R_{ECO} with soil temperature at the rewetted sites was similar during the day and night, though the average R_{ECO} flux at 17 and 18 °C was slightly higher at night in RW-91 (Fig. 4.12). Maximum soil temperatures were again found during the daytime at the rewetted sites, however the difference between daytime and night time temperatures was smaller than the difference observed in the shallow-drained site.
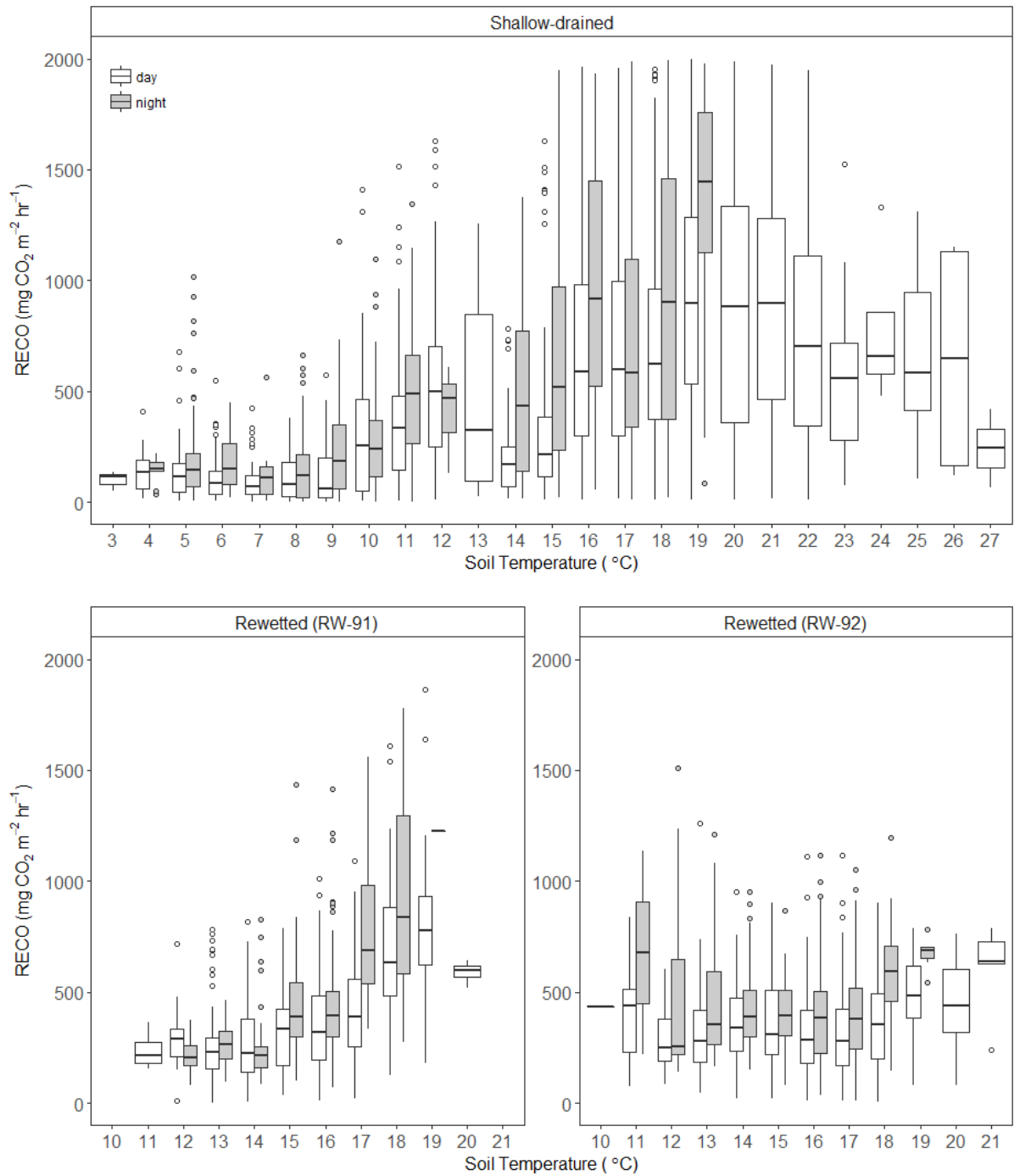
Figure 4.12. Boxplots showing the day time and night time relationship between observed R_{ECO} fluxes and soil temperature for the entire study period. Soil temperature was binned into 1 °C classes.

4.3.4.2. R_{ECO} model performance

The R_{ECO} model fits using soil temperature are shown in Fig. 4.13. Model performance was again rated using the criteria defined in Hoffmann et al. (2015), with the shallow-drained site models mostly satisfactory (r^2 0.26 to 0.44, Table 4.3), except for collars SD.1 and SD.9 where unsatisfactory model fits were obtained (r^2 of 0.18 and 0.15 respectively). The relatively low r^2 values obtained for models in the shallow-drained site are likely due to the ineffectiveness of the Arrhenius type model at capturing the high variability in R_{ECO} across small soil temperature ranges (Fig. 4.13a). For example, R_{ECO} fluxes measured in the shallow-drained grassland with soil temperatures 17.5 to 18.5 C ranged from 17 to 4138 $mgCO_2$ m^{-2} hr^{-1}. Consequently, the models underestimated high R_{ECO} fluxes which is particularly evident in SD.1, and SD.7 to SD.13 (Fig. 4.13a and Appendix C, Fig. C4). Although the exponential shape of the model fails to capture the decline in CO_2 fluxes at soil temperatures above 21 °C (Fig. 4.13), the low slope of the curve means that the R_{ECO} fluxes at these temperatures are only slightly overestimated.
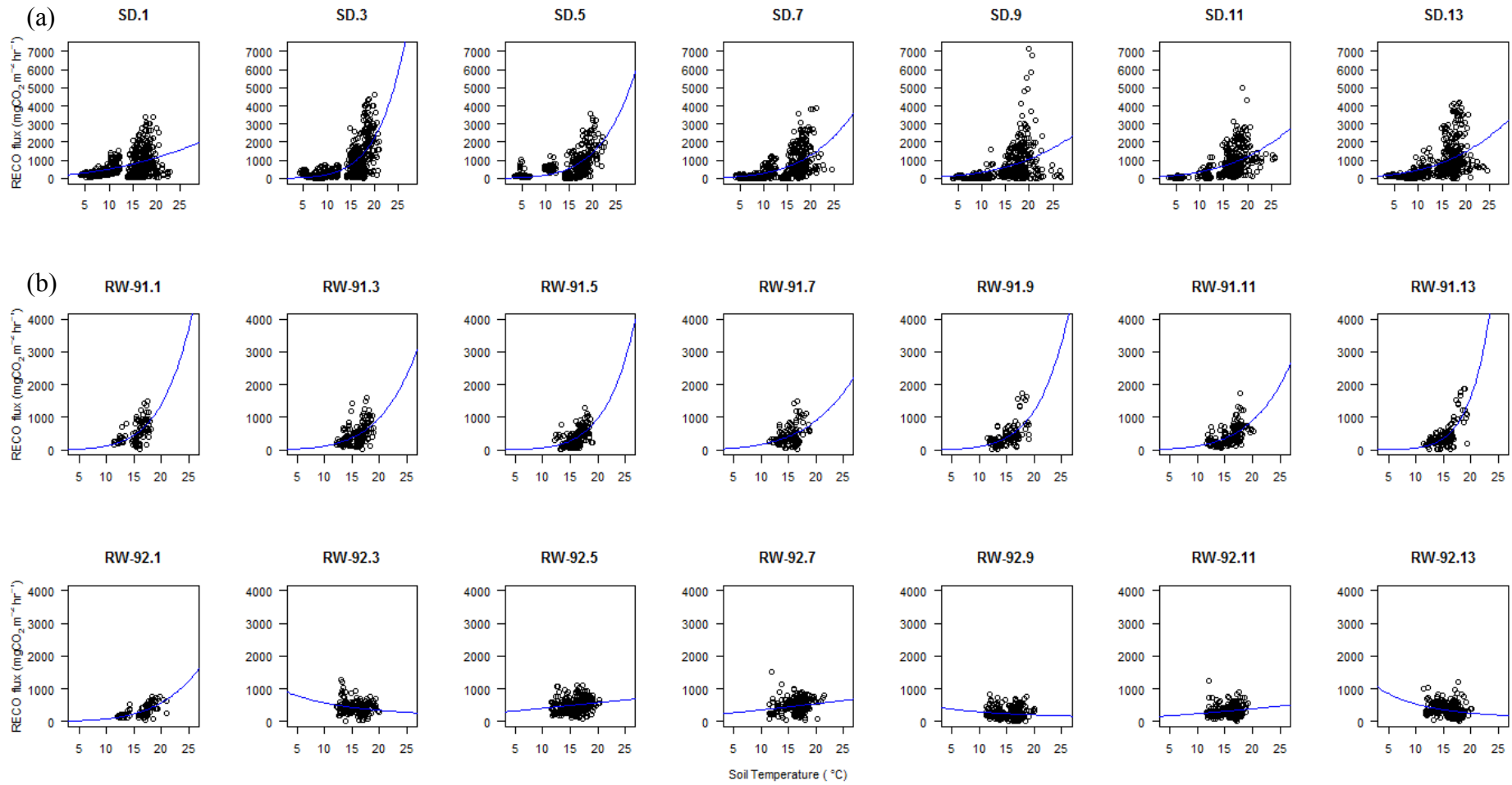
Figure 4.13. Relationship between R_{ECO} fluxes and soil temperature at 5 cm depth for individual measurement collars using data collected over the entire study period for (a) the shallow-drained; and (b) the rewetted grasslands. The Lloyd and Taylor model fitted for each collar is represented by the blue line.

R_{ECO} model fits in the rewetted site RW-91 ranged from satisfactory to good (r^2 0.26 to 0.64, Table 4.3), with the exception of RW91.7 where an unsatisfactory model fit was obtained (r^2 of 0.16). The models for RW91 tended to underestimate the highest R_{ECO} fluxes, however there was generally good agreement between the measured and modelled fluxes (Fig. C12). The R_{ECO} models at the rewetted site RW-92 performed poorly, with all collars except RW92.1 having unsatisfactory model fits (Table 4.3, due to the lack of R_{ECO} response to soil temperature (Fig. 4.12 and Fig. 4.13b). The R_{ECO} fluxes at collars RW92.3 to RW92.13 showed weak correlations with soil temperature, with high variability in fluxes recorded at soil temperature 11 to 13 °C. These soil temperatures were observed in May 2016 and September 2015, with R_{ECO} fluxes consistently higher and also more varied in May compared to those observed in September, possibly due to the senescence of vegetation beginning in September and the higher water tables. The lack of measurements at collar RW92.1 during May (due to equipment limitations) meant this was the only collar in rewetted RW-92 to achieve a satisfactory R_{ECO} model fit.

4.3.4.3. Annual R_{ECO} fluxes

The modelled annual R_{ECO} fluxes from the collars in the two rewetted grasslands were combined to produce an average annual R_{ECO} flux under rewetted conditions, rather than site averages, due to all except one of the R_{ECO} models being unsatisfactory in RW-92. Annual modelled R_{ECO} was highest in the shallow-drained grassland, with an average CO_2 emission of 1165 gC m^{-2} a^{-1}, whilst the average emission from the rewetted grasslands was 534 gC m^{-2} a^{-1} (NB, site averages were calculating using only collars were model fits were rated as satisfactory). There was a greater variability in annual R_{ECO} fluxes within the shallow-drained site compared to the rewetted grasslands (Table 4.4). Within the rewetted grassland RW-92, the highest fluxes were observed where a negative correlation between R_{ECO} and soil temperature was found (RW92.3 and RW92.13). Consequently, these models will have considerably overestimated R_{ECO} fluxes at lower soil temperatures, particularly given the waterlogged nature of the site, thus these annual CO_2 emissions (1148 and 1176 gC m^{-2} a^{-1} for RW92.3 and RW92.13 respectively) are highly unlikely. The lack of measurements during May at RW92.1 means that the annual R_{ECO} flux estimate for this collar likely represents the lower end of annual flux values from rewetted grasslands; it was evident from the other collars in RW92 that higher fluxes were observed earlier in the season which would not have been captured by the RW92.1 model (Fig. 4.13). This is supported by the slightly higher annual R_{ECO} estimates from site RW91.

4.3.5. Annual CO_2 balance

Net ecosystem exchange differed significantly between the shallow-drained and rewetted grasslands, with the former acting as a small net CO_2 source on average (23 ± 100 gC m^{-2} a^{-1}) whilst the rewetted grasslands were on average a net sink (-290 ± 60 gC m^{-2} a^{-1}). Annual NEE fluxes are shown in Fig. 4.14 for collars where GPP and R_{ECO} model fits were at least satisfactory (annual NEE for all collars is shown in Appendix C, Fig.C5). Annual NEE varied amongst collars within the shallow-drained site, with SD.5 acting as an apparent large net CO_2 sink, whilst the remaining collars all acted as small to moderate net CO_2 sources (Fig. 4.14). The net CO_2 sink effect was broadly consistent across the rewetted grasslands where satisfactory GPP and R_{ECO} model fits were achieved, with the exception of RW91.9 which acted as a small net CO_2 source (Fig 4.14). The remaining collars which acted as net CO_2 sources (Fig. C5) were associated with unsatisfactory R_{ECO} model fits (RW92, collars 3, 5 and 13) and restricted GPP measurements which limited modelling ability at the start of the growing season (RW91.1).
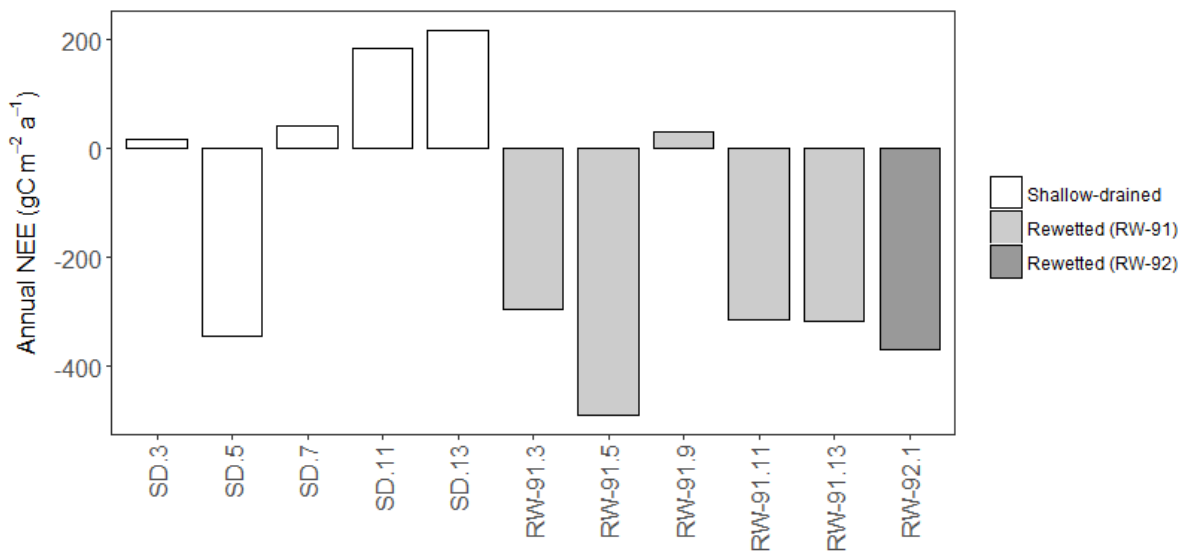
Figure 4.14. Modelled annual NEE flux values for each collar where at least a satisfactory model fit was achieved for both R_{ECO} and GPP modelling.

NEE varied seasonally across all sites, with peak CO_2 sequestration occurring in June and peak emissions in July, however the length of the net uptake period differed among sites (Fig. 4.15). The shallow-drained site acted as a net sink of CO_2 from April to July and briefly in September/October, with June being the only month where NEE was consistently negative across all collars. The rewetted site RW-92 was a CO_2 sink for the majority of the year, acting as a small CO_2 source during the December, whilst the rewetted site RW-91 was a net source

from September until January. Maximum CO_2 uptake and emissions were greatest in the shallow-drained site, although the mean NEE was similar across the three sites from April to July.
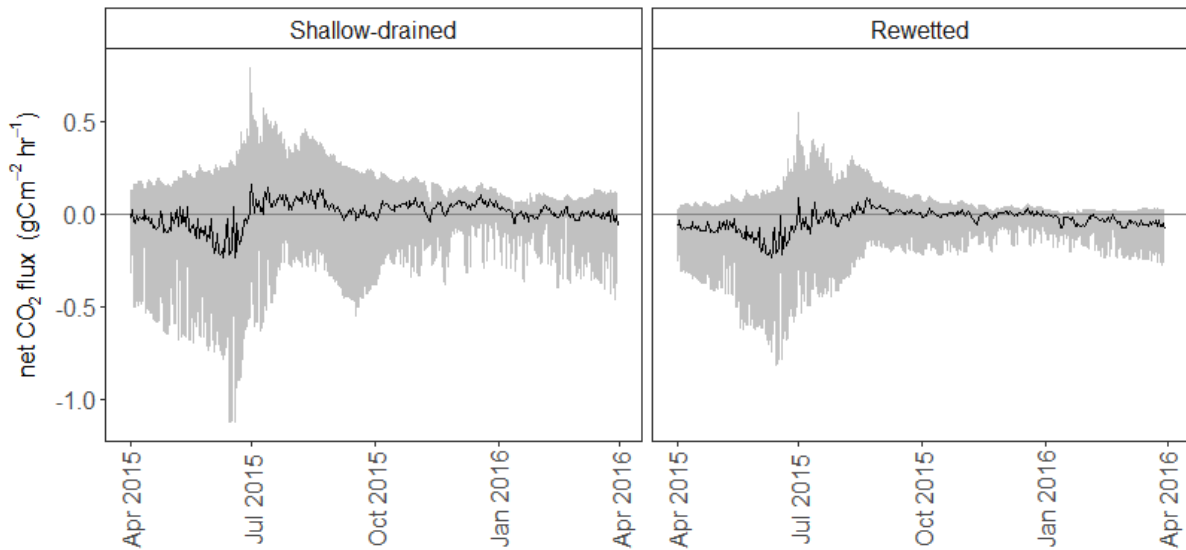
Figure 4.15. Modelled net ecosystem exchange during 1 April 2015 – 31 March 2016. The black line represents mean daily NEE, with the shaded area representing the minimum NEE (i.e. maximum CO_2 uptake) and maximum NEE (i.e. maximum CO_2 emission).

4.3.6. CH$_4$ Fluxes

4.3.6.1. CH$_4$ flux drivers

CH_4 fluxes from the shallow-drained grassland were low throughout the study period, ranging from -0.5 to 1.6 mgCH$_4$ m^{-2} hr^{-1} (Fig. 4.16), with a significant negative correlation between CH_4 fluxes and soil temperature (Table 4.6). CH_4 fluxes at the rewetted grassland showed significant correlations with soil temperature (Table 4.6), with an increase in CH_4 fluxes observed as temperatures increased. Peak emissions were observed in both rewetted sites at soil temperatures of ~ 16°C (Fig. 4.17), above which CH_4 fluxes decreased. The limited equipment deployment at the rewetted grasslands meant a seasonal pattern could not be identified from the measured fluxes, though the highest fluxes were observed during June 2016 coinciding with the highest soil temperatures (Fig 4.16). Although a decrease in CH_4 emissions was observed with increasing soil moisture, this result is treated with caution due to the unreliability of the soil moisture sensors mentioned in the methods section. Diurnal patterns in CH_4 fluxes could not be determined due to measurements not covering the full 24 hour cycle.

This was partially due to not being able to meet the power demands of the Los Gatos UGGA and partly due to equipment failure.

Table 4.6. Kendall tau correlation coefficients for measured environmental variables and CH_4 fluxes. Non-significant correlations are denoted by [NS].

Site	Soil Temperature	VMC	PAR
Shallow-drained	-0.193	0.073	-0.013[NS]
Rewetted (RW-92)	0.213	-0.142	0.073[NS]
Rewetted (RW-91)	0.414	0.021[NS]	0.052 [NS]
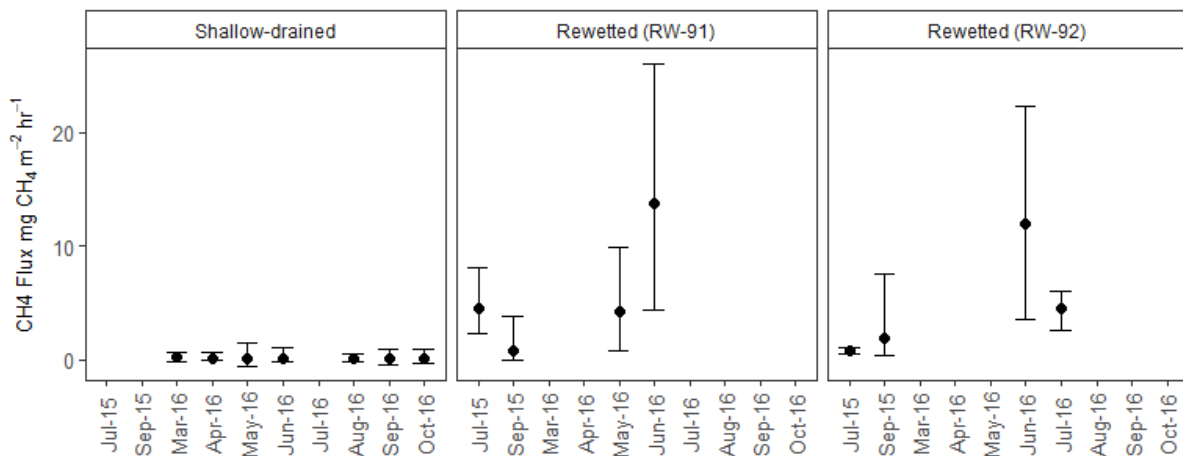
Figure 4.16. Average measured CH_4 fluxes during each sampling campaign. Error bars represent minimum and maximum measured fluxes.

4.3.6.2. CH_4 model performance

The exponential regression between CH_4 flux and 5cm soil temperature provided a satisfactory fit at rewetted site RW-92, with an r^2 of measured v modelled fluxes of 0.33 (Fig 4.17). Although the model fits at the other rewetted site (RW-91) and shallow-drained sites were unsatisfactory (r^2 of 0.20 and 0.02 respectively), significant correlations were found between soil temperature and CH_4 flux at both sites (Table 4.6) and therefore the models were retained.
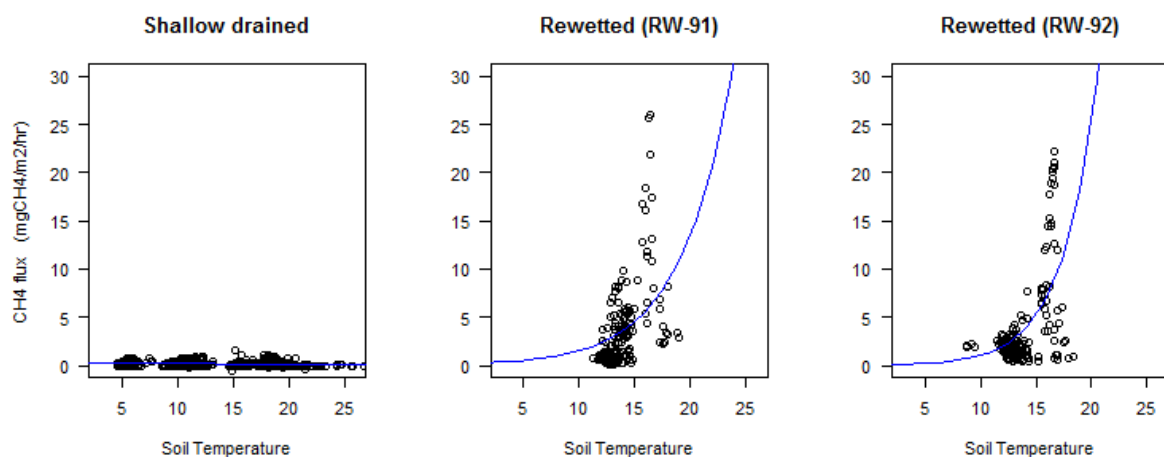
Figure 4.17. Relationship between CH₄ fluxes and soil temperature at 5 cm depth. The solid line represents the Q10 model fit.

4.3.6.3. Annual CH₄ fluxes

The shallow-drained site acted as a small net source of CH$_4$ (1.59 gCH$_4$ m^{-2} a^{-1}) whilst the rewetted sites acted as significantly larger sources (26.7 and 31.1 gCH$_4$ m^{-2} a^{-1} for RW-91 and RW-92 respectively). Modelled CH$_4$ fluxes from the shallow-drained site showed a seasonal pattern with the smallest fluxes measured from May – August, following the seasonal change in soil temperature (Fig. 4.18). CH$_4$ fluxes from the rewetted sites showed the opposite annual trend, with modelled fluxes peaking in June and July, with low emissions over winter.
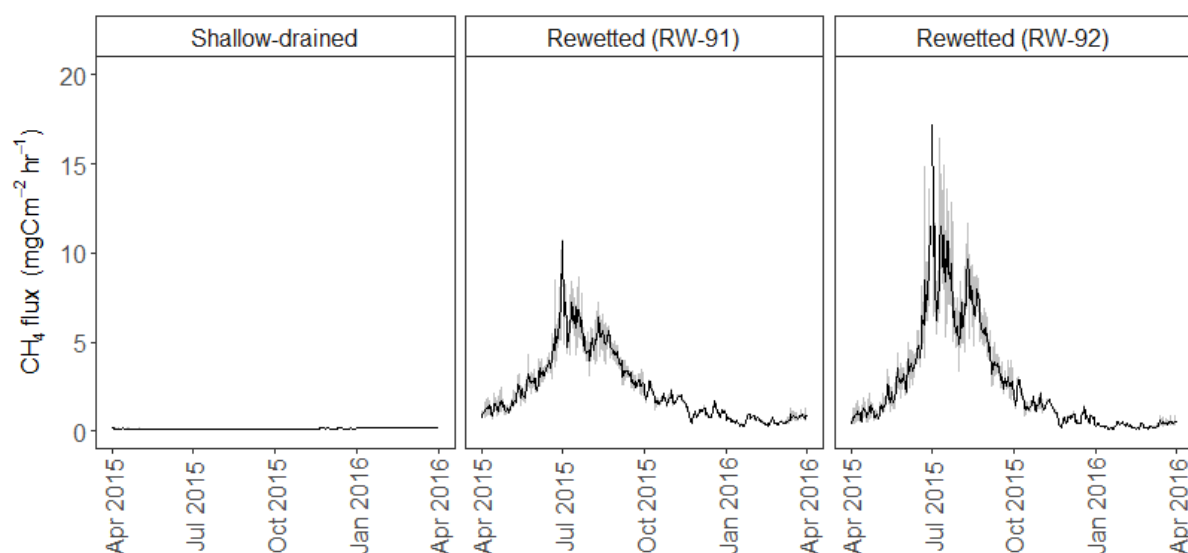
Figure 4.18. Modelled CH₄ fluxes during 1 April 2015 – 31 March 2016. The black line represents mean daily CH₄, with the shaded area representing the minimum and maximum CH₄ fluxes.

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4.3.7. Net GHG balance

The net GHG balance of the shallow-drained grassland was positive; 1.25 ± 3.7 tCO$_2$-eq ha^{-1} a^{-1}, indicating the site had a climatic warming effect. In contrast the rewetted grasslands both had a climatic cooling effect, with an average GHG balance of -3.14 ± 2.6 tCO$_2$-eq ha^{-1} a^{-1} (Fig. 4.19). The treatment averages were calculated using collars for which satisfactory R_{ECO} and GPP models were found (Appendix C, Fig. C6). Although the rewetted grasslands were strong net CO$_2$ sinks, the higher radiative forcing of CH$_4$ meant that just over half of this CO$_2$ uptake was counterbalanced by the annual CH$_4$ balance in CO$_2$-equivalents. Similarly, in the shallow-drained grassland annual CH$_4$ emissions in CO$_2$-equivalents amounted to approximately half of the CO$_2$ balance, though the shallow-drained grassland was a source of both CO$_2$ and CH$_4$.
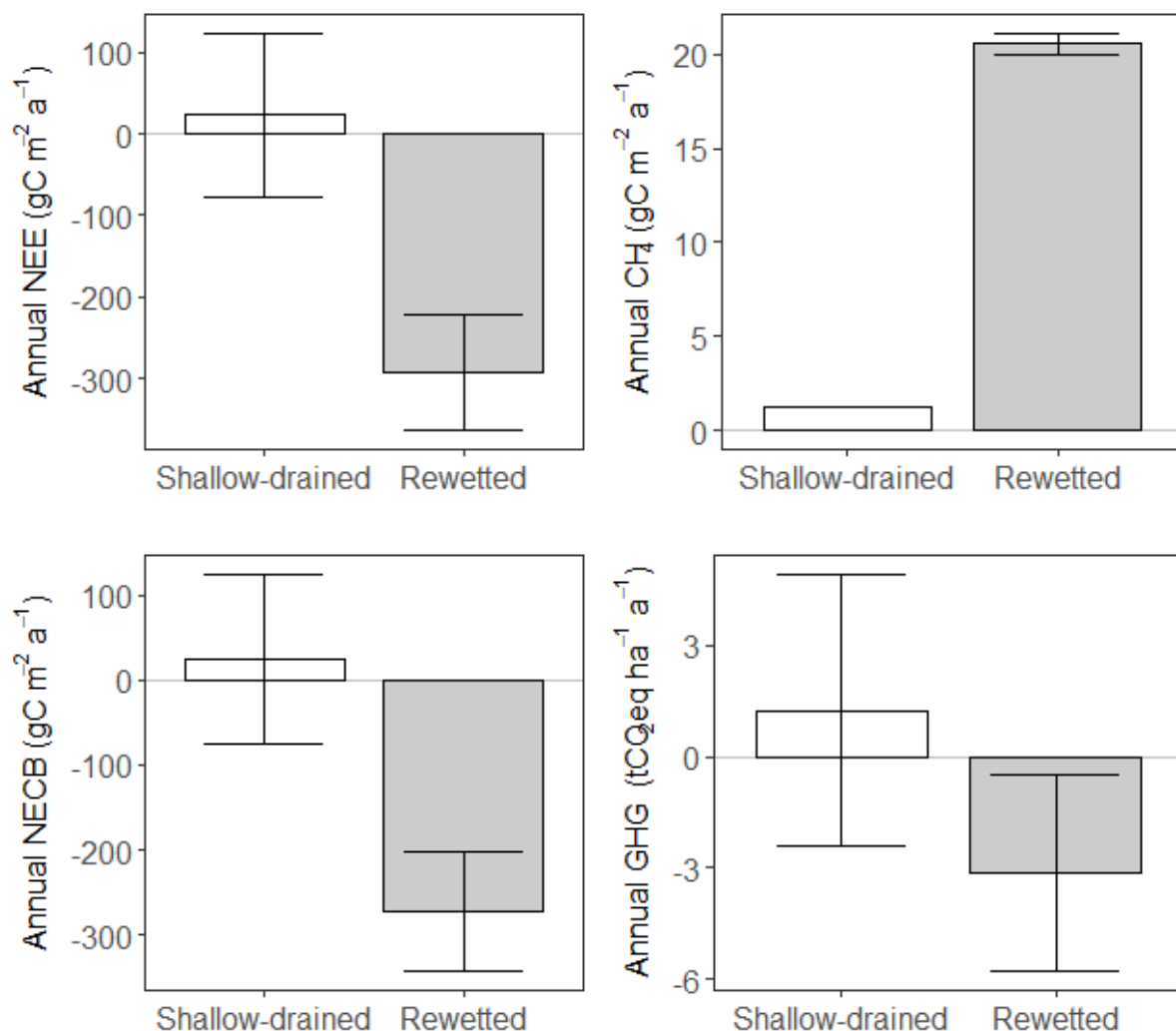
Figure 4.19. Average annual NEE, CH$_4$, NECB and GHG balance for the shallow-drained and rewetted grasslands. Error bars shown standard error. CO$_2$ flux values are those calculated

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from collars where satisfactory R_{ECO} and GPP models were found. Radiative forcing of 25 was used for converting CH_4 to CO_2-equivalents.

4.4. Discussion

4.4.1. Automatic chamber system operation

GHG flux measurements with automatic chambers enable high frequency measurements of gas fluxes and environmental parameters which are limited by the labour efforts required in manual chamber measurement campaigns. However, the more complex technical requirements means that operating automatic chamber systems can also come with considerable challenges. The automatic chamber system worked well for the majority of the study period, but data gaps existed for a number of reasons. A total of 2.3 % of potential CO_2 flux measurements using the LI-8100A system were not achieved due to equipment failure, predominately caused by a lack of power. The loss of CH_4 (etc.) measurements due to equipment failure was much greater, with only 21 % of potential measurements achieved using the Los Gatos UGGA. A high proportion of CH_4 measurements were lost due to a lack of power; the Los Gatos analyser ran continuously and thus had significantly greater power demands compared to the LI-8100A system, which was able to shut down between chamber measurements to conserve power. The reliance on batteries and solar panels to power the system was unavoidable, however due to the high cloud cover experienced on the west coast of the UK maintaining a constant power supply was challenging. Additional CH_4 measurements were lost due to poor weather; the Los Gatos analyser is not weather proof and consequently was taken out of the field during heavy rainfall events to reduce the risk of equipment failure. Major data gaps in this study arose from the inundated conditions at the rewetted grasslands which limited the deployment of the system due to the chambers and analysers sitting at the ground level. Although no studies of automatic chambers being used at peatlands where the water table is above the surface could be found, systems have been found to successfully operate under snowpack conditions (e.g. Koskinen et al., 2014). These studies have, however, used custom made larger chambers and been set as permanent experiment plots enabling the development of infrastructure to cope with a range of environmental conditions which was not possible at Cors Fochno due to the presence of livestock and short term nature of the study.

Whilst chamber measurements enable small scale spatial heterogeneity to be captured, automatic chambers limit the area over which this heterogeneity can be assessed due to the

physical limitations of the system. The LI-8100A system used in this study had 15 m long gas tubing, enabling the system to cover an area of ~ 30 m diameter. Although this coverage was sufficient to capture the range of vegetation found within each site, the chambers were too close together to capture the full spatial variability in water table depth. In order to capture field scale heterogeneity, such as water table gradients, it would be necessary to use manual chambers over a larger spatial scale or deploy automatic chambers at a number of locations within each site. The LI-8100A chambers also limited measurements to small stature vegetation due to the chamber height of ~ 30 cm. Furthermore, the closing mechanism of the chamber meant that as the chamber moved from the open to closed position the base of the chamber was only ~ 10 cm above the baseplate. Thus vegetation had to be low enough to ensure that the chamber movement was unrestricted and the chamber could lower and seal onto the baseplate without vegetation being trapped. Whilst the LI-8100A chambers were suitable for the majority of vegetation within the shallow-drained and rewetted grasslands in this study, patches of taller vegetation (e.g. rushes, sedges) were omitted from measurements, which could influence the calculated carbon/GHG balance (notably the estimated CH_4 flux) when considering the wider study area.

4.4.2. Model evaluation

4.4.2.1. Gross primary productivity

Gross primary productivity is controlled by incident photosynthetically active radiation and changes throughout the diurnal cycle, as well as seasonally. The use of automated chambers enables the diurnal response of CO_2 uptake to PAR to be captured, which is seldom achieved with manual chambers, though it may to some extent be replicated through manual shading (e.g. Elsgaard et al., 2012; Gatis et al., 2015). In this study, GPP fluxes were measured across a wide range of light levels during most measurement campaigns, enabling specific GPP models to be fitted for the different stages of seasonal vegetation development. The GPP modelling performed well in this study, though limited PAR ranges occasionally lead to unsatisfactory model fits, normally near the start and the end of the growing season when vegetation is either still developing or has started to senesce. The interpolation of model parameters between measurement dates in GPP modelling is widely used throughout the literature (e.g. Beetz et al., 2013) though it can lead to overestimation of CO_2 uptake. This was a particular issue in the rewetted grasslands where inundation only allowed measurements to be carried out from May to September, and therefore models may artificially increase CO_2

uptake over the winter. For example, collar RW92.7 had a consistently greater ecosystem assimilation capacity (GPmax parameter) compared to the other collars in RW-92, partly due to the steep light response curve in June. This led to RW92.7 having a modelled annual GPP of -1219 gC m^{-2} a^{-1}, whereas GPP estimates for the other collars in the site ranged from -667 to -868 gC m^{-2} a^{-1}. Although GPP model fits for RW92.7 were good (r^2 0.74 to 0.97), measurements were limited to May, June and July, when productivity is at its highest, and the lack of measurements at the end of the growing season likely results in an overestimation of CO_2 uptake. Although automated chamber measurements could not be performed during inundation, carrying out manual chamber measurements during this period would be a possibility to overcome missing measurements and further improve the accuracy of annual GPP estimates.

4.4.2.2. Ecosystem respiration

Whilst GPP modelling can be significantly improved by capturing diurnal changes in environmental drivers, R_{ECO} is predominately driven by seasonal cycles, including temperature control on microbial activity and variations in plant productivity affecting substrate availability for autotrophic respiration and heterotrophic respiration through root exudates (Davidson et al., 2006). Consequently, successful R_{ECO} modelling relies upon capturing a wide range of temperatures, i.e. measurements covering as much of the year as possible. Within the shallow-drained grassland R_{ECO} flux measurements covered soil temperatures ranging from 3 to 26 °C, however the relationship between R_{ECO} and soil temperature was noisy, reducing the accuracy of R_{ECO} models. The relationship between R_{ECO} and soil temperature also varied between daytime and night time measurements with higher fluxes observed at night compared to corresponding temperatures during the daytime. Higher nocturnal R_{ECO} fluxes have been widely reported from automatic chamber studies where atmospheric turbulence has been low (Schneider et al., 2009; Lai et al., 2012; Koskinen et al., 2014). Under calm conditions (e.g. friction velocity < 0.1 ms^{-1}) CO_2 emissions accumulate at the peat surface, resulting in a surface atmospheric layer with a steep concentration gradient. The high CO_2 concentration at the surface therefore reduces the CO_2 gradient between the soil pores and the atmosphere, reducing the rate of CO_2 diffusion. As the chamber moves, this stratified layer of air is disturbed and the CO_2 concentration at the surface decreases, resulting in a steep CO_2 gradient between the soil and the atmosphere. Consequently the diffusive flux rapidly increases (Schneider et al., 2009). An immediate increase in CO_2 concentration following chamber closure has also been

attributed to fans within the chamber, mixing the headspace and increasing the diffusion of CO_2 from the soil (Schneider et al., 2009; Koskinen et al., 2014; Görres et al., 2016). The discrepancy between daytime and night time R_{ECO} fluxes was most evident in the shallow-drained grassland in June 2016. The lack of weather station data means it could not be determined whether stable atmospheric conditions persisted, however the location of Cors Fochno on the west coast of the UK makes it highly unlikely that atmospheric turbulence was sufficiently low to allow this effect to operate on a consistent basis. Furthermore, although the average R_{ECO} flux was higher at night, the range of R_{ECO} fluxes observed at corresponding soil temperatures was similar for daytime and night time observations. The inclusion of water table or soil moisture in R_{ECO} modelling has been shown to improve model fits (e.g Tuittila et al., 1999; Samartini et al., 2011), however the due to the unreliability of VMC sensors, it was not possible to test this for the current study.

Inundation at the rewetted grasslands resulted in a lack of measurements at low soil temperatures, narrowing the range of observations and contributing to the relatively poor R_{ECO} model fits at these sites. The effect of this was particularly evident at collars RW92.3 and RW92.13 where high R_{ECO} fluxes measured in May at soil temperatures ~11 °C, resulted in a negative correlation between R_{ECO} and soil temperature, and therefore the models being discarded. Measurement campaigns throughout the summer meant that R_{ECO} fluxes were measured at soil temperatures of up to 22 °C, which was also the maximum soil temperature of the continuous weather station data used for modelling annual fluxes. Thus it was not required for R_{ECO} fluxes to be modelled at soil temperatures above the measured range, which can be a large source of uncertainty with non-linear R_{ECO} models, as R_{ECO} becomes limited at higher soil temperatures, for example by soil moisture availability. R_{ECO} fluxes were modelled at soil temperatures lower than the measured minimum (11 °C), and therefore model fits could have been improved by conducting manual chamber measurements during the winter when inundation precluded autochamber deployment. Despite this, the low fluxes associated with low temperatures results in relatively small uncertainty when modelling beyond the measured temperature range.

4.4.2.3. CH$_4$

The use of automatic chambers in CH$_4$ flux measurements significantly decreases the likelihood of user induced ebullition events which can occur during the placement of manual

chambers. Furthermore, the high variability in CH_4 fluxes, particularly those related to ebullition, means that the relatively infrequent manual chamber measurements can easily over or underestimate fluxes (Juszczak and Augustin, 2013). The relationship between CH_4 and soil temperature was similar in the rewetted grasslands, with peak emissions between 15 and 17 C, agreeing with the widely found control of soil temperature on methanogenic activity (Jordan et al., 2016). Although the model uncertainty was relatively high, similar r^2 values have been reported by Hendriks et al. (2007) in modelling CH_4 fluxes from an abandoned peat meadow (r^2 0.086 to 0.457). Within the Defra Lowland Peat Project, reliable CH_4 models were only found within arable peatlands where CH_4 uptake closely following seasonal temperature variations, with no sites under semi-natural or grassland vegetation producing robust models (Evans et al., 2017). The poor CH_4 model performance in the shallow-drained grassland is consistent with models from drained peatlands, where low water tables result in very small emissions or uptake of CH_4 (Renou-Wilson et al., 2014; Scrhrier-Uijl et al., 2010). Overestimation of CH_4 fluxes has been associated with low atmospheric turbulence, similar to that observed with R_{ECO} fluxes (Lai et al., 2012), however in this study the lack of nocturnal CH_4 flux data meant it could not be determined whether this occurred.

4.4.3. Annual C balance: The effect of rewetting grasslands on peat

During the study year, the shallow-drained grassland acted as a net source of C, whilst the rewetted grasslands acted as net C sinks. The seasonal pattern of NEE at all study sites was characteristic of grasslands on peat with permanent vegetation and low intensity management, with small typically positive fluxes in winter and large (either positive or negative) fluxes in summer. CO_2 uptake was greatest early in the growing season, reaching its peak in June, whilst R_{ECO} peaked slightly later in the year in July. GPP was consistently greater (more negative) under shallow-drained conditions compared to the rewetted grassland, however there was a larger difference in R_{ECO} than in GPP between the sites. The higher water table at the rewetted grassland reduces aerobic decomposition of organic matter and thus CO_2 produced and released from ecosystem respiration. In contrast at the shallow-drained site, the increased aeration of the surface peat layer leads to greater ecosystem respiration and thus higher R_{ECO} fluxes are observed. The NEE values reported from the shallow-drained site in this study are at the lower end of estimates reported in the literature from temperate grasslands, and given the tendency for the R_{ECO} models to underestimate higher fluxes, are likely under-estimates. Comparison with CO_2 emissions from grasslands on peat across Germany collated by Tiemeyer et al. (2016)

show the NEE estimates from this study to be considerably smaller than the values reported for nutrient poor grasslands (350 gC m^{-2}a^{-1}) and bog peat (290 gC m^{-2}a^{-1}), though the synthesis by Tiemeyer et al. (2016) makes no differentiation between management intensity and style, which could have considerable effects on R$_{ECO}$ for example if fertilisation occurs, or on GPP if harvesting occurs. GPP and R_{ECO} fluxes from this study do however compare well to values from temperate grasslands on peatlands under extensive management (e.g. Beetz et al., 2013; Renou-Wilson et al., 2014; Renou-Wilson et al., 2016). The situation of Cors Fochno on the west coast of Wales closely compares to study sites reported by Renou-Wilson et al. (2014; 2016) on the north-west coast of Ireland, with both experiencing maritime temperate climates and shallow-drained nutrient poor organic soils. Annual NEE values reported from the Irish site range from a sink of -99 ±68 gC m^{-2}a^{-1} (Renou-Wilson et al., 2014) to a source of 90 ±30 gC m^{-2}a^{-1} (Renou-Wilson et al., 2016), with the NEE of the shallow-drained site here, 23 gC m^{-2}a^{-1}, well within this range. The extensively managed grassland sites in the Defra Lowland Peat Project also acted as net sources of CO_2, with an average NEE of 95 gC m^{-2}a^{-1} and 123 gC m^{-2}a^{-1} for sites in the Somerset Levels and East Anglia respectively (Evans et al., 2017). Although these values are slightly higher than those reported from this study and for maritime temperate grasslands (e.g Beetz et al., 2013; Renou-Wilson et al., 2016), this may be due to low water tables during the summer aerating the entire peat profile (Bakers Fen, East Anglia) as well as hay-cutting in the Somerset Levels site once a year during the summer resulting in a short period of high R_{ECO} fluxes even during the day.

Whilst the annual NEE flux of the shallow-drained site compared fairly well with the literature, the magnitude of GPP and R$_{ECO}$ fluxes from the rewetted sites are smaller than those reported for rewetted grasslands on peat (e.g. Hendriks et al., 2007; Renou-Wilson et al., 2016). The rewetted sites act as substantial net sinks of CO_2, with CO_2 fluxes from RW-92 in line with those reported from *Eriophorum* spp. vegetation at a rewetted cutover Irish bog by Wilson et al. (2016) where a five year mean annual NEE of -260 ± 179 gC m^{-2}a^{-1} was found. High net uptake of CO_2 was also reported by Hendriks et al. (2007) from an abandoned peat meadow in the central Netherlands where annual NEE fluxes ranged from -232 ± 58 to -446 ± 83 gC m^{-2}a^{-1}. The large CO_2 uptake in the study by Hendriks et al. (2007) was attributed to the temperate climate and lack of management which allows full vegetation growth during the long growing season, and high water tables which inhibit the oxidation of soil organic matter and plant litter. The lack of winter/early spring measurements at the rewetted grassland in this study and subsequent interpolation of model parameters could lead to the overestimation of ecosystem

assimilation over the winter. In an attempt to reduce this affect, the model parameters derived at the end of the growing season (September) were held constant throughout the winter until 1 March. GPP fluxes were low throughout the winter, however the modelled daily CO_2 balance remained negative, suggesting that the annual modelled GPP may be slightly overestimated. The high water tables at the rewetted grasslands would limit R_{ECO} and therefore the lower CO_2 emissions that were observed compared to the shallow-drained grassland are expected. However, the annual R_{ECO} fluxes measured here (339 to 670 gC $m^{-2}a^{-1}$) are notably lower than those reported in the literature for rewetted grasslands on peat which range from 866 gC $m^{-2}a^{-1}$ (Hendriks et al., 2007) to 1482 gC $m^{-2}a^{-1}$ (Renou-Wilson et al., 2016), being closer to those reported from semi-natural bogs by Beetz et al. (2013) and Sphagnum communities by Beyer and Hoper (2014). The lack of CO_2 measurements at low temperatures may have led to the models not accurately capturing R_{ECO} fluxes outside of the growing season, and thus fluxes are likely underestimated at low temperatures. Combined with the potential overestimation of GPP fluxes, the underestimation of R_{ECO} fluxes means that the NEE of the rewetted grasslands is likely to be at the less negative end of observed values.

CH_4 emissions from the rewetted sites (20 to 23 g C $m^{-2}a^{-1}$) were significantly higher than those from the shallow-drained site (1.2 g C $m^{-2}a^{-1}$). The high water table at the rewetted grassland creates anaerobic conditions resulting in the decomposition of organic matter by methanogenic communities which produces CH_4. Methanogenic activity is greatest at the water table interface and thus when this boundary is near to the surface, the potential for CH_4 oxidation is reduced and a greater proportion of CH_4 produced is released to the atmosphere. In contrast, the low water tables in the shallow-drained site means there is greater opportunity for methanotrophic processes to operate; oxidising CH_4 and so less is released to the atmosphere. The low annual CH_4 emissions from the shallow-drained grassland correspond well with values reported from the literature which range from -0.22 g C m^{-2} a^{-1} for (Evans et al., 2017; Bakers Fen grass vegetation) to 1.52 gC m^{-2} a^{-1} (Beetz et al., 2013). Similarly, the higher annual CH_4 fluxes from the rewetted grasslands in this study are within the range found in the literature from grasslands on peat; 9.21 (Renou-Wilson et al., 2016) to 104 ± 30.4 g C m^{-2} a^{-1} (Hendriks et al., 2007). In addition to high water tables, the vegetation present can have a significant impact on CH_4 fluxes, with plants possessing aerenchymatous tissue e.g. *Juncus* or *Eriophorum* species, being associated with higher CH_4 emissions (Tuittila et al., 2000; Cooper et al., 2014; Wilson et al., 2016). *J.effusus* stands occurred across the rewetted grasslands, including the measurement

collars, which may have also contributed to the higher CH_4 fluxes observed within the rewetted sites due to the CH_4 produced being able to bypass the aerated water column or soil layer.

4.4.4. Diurnal fluxes

The daily switch between CO_2 source and sink was similar in the shallow-drained and rewetted grasslands and followed a seasonal pattern. The maximum diurnal length of CO_2 sequestration occurred during June, when productivity and R_{ECO} also peaked, whilst the shortest uptake period was found in March and October. Although it was not possible to measure at the sites concurrently, comparison of measurements within a few days of each other showed the diurnal fluctuations at the shallow-drained grassland to be greater than those at the rewetted grassland. The difference in diurnal NEE fluctuations was driven by high nocturnal R_{ECO} at the shallow-drained grassland, with net CO_2 uptake similar across the sites. The lower water table (and soil moisture) at the shallow-drained grassland means there is a larger aerobic layer of soil in which decomposition of organic matter can occur and therefore CO_2 production and emissions are greater. The higher rates of R_{ECO} are particularly evident during June 2016 when nocturnal CO_2 fluxes from the shallow-drained grassland were 4-6 times greater than fluxes from the rewetted grassland. The nocturnal increase in CO_2 emissions was also evident in R_{ECO} measurements at the shallow-drained grassland. High nocturnal CO_2 emissions from chamber measurements under stable atmospheric conditions have been reported, though as discussed in section 4.4.1, the likelihood of these conditions on the west coast of Wales are low. Few studies have compared chamber derived R_{ECO} fluxes measured in the daytime and nighttime, which is often confounded by the influence of varying conditions (Juszczak et al., 2012). The concurrence of autotrophic and heterotrophic respiration during R_{ECO} flux measurements further complicates isolating the driving variables. Leaf respiration has been reported to be inhibited in light conditions, with reduction in respiration between 16 and 77 %, which could significantly contribute to the measurement of lower daytime R_{ECO} fluxes (Juszczak et al., 2012). On the other hand, delays have been observed between soil temperature changes and the response of heterotrophic respiration, with up to 0.57 hours per cm depth, in a *Sphagnum* dominated peatland by D'angelo et al. (2016), attributed to the lower thermal diffusivity of peat soils. A disconnect between soil respiration and environmental variables was also found by Bahn et al. (2009) in an isotopic labelling experiment in an alpine grassland where possible mechanisms included a shift in the proportional contribution of autotrophic and heterotrophic components to soil respiration, or a shift in substrate supply from freshly assimilated carbon to supplies at

least one day old. A final potential reason for the diurnal cycles observed in R_{ECO} may be a lag in the transportation of CO_2 to the atmosphere, associated with daily temperature fluctuations affecting gas diffusivity (Zhang et al., 2015). The relatively recent development of automated chambers and lack of studies addressing diurnal scale variations in CO_2 fluxes means that further investigation is required to determine whether a lag between temperature and chamber derived R_{ECO} fluxes exists, together with monitoring of atmospheric conditions to rule out possible chamber induced disturbance effects.

CH_4 fluxes were negligible from the shallow-drained site and showed no diurnal pattern throughout the study period. In the rewetted grassland, May and September were the only months when enough measurements were obtained to assess whether a diurnal pattern existed. RW-92 showed peak CH_4 emissions during late evening (9pm) in September, whilst RW-91 showed an overall decrease in emissions from 5am to midnight, with a small increase around 9pm also in September. Diurnal controls on CH_4 emissions in peatlands have been found in vegetation which employ pressure-induced flow such as *Phragmites australis* (Hendriks et al., 2010), however those which rely on molecular diffusion such as *Eriophorum* and *Carex* species have not shown significant diurnal variations (Greenup et al., 2000). Although a strong correlation was found between CH_4 emissions and soil temperature at the rewetted sites, a wide range of CH_4 emissions were measured at high temperatures, suggesting an overall seasonal effect, rather than diurnal.

4.5. Conclusion

This study chapter has modelled annual estimates of CO_2 and CH_4 fluxes from grasslands on peat soils with contrasting water table depths. The use of the LI-8100A automatic chamber system greatly improved the GPP light response models compared to those typically achieved using manual chamber techniques, though modelling R_{ECO} fluxes remained a challenge. R_{ECO} modelling at the rewetted grassland was further limited by the inundated conditions over winter and it is therefore recommended that manual chamber measurements are used to complement the LI-8100A system when water tables are above the surface. The power requirements of the system also limited the ability to identify diurnal patterns in CH_4 emissions, which would benefit from further investigation. Similarly, diurnal measurements for longer than 48 hours would enable full investigation of the high nocturnal CO_2 emissions from the drained grassland;

if the lag in peak R_{ECO} and peak soil temperature is real, this could have substantial effects on upscaling R_{ECO} fluxes.

Rewetting is a widely used restoration technique across lowland peatlands in the UK and the results from this study support the notion that rewetting can contribute towards mitigating CO_2 emissions. The shallow-drained grassland was a small net CO_2 source, whilst the rewetted grassland was a large net CO_2 sink, with R_{ECO} the primary difference between the sites. Not only did the shallow-drained grassland have a greater temperature response compared to the rewetted grassland, high nocturnal CO_2 emissions were observed during the summer from the drained site. Due to the high variability in measured R_{ECO} fluxes from both grasslands, the ability of the models used to capture the response of CO_2 emissions to soil temperature alone was limited, and therefore annual R_{ECO} fluxes were likely underestimated in this study. The confidence in annual GPP estimates however, suggests that the direction of net CO_2 exchange is real, though the net CO_2 sink effect of the rewetted grasslands is likely smaller and the net CO_2 source effect of the shallow-drained grassland greater than presented here. The low CH_4 emissions from the drained grassland were expected, whilst the high CH_4 emissions from the rewetted grassland highlight the balance which needs to be achieved during the rewetted of peatlands to avoid a positive GHG balance. The negative GHG balance measured at the rewetted grassland, suggests that extensive grazing following rewetting could be a viable option for conservation management of lowland grasslands on peat in the UK, providing both agricultural use and carbon sequestration.

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Chapter 5

Synthesis and Discussion

5.1. Summary with respect to research aims and questions

The aim of this study was to investigate the response of CO_2 and CH_4 fluxes to restoration management in Welsh lowland peatlands. To achieve this a field based approach was used which enabled the modelling of annual CO_2 and CH_4 fluxes from peatlands subjected to vegetation removal, topsoil removal and rewetting. This chapter begins by summarising the findings of this thesis with respect to the specific research questions. The subsequent discussion places the thesis within a wider context, reviewing the GHG flux measurement methodologies used, comparing annual flux estimates with the Defra Lowland Peat Project and emission factors (both Tier 1 and draft UK specific Tier 2) as well as considering the effect of restoration on peatland CO_2 and CH_4 fluxes and the implications for policy makers.

The specific research objectives to be addressed within each chapter were:

1. To measure CO_2 and CH_4 fluxes from peatlands under restoration management, as well as unrestored peatlands.
2. To identify the controlling environmental drivers of CO_2 and CH_4 fluxes.
3. To produce annual CO_2 and CH_4 balances from the peatlands studied using modelling with environmental drivers.
4. Compare CO_2 and CH_4 balances to those published in the literature, IPCC Tier 1 emission factors and the Defra Lowland Peat project.

Research question 1: *To investigate the effect of restoration interventions, namely vegetation removal and topsoil removal, on CO_2 and CH_4 fluxes from a temperate lowland fen.*

This research question was addressed in Ch. 2 with a field based approach which measured CO_2 and CH_4 fluxes from Cors Erddreiniog; a fen where restoration activities included vegetation removal (by burning and hand-cutting) and the removal of nutrient enriched topsoil from a former improved pasture. Annual CO_2 and CH_4 flux balances were modelled using the relationships between measured fluxes and environmental variables including soil temperature, radiation and water table depth (WTD). The key findings from this study were:

- The tall fen control was a net CO_2 sink, with an increase in net CO_2 uptake observed in the intervention plots. All plots were a net CH_4 source, with the highest emissions observed where *P.australis* dominated. The overall net GHG balance was negative across the plots, with the exception of the *P.australis* stands in the control plot.

- The short fen control plot was a net CO_2 sink, whilst the intervention plots were small to moderate CO_2 sources. All plots were CH_4 sources and the highest emissions originated from the brown moss hollows. A small negative net GHG balance was observed at the control plot when considering the vegetation composition across the site, whilst the intervention plots both had a net warming effect.

- The grassland vegetation within the restored pasture (control plot), was a net CO_2 sink and where topsoil had been removed, the CO_2 balance was near-neutral (following recolonising) or positive (where a bare surface persisted). The control plot was a net CH_4 source, though this did not offset the net CO_2 uptake resulting in a negative GHG balance. CH_4 emissions from the recolonising and bare soil plots were negligible, resulting in neutral and positive GHG balance respectively.

- WTD was a key control on annual CO_2 fluxes across Cors Erddreiniog, with a decrease in annual R_{ECO} fluxes observed as the water table approached the surface. Annual CH_4 emissions were also strongly influenced by WTD, with an increase in emissions observed as the water table rose towards the surface. However, where topsoil removal had occurred CH_4 emissions were negligible, despite the water table being at or near the surface for the majority of the year, suggesting a lag in establishment of methanogenic communities. The absence of plant-mediated CH_4 transport when the water table was below the surface also likely contributed to low CH_4 emissions.

Research question 2: *To investigate the effect of rewetting through the construction of low-lying bunds on CO_2 and CH_4 fluxes from a cutover lowland raised bog*

Research question 3: *Do GHG fluxes from microtopography created by hand-cutting of peat behave in the same way as hummock-hollow complexes in natural peatlands?*

Chapter 3 addressed research questions 2 and 3, with manual chamber measurements carried out at an abandoned cutover lowland raised bog in west Wales, which was undergoing rewetting through the construction of a low-lying bund. The site was rewetted in two stages providing comparisons of CO_2 and CH_4 fluxes from rewetted and non-rewetted microforms as well as before and after rewetting. The legacy effect of hand cutting peat was investigated by measuring fluxes from lowered cuttings and raised buffer strips which had not been cut, a microtopography somewhat analogous to that of natural hummock-hollow complexes. The key findings from this chapter were:

- CO_2 fluxes from corresponding microforms showed no significant difference with rewetting, potentially due to the prolonged drained state of the site with not enough time for microbial communities to recover.

- CO_2 differences between microforms (i.e. ridges and troughs) were dependent upon WTD, with no difference observed when the water table was low. However following the second rewetting treatment the ridges had a significantly greater CO_2 uptake compared to the troughs due to the increased productivity of bryophytes and deep inundation of troughs which submerged vegetation.

- CH_4 fluxes differed with WTD, with fluxes highest from the troughs compared to the ridges, and from rewetted compared to non-rewetted microforms. Additionally, fluxes were high where aerenchymatous species were present.

- The recently created bund acted as a hotspot for CO_2 and CH_4 fluxes, particularly where aerenchymatous vegetation was present.

Research question 4: *To investigate the effect of rewetting on CO_2 and CH_4 fluxes from a grassland on peat.*

Research question 5: *Are diurnal fluctuations in CO_2 fluxes higher in a shallow-drained grassland on peat compared to rewetted grasslands on peat?*

Research questions 4 and 5 were answered in Ch. 4, in a field study using automatic chamber measurements within a shallow-drained and rewetted grassland on peat in mid Wales. Annual CO_2 and CH_4 fluxes were modelled based on the relationship with soil temperature and radiation. Whilst diurnal measurements greatly improved GPP modelling, high scatter in R_{ECO} measurements resulted in large uncertainty of annual flux estimates. The key findings from this chapter were:

- The shallow-drained grassland on peat was a small net CO_2 source, whereas the rewetted grasslands were strong net CO_2 sinks. However, the uncertainty in R_{ECO} modelling suggests that the shallow-drained grassland was likely a larger CO_2 source, and the rewetted a smaller CO_2 sink.

- The rewetted grassland was a large CH_4 source, with fluxes driven by both soil temperature and WTD, whilst emissions from the shallow-drained site were negligible.

- The net GHG balance was dominated by CO_2, and thus the radiative forcing of the shallow-drained grassland was positive and the rewetted grassland negative.

- Diurnal fluctuations in CO_2 were greatest in the shallow-drained site, with high nocturnal emissions in the summer, suggesting a lag between peak soil temperature and peak R_{ECO}.

5.2. Review of the methodology

5.2.1. Chamber GHG flux measurements and modelling

The chamber based approach of the three field studies in this thesis is a common method used in determining environmental controls on GHG fluxes on small spatial scales. The accurate quantification of gas fluxes from closed chamber measurements requires careful consideration of chamber design and deployment, gas collection and analysis as well as the flux calculation method used. Flux uncertainty can be minimised through increasing chamber size (height > 0.3 m), ensuring a gas tight seal between the chamber and collar (Pihlatie et al., 2013) and using fans to ensure mixing of the chamber headspace (Christiansen et al., 2011). The chamber designs used in Ch. 2 and Ch. 4 met these best practice requirements, and also aligned with the Defra Lowland Peat Project and because of using the commercial LI-8011A system. The manual chambers constructed for Ch. 3 included the features detailed within the Lowland Peat Project design (e.g. pressure equalisation balloon, fans, air tight seal) however a circular collar and chamber were used following the distortion of some of the square collars used at Cors Erddreiniog resulting from tussock growth. The chambers in Ch. 3 were also of a smaller basal area to ensure that the small scale microforms could be discretely measured (e.g. bund of width < 50 cm).

The use of an in situ GHG analyser throughout this thesis eliminated the potential errors associated with gas flux measurements using syringe sampling (e.g. low frequency sampling, sample storage and transport). The ability to evaluate the chamber headspace gas concentration in real time also enabled disturbance events to be identified (e.g. during chamber placement or ebullition events, either natural or fieldworker induced) and thus measurements could be repeated on the same day, reducing data loss. Additionally if a non-linear change in concentration was observed, general observations could be made to explain this, for example, a reduction in CO_2 uptake during transparent chamber measurements could be linked to clouds passing overhead or the chamber fogging up.

Whilst the manual chamber measurements conducted in Ch. 2 and Ch. 3 enabled small spatial heterogeneity to be assessed, capturing temporal variation was a greater challenge. Seasonal CO_2 and CH_4 variation was captured due to measurements spanning at least 12 months, however the low frequency of manual chamber measurements (once to twice a month) potentially missed emission peaks such as those following precipitation events or mowing of vegetation; a weakness common to manual chamber studies. Furthermore, manual chamber measurements were only conducted during the day, assuming that R_{ECO} fluxes measured during the day with an opaque chamber would the same as night time NEE measurements. The use of automatic chambers, such as the LI-8100A system used in Ch. 4, sought to improve the temporal frequency of gas flux measurements whilst maintaining small scale spatial coverage. A high frequency of gas flux measurements is key for improving our ability to model fluxes from environmental data; GPP light response curves could be fitted on a per campaign basis using the automatic chamber in Ch. 4, rather than pooling the entire annual data set as in Ch. 2, enabling the changing productivity with vegetation growth to be captured and greatly reducing the uncertainty surrounding modelled annual fluxes. The diurnal R_{ECO} measurements, however, revealed high nocturnal fluxes and high scatter with regards to soil temperature which is commonly used as the sole explanatory variable in annual flux modelling. Although automatic chambers have been associated with overestimations of night time fluxes under stable atmospheric conditions (e.g. Schneider et al., 2009; Juszczak et al., 2012; Koskinen et al., 2014), the probability of low turbulence was reduced by the location of the study site on the west coast of the UK. As a result of the R_{ECO} fluxes peaking at night, the uncertainty associated with annual R_{ECO} fluxes in Ch. 4 was relatively high. Due to the logistics of moving the automatic chamber system between sites as well as the livestock presence, it was not feasible to extend the system deployment to more than 48 hours, which might have given an indication of whether high nocturnal CO_2 fluxes were persistent. The high power demands of the system, particularly with the incorporation of the Los Gatos GHG analyser, also limited high frequency CH_4 flux measurements and therefore the diurnal cycle could not be investigated. Again due to the required deployment and removal of the system on a monthly basis, a larger power supply could not be established at the site. The modelling of annual fluxes using automatic chambers is sparse throughout the literature and is an area which would benefit from further research to determine under what conditions fluxes are unreliable and the optimum data collection period from which fluxes can be upscaled.

5.3. Emissions from UK lowland peatlands under restoration management: Comparison of measured and modelled fluxes to the Lowland Peat Project, Tier 1 and Tier 2 emission factors

Although lowland peatlands within the UK have been disproportionately modified compared to upland peatlands, their contribution to GHG emissions has only recently been addressed by the Defra Lowland Peat Project (Evans et al., 2017a). The Lowland Peat Project quantified GHG fluxes from lowland peatlands across a range of land uses from conservation managed near-natural fens to managed grasslands and croplands. Whilst the peatlands studied within this thesis did not extend to those under arable management, comparisons can be made for conservation managed fens and grasslands, as well as between topsoil removal and extraction sites. Additionally, the emission factors (EFs) derived under the '2013 Supplement to the 2006 Guidelines for National GHG Inventories: Wetlands' (IPCC 2014, hereafter Wetland Supplement) will be considered. The Wetland Supplement provided default Tier 1 EFs for rewetted organic soils, enabling GHG removals through peatland restoration (specifically rewetted organic soils) to be included in national GHG reporting. The Tier 1 EFs are classified by broad climate regions (boreal/temperate/tropical), peat type ('nutrient rich' fen peat and 'nutrient poor' bog peat) and land-use categories (e.g. forest, cropland, grassland and extraction). However, the application of default EFs to peatlands within the UK is limited; for example there is no specific classification for modified peatlands which retain semi-natural vegetation or for blanket bogs. Additionally, within the 'temperate' climate zone used in the Wetlands Supplement, many studies were located in drier continental regions, thus their suitability for describing GHG flux dynamics in the oceanic conditions experienced in the UK is limited. The use of Tier 2 (i.e. country specific) EFs seeks to reduce these uncertainties, with UK specific EFs explicitly considering the broad range of peat condition categories and using only climatically relevant studies for deriving EFs (Evans et al., 2017b). Tier 3 EFs are based on empirical relationships between GHG fluxes and drivers and are currently being refined for implementation with organic soils. The discussion below compares the annual flux estimates measured within this thesis with those from the Lowland Peat Project as well as Tier 1 and (draft) UK Tier 2 EFs (data tables in Appendix D).

5.3.1. Conservation Managed Fens

The conservation managed fens included in this thesis were located at Cors Erddreiniog (Ch. 2), with an average CO_2 flux of -1.46 t CO_2-C ha^{-1} yr^{-1} observed. The IPCC Tier 1 EF, although

derived for rewetted fens, is based on data from both rewetted and near-natural sites following statistical analysis which found no significant difference between the two systems (IPCC, 2014, Chapter 3). Unlike the findings within this thesis, the Tier 1 EF gives a net CO_2 emission of +0.5 t CO_2-C ha^{-1} yr^{-1}, which appears inappropriate as fens must have been prolonged net CO_2 sinks in order for the peat to have formed (Evans et al., 2017a). Wilson et al. (2016) attributed this discrepancy to the inclusion of recently and/or incompletely rewetted peatlands in the analysis. Indeed, the Lowland Peat Project found conservation managed fens (including EC data from the TF-control and SF-control areas of Cors Erddreiniog) to be net CO_2 sinks (-1.43 t CO_2-C ha^{-1} yr^{-1}), as did the draft Tier 2 EF for near-natural fens (-1.5 t CO_2-C ha^{-1} yr^{-1}, Evans et al., 2017b); in good agreement with the findings of this thesis (Fig 5.1). The Tier 2 EF for rewetted fens indicates a small net CO_2 source (0.2 t CO_2-C ha^{-1} yr^{-1}), and whilst the water table was not altered during the restoration interventions within the TF and SF at Cors Erddreiniog, it is likely that historical drainage and recent ditch blocking have modified the water table within the restoration plots. The strong net CO_2 uptake observed at the TF-burnt plot, attributed to inundation causing low R_{ECO}, was beyond the 95 % confidence interval of the rewetted fen Tier 2 EF (-2.2 t CO_2-C ha^{-1} yr^{-1}). No studies from inundated fens were included in calculating the rewetted Tier 2 EF, which may explain the discrepancy. Whilst inundation is generally considered an undesirable outcome following rewetting due to the potential for high CH_4 emissions, it may be important to consider fluxes under shallow-inundation, particularly if rewetting aims to create habitats for birds which require some open water.

The mean annual CH_4 emissions within this study (100 kg CH_4-C ha^{-1} yr^{-1}) are similar to, butslightly lower than, CH_4 emissions reported by the Lowland Peat Project (129 kg CH_4-C ha^{-1} yr^{-1}) and the draft Tier 2 EFs for near natural and rewetted fens (116 and 127 kgCH_4-C ha^{-1} yr^{-1} respectively) (Fig. 5.1). Although these emissions are within the (extremely) wide range of emissions used to derive the IPCC Tier 1 EF (0 to 856 kg CH_4-C ha^{-1} yr^{-1}), the fluxes within this study are around 50 % lower than the Tier 1 value (216 kgCH_4-C ha^{-1} yr^{-1}). One explanation for the lower CH_4 emissions from this study compared to the Tier 1 EF is the stable water table conditions within the fens measured and inundation only occurring in some areas over winter rather than year round. Soon after rewetting, CH_4 emissions may peak as fresh material becomes saturated, providing increased substrate for methanogenic communities. As with CO_2 fluxes, there is considerable variability in annual flux balances under inundated conditions, as well as how fluxes change with time since rewetting.

The large range in annual fluxes measured within this study, and the Tier 2 EFs (compared to bogs) reflects the high diversity usually found in fens; within Cors Erddreiniog vegetation ranged from highly productive sedges and reeds to low productivity brown mosses. This spatial heterogeneity persists partially due to the previous land use of the site affecting the water table position and nutrient status, as well as the intrinsic variability found in fens arising from the spatially heterogeneous groundwater inputs which in turn determine fen biogeochemistry and species composition. Consequently, the variability in annual GHG balance found across Cors Erddreiniog here, is likely common to fens across the UK, particularly where impacted by either direct or indirect human impact. Consequently, further investigation into the driver of GHG fluxes, such as vegetation and WTD, are necessary for further refinement of EFs.
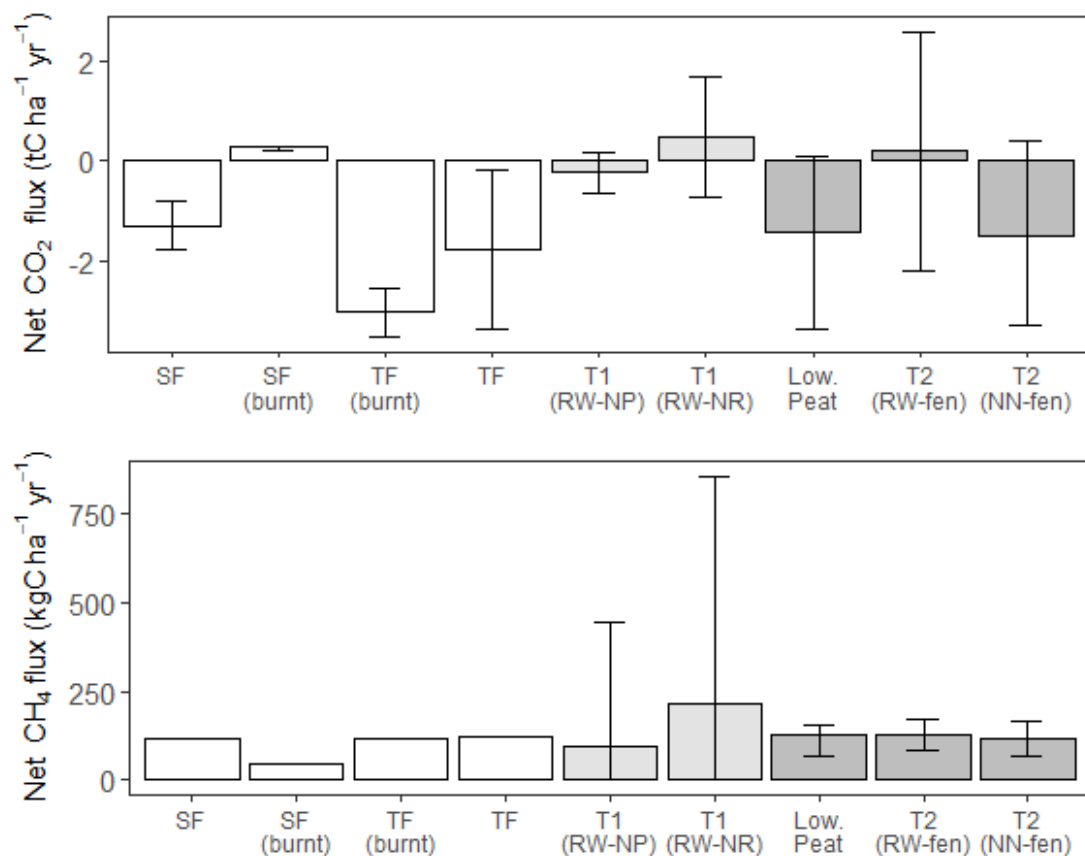
Figure 5.1. Conservation managed fen annual flux balances and emissions factors for (a) CO₂ and (b) CH₄. SF = short fen control, SF (burnt), TF (burnt), TF = tall fen control. T1 = Tier 1 EFs from the Wetlands Supplement for RW-NP = rewetted nutrient poor peatlands; RW-NR = rewetted nutrient rich peatlands (IPCC, 2014); LP = Lowland Peat project (Evans et al.,

5.3.2. Managed Grasslands

Grasslands are one the most widespread land uses for lowland peatlands, however management can range with regards to grazing (low to high density), fertilisation and hay-cropping, creating variability in CO_2 and CH_4 fluxes. Within this study, grasslands on peat were measured both at Cors Erddreiniog (Ch. 2) and Cors Fochno (Ch. 4), with net CO_2 balances ranging from a moderate CO_2 sink to CO_2 source (-2.7 to 2.4 t CO_2-C ha^{-1} yr^{-1}). The Tier 1 EFs for shallow-drained grasslands on peat indicate net CO_2 emission; 3.6 t CO_2-C ha^{-1} yr^{-1} and 5.3 t CO_2-C ha^{-1} yr^{-1} for nutrient rich (i.e. fen peat) and nutrient poor (i.e. bog peat) conditions respectively (IPCC, 2014). The Tier 2 EFs also indicate a net CO_2 emission, but are based on management intensity categories (intensive and extensive) rather than peat type. The extensive grassland Tier 2 EF (3.6 t CO_2-C ha^{-1} yr^{-1}) and annual CO_2 balance of the managed grasslands in the Lowland Peat Project (2.85 t CO_2-C ha^{-1} yr^{-1}) both represent larger net CO_2 emissions than measured in this thesis (Fig. 5.2a). Net CO_2 emissions were measured at two sites within this study (SF-cut at Cors Erddreiniog and SD in Cors Fochno), with the annual CO_2 balance of the SF-cut plot (2.35 t CO_2-C ha^{-1} yr^{-1}) towards the lower range of the Tier 2 EF (2.1 t CO_2-C ha^{-1} yr^{-1}). An explanation for the net CO_2 uptake, rather than net emission, observed at the remaining grasslands in this thesis may be the difference in mean annual WTD. Although no WTD was specified for the Tier 2 grassland EFs, the studies used to derive the EFs had mean annual WTDs of at least 10 cm below the surface. In contrast, the mean annual WTD of the grasslands measured within this thesis were all between 2 cm above the surface and 11 cm below the surface, with the exception of the drained grassland in Cors Fochno which was 37 cm below the surface. Given the high water tables, these sites (omitting SD in Cors Fochno) are more in line with 'rewetted fens' within the IPCC Tier 1 classification which has an EF of 0.5 t C ha^{-1} yr^{-1}, again highlighting the requirement of refining EFs from the global defaults. The relationship between annual net CO_2 balance and mean annual WTD using data measured within the thesis, as well as the Lowland Peat Project and by Renou-Wilson et al. (2016) from a rewetted and shallow-drained extensive grassland in Ireland is presented in Fig. 5.3a. The strong relationship between net CO_2 balance and WTD ($r^2 = 0.52$, $p = 0.001$) indicates that raising the water table may result in extensive grasslands functioning as net CO_2 sinks, with the regression analysis suggesting that CO_2 emissions are reduced by 3.8 t CO_2-C ha^{-1} yr^{-1} as

the WTD rises by 10 cm. The uncertainty surrounding the R_{ECO} modelling at the Cors Fochno grasslands meant that the annual CO_2 balances here may have overestimated the CO_2 sink function, however there is good agreement in the net CO_2 balance from the drained grassland at Cors Fochno and Bakers Fen (Lowland Peat Project), which had similar water tables and management regimes (low intensity grazing with no biomass harvesting). The R_{ECO} underestimation at the rewetted Cors Fochno grasslands may have resulted in the net sink function at the site being overestimated, but the confidence in the high productivity modelled suggests that the sites are in fact functioning as net CO_2 sinks. The strong relationship across the dataset shown in Fig. 5.3a indicates the high potential of deriving Tier 3 EFs based on the mean annual WTD of grassland systems on peat soils.

The grasslands measured in this thesis were net sources of CH_4, with emissions ranging from 11.9 kg CH_4-C ha^{-1} yr^{-1} to 217 kg CH_4-C ha^{-1} yr^{-1}. The wide range in emissions reflects the range of water tables included within this study; where the mean annual water table was at least 5 cm below the surface, annual CH_4 emissions averaged 41 kg CH_4-C ha^{-1} yr^{-1}, similar to the Tier 1 EF for nutrient rich shallow drained peatlands (29 kg CH_4-C ha^{-1} yr^{-1}) and the Tier 2 extensive grassland EF (55 kg CH_4-C ha^{-1} yr^{-1}) (Fig 5.2b). In contrast, where water tables were within 3 cm of the surface, annual CH_4 emissions were much higher (130 to 217 kg C ha^{-1} yr^{-1}), closer to the Tier 1 EF derived for rewetted nutrient rich peatlands (216 kg CH_4-C ha^{-1} yr^{-1}) and updated Tier 1 EFs by Wilson et al. (2016) (90 and 236 kg CH_4-C ha^{-1} yr^{-1} for rewetted grasslands on bog and fen peat respectively). The near zero annual CH_4 emissions observed from the grasslands in the Lowland Peat Project reflect the low water tables at those sites. An approximately linear relationship was observed between CH_4 emissions and mean annual WTD (Fig. 5.3b), where this was within ~ 20 cm of the surface (r^2 0.71 including only thesis data; r^2 0.76 including thesis and Renou-Wilson et al. (2016) data). Again, this good correlation with WTD is promising with regards to developing Tier 3 EFs based on simple empirical models. The CH_4 emissions from the rewetted grassland at Cors Fochno are slightly higher than the regression; possibly because of the use of a temperature driven model to estimate annual fluxes which may overestimate summer CH_4 fluxes when water tables were low.
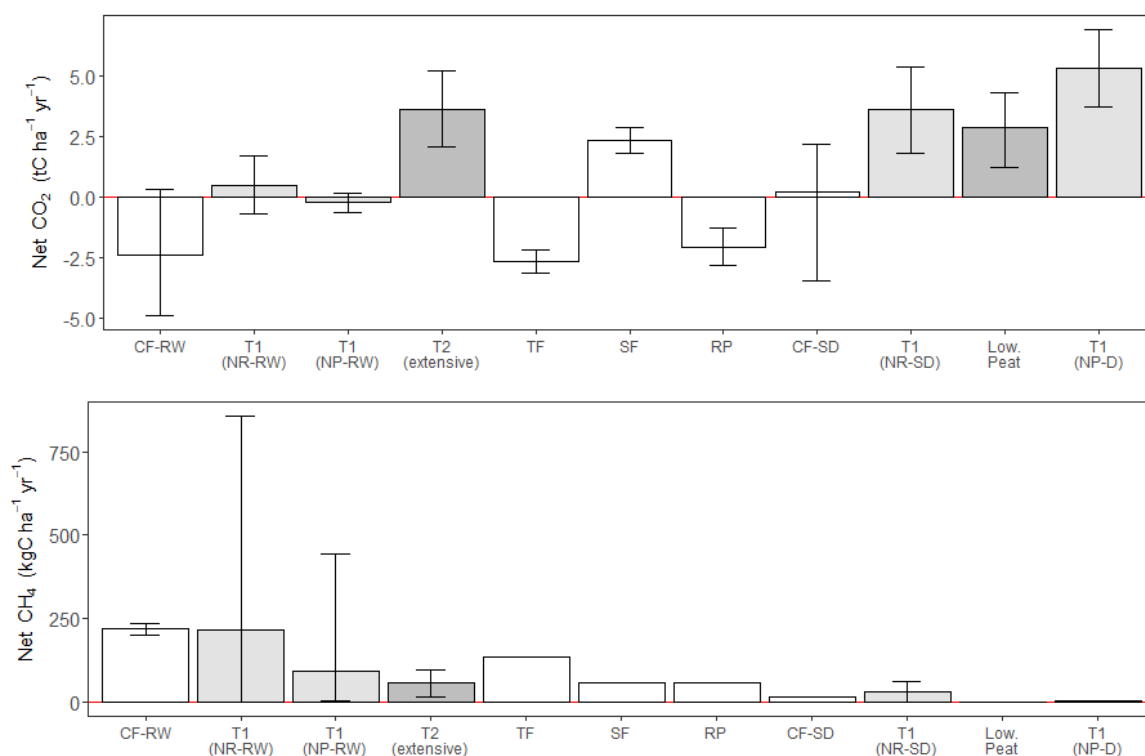
Figure 5.2. Managed grassland annual flux balances and emissions factors for (a) CO2 and (b) CH4. SF = short fen hand-cut; TF = tall fen hand-cut; RP = restored pasture control, CF-RW = Cors Fochno rewetted; CF-SD = Cors Fochno drained; T1 = Tier 1 EFs (IPCC, 2014) for RW-NP = rewetted nutrient poor peatlands; RW-NR = rewetted nutrient rich peatlands; NR-SD = nutrient rich shallow drained; LP = Lowland Peat project (Evans et al., 2017a); T2 = Tier 2 EFs for extensive grasslands (Evans et al., 2017b).
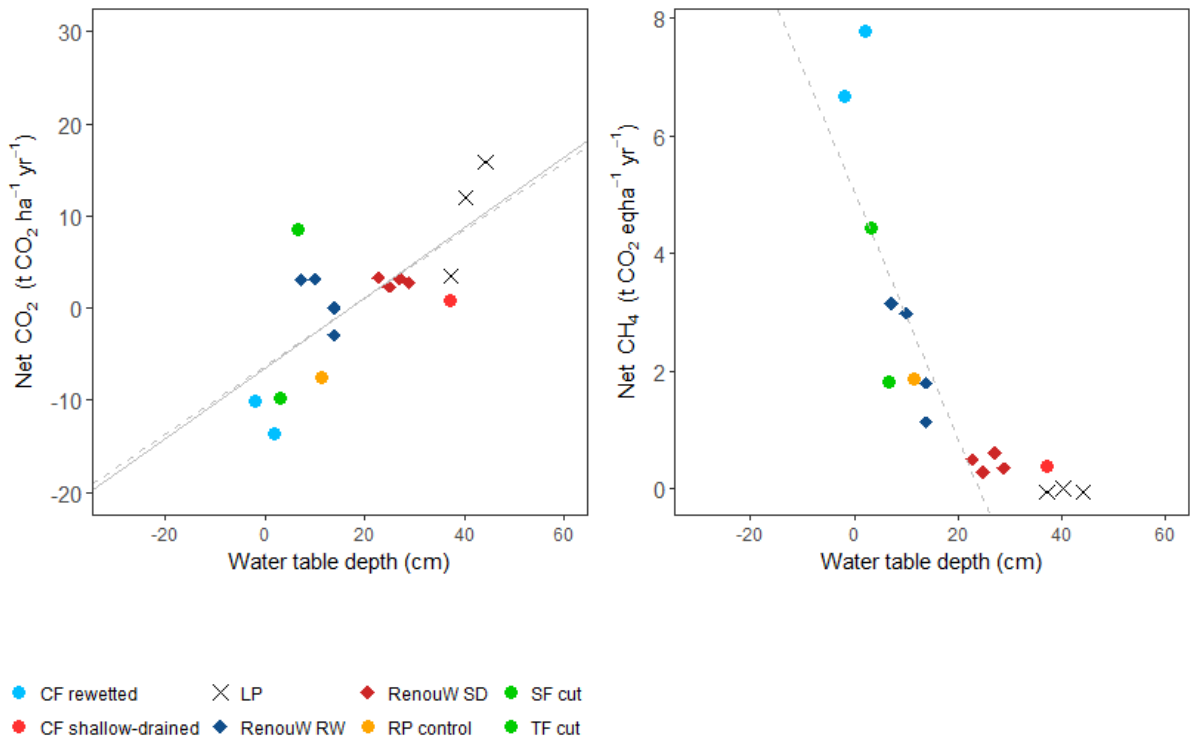
Figure 5.3. Relationships between (a) net CO_2 fluxes; (b) terrestrial CH_4 flux; and mean annual water table depth for grassland study sites measured in this thesis (circles); the Lowland Peat Project (crosses) and Renou-Wilson et al. (2016). Linear regression lines are shown using the entire dataset (solid line) and from the Lowland Peat Report (dashed line), and (b) using points with water table < 20 cm. Note a positive WTD indicates below the surface.

5.3.3. Bare soil/ peatlands under extraction

The removal of significantly enriched topsoil (e.g. RP-bare soil in Ch. 2) is an extreme intervention in peatland restoration and draws many similarities with restoring bare surfaces following peat extraction. The bare soil plot in Cors Erddreiniog was an annual CO_2 source of 1.84 t CO_2-C ha^{-1} yr^{-1}, in line with the peat extraction sites within the Lowland Peat Project (1.4 t CO_2-C ha^{-1} yr^{-1}) and Tier 2 EFs (1.3 and 1.8 t CO_2-C ha^{-1} yr^{-1} for domestic and industrial extracted peatlands) (Fig. 5.4). The IPCC Tier 1 EF for peatlands under extraction (2.8 t CO_2-C ha^{-1} yr^{-1}) is ~ 50 % greater than the annual CO_2 emission measured in this thesis, though the range of values used to derive the Tier 1 EF overlap (1.1 to 4.2 t CO_2-C ha^{-1} yr^{-1}). The topsoil removal restoration at Cors Erddreiniog aimed to both reduce the eutrophic status of the site and raise the water table, however due to the fact that measurements were taken soon after restoration intervention, the mean annual WTD was still 11 cm below the surface, and little vegetation re-establishment had taken place. Thus the annual CO_2 emissions are in line with

those from drained extraction sites, rather than rewetted (and revegetated) extracted peatlands where EFs suggest a net CO_2 uptake; -0.23 tCO_2-C ha^{-1} yr^{-1} (IPCC Tier 1) and -0.33 tCO_2-C ha^{-1} yr^{-1} (Wilson et al., 2016). In order for a negative CO_2 balance to be observed, vegetation must be present indicating that the rewetted EFs are applicable to sites only where colonisation has occurred. In contrast, although 'rewetting' had occurred following topsoil removal at Cors Erddreiniog, the plot was maintained as bare soil and therefore a net uptake of CO_2 was impossible. Given these EFs, it would be expected that as colonisation occurs and the water table rises, the annual CO_2 balance will become negative as the C sink function is re-established. Indeed, the plot where recolonisation had occurred following topsoil removal had an annual net CO_2 balance of -0.02 t CO_2-C ha^{-1} yr^{-1}, indicating a shift in the C balance from source to neutral within two years.

CH_4 emissions were negligible from the bare soil plot in Cors Erddreiniog (1 kg C ha^{-1} yr^{-1}), similar to the findings from the extraction sites within the Lowland Peat Project which ranged from -0.4 to 1.8 kg CH_4-C ha^{-1} yr^{-1} (average of 0.7 kg CH_4-C ha^{-1} yr^{-1}) (Fig 5.4). The Tier 1 EF for peatlands under extraction also shows a net emission of CH_4, but of a greater magnitude; 4.6 kg CH_4-C ha^{-1} yr^{-1}, with the fluxes used to derive this value ranging from 1.2 to 8 kg CH_4-C ha^{-1} yr^{-1}. Wilson et al. (2016) found CH_4 emissions from rewetted extraction sites to be higher (90 kg CH_4-C ha^{-1} yr^{-1}) with the suggestion that rewetting nutrient poor cutover peatlands will result in a steady increase in CH_4 emissions immediately after rewetting as the emergent vegetation provides fresh substrate for CH_4 production. However, at Cors Erddreiniog the observed CH_4 emissions remained low after two years of rewetting following topsoil removal, indicating there may be a lag between vegetation colonisation and the recovery of methanogenic communities. This is an area which requires more investigation to refine emission factors; specifically the effect of time since rewetting on CH_4 production, whether this lags behind vegetation development, and when environmental parameters such as WTD and vegetation start to control the CH_4 balance. Further to this, the rewetting of bare peat is a transitional phase, and therefore likely susceptible to large interannual variations, making accurate quantification of GHG fluxes more challenging. Consequently it may be necessary to quantify CH_4 emissions with regards to time since rewetting using long timescales (decades), which will capture the development of vegetation from bare soil. This would also address the issue of rewetted extraction EFs currently including vegetated sites, rather than just rewetted bare soil.
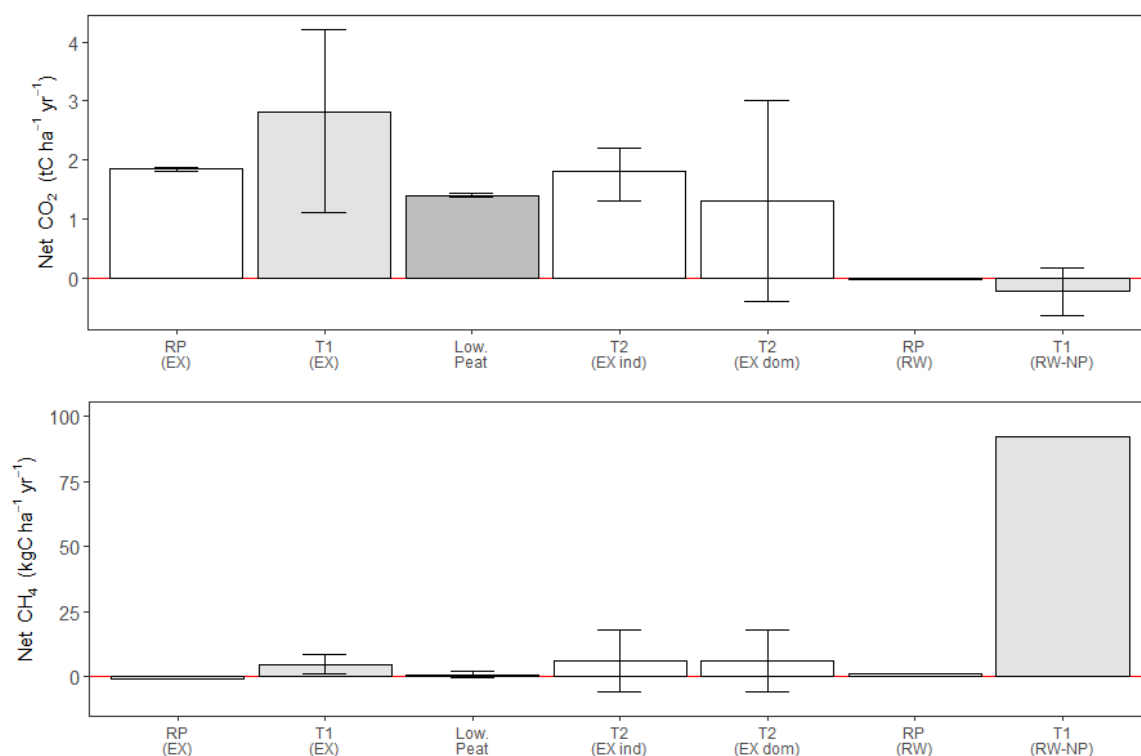
Figure 5.4. Bare soil and extraction site annual flux balances and emissions factors for (a) CO_2 and (b) CH_4. RP = restored pasture bare soil; T1 = Tier 1 EFs (Wetlands Supplement) for EX = peatlands under extraction (IPCC, 2014); LP = Lowland Peat project (Evans et al., 2017); T2 = draft Tier 2 EFs for peatlands under industrial (ind) and domestic (dom) extraction (Evans et al., 2017).

The lack of automatic weather station data available at Pant-y-Dwn (Ch. 3) meant that annual CO_2 and CH_4 balances could not be calculated and therefore comparisons could not be made to EFs from rewetted bogs. Tier 1 EFs indicate rewetted bogs are net CO_2 sinks (-0.23 t CO_2-C ha^{-1} yr^{-1}), with Tier 2 EFs suggesting a stronger net sink effect (-0.6 t CO_2-C ha^{-1} yr^{-1}). The rewetted raised bogs within the Lowland Peat Project (Astley Moss and Thorne Moors) were both found to be net CO_2 sinks, though there was high uncertainty surrounding the modelling from closed-chamber measurements. The measured net CO_2 fluxes from rewetted microforms at Pant-y-Dwn were generally within the range of those observed at Astley Moss, with the net CO_2 uptake from the rewetted ridges similar to that from *Molinia* dominated areas during the growing season (Fig. 5.5a). The NEE of the rewetted troughs was similar to that from the *Sphagnum* dominated areas during the winter, however the troughs were a stronger net CO_2 sink during the growing season (Fig. 5.5b). In contrast, there was little correlation with modelled NEE fluxes from the *Molinia* and *Eriophorum* plots within Astley Moss, where a net

CO_2 emission was measured throughout the year (Fig. 5.5c) or Thorne Moors where the net CO_2 exchange was considerably greater than that observed at Pant-y-Dwn (Fig. 5.5d).
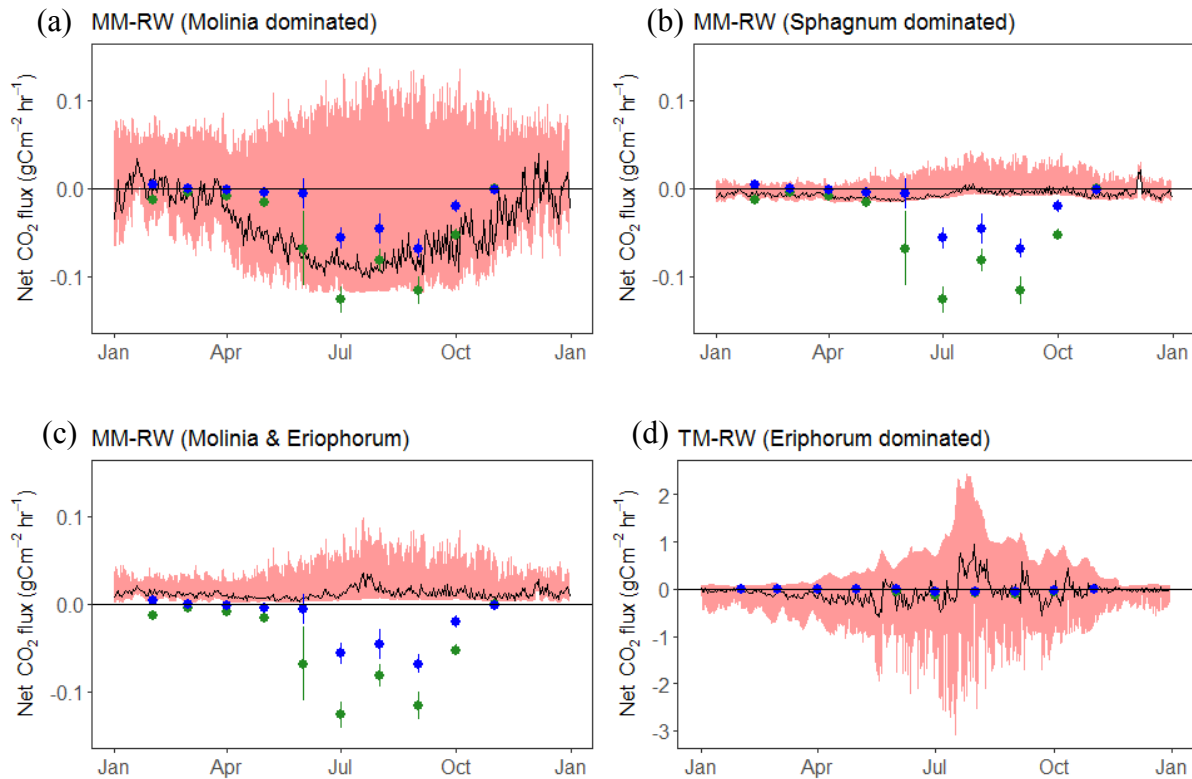
Figure 5.5. Modelled annual net CO_2 exchange at the (a –c) Manchester Mosses rewetted (MM-RW) and (d) Thorne Moor rewetted (TM-RW) peatland sites from the Lowland Peatland Project (Evans et al., 2017a). The mean monthly measured NEE for troughs (blue) and ridges (green) are shown, with error bars representing ± 1 SE of the mean.

5.3.4. Towards UK specific and Tier 3 emission factors

Modification of UK peatlands is extensive, and restoration efforts will not necessarily return systems to their pre-modified state, For example, lowland raised bogs drained for agriculture (e.g. Ch. 4) are more likely to support fen vegetation following restoration due to the legacy of agricultural inputs into the peat. Therefore, the separation of EFs by peat type (fen peat or bog peat as in Tier 1 reporting) may not always be appropriate. For this reason, the development of UK Tier 2 EFs found grasslands on peat were more representative when categorised as either extensive or intensive, with the former typically (but not exclusively) on bog peat and the latter on fen peat. Tier 3 reporting uses empirical models to derive EFs, however as Tier 2 categories

become more specific, the range in environmental drivers will also narrow (e.g. WTD) and therefore it may be necessary to consider peatland systems within a broader context. Given the relatively small water table ranges within each of the land use categories described above, the annual fluxes modelled within this thesis were collated with those from the Lowland Peat Project, in order to determine whether WTD was an overarching driver of GHG fluxes (Fig. 5.6a). A significant linear relationship was found between net CO_2 flux and WTD ($r^2 = 0.70$, $p \leq 0.001$, n = 27), with an intercept of -4.61 tCO_2 ha^{-1} yr^{-1} indicating a net CO_2 sink when the water table is at the surface. The gradient of 0.34 implies for every 10 cm increase in the water table towards the surface, the net CO_2 loss is reduced by 3.3 tCO_2 ha^{-1} yr^{-1}, whilst the intersection with the x-axis suggests that when the water table is within 14 cm of the surface, net CO_2 uptake occurs. The regression derived in the Lowland Peat Project suggested a greater net CO_2 uptake (slope = 0.37, intercept = -6.31 t CO_2 ha^{-1} yr^{-1}, x-axis intersection = 17 cm), though this analysis omitted the annual fluxes from the rewetted extraction sites. The inclusion of the Lowland Peat Project rewetted extraction sites here, together with CO_2 emissions from SF-cut and the RP-bare soil plot (Ch. 2) are likely the reason why a smaller intercept and gradient were found from the collated dataset. High uncertainty surrounded the GPP modelling at Astley Moss where the annual net CO_2 uptake was negligible when the water table was at or near the surface. Further evidence for suppression of GPP under inundated conditions was present in the troughs at Pant-y-Dwn following the second rewetting treatment (Ch. 3). In contrast the annual CO_2 emission modelled at Thorne Moor was attributed to high summer R_{ECO} under high temperatures which dominated over productivity. Similarly, high R_{ECO} was observed at the SF-cut (Ch. 2) resulting in a net annual CO_2 emission. The high variability in CO_2 fluxes where the water table is close to the surface not only reflects the large variability in lowland peatlands (e.g. the botanical diversity of fens), but also highlights that the equilibrium between vegetation communities and WTD takes time to re-establish following rewetting.

CH_4 emissions from the collated dataset also showed a significant linear relationship with mean annual water table depth (Fig. 5.6b), where the water table was above 22 cm ($r^2 = 0.52$, $p = 0.002$, n = 17). The regression analysis indicates that when the water table is at the surface, an annual CH_4 emission of 4.9 t CO_2-eq ha^{-1} yr^{-1} can be expected, with emissions increasing by 2.2 t CO_2-eq ha^{-1} yr^{-1} with every 10 cm increase in mean annual WTD. The high scatter surrounding CH_4 emissions where the water table was within 10 cm of the surface is typical of the highly variable hydrological and botanical conditions found within fen systems. The near zero CH_4 emissions observed where topsoil removal occurred at Cors Erddreiniog (Ch. 2),

despite the water table being above the surface, suggests the re-establishment of microbial communities lags behind vegetation colonisation.
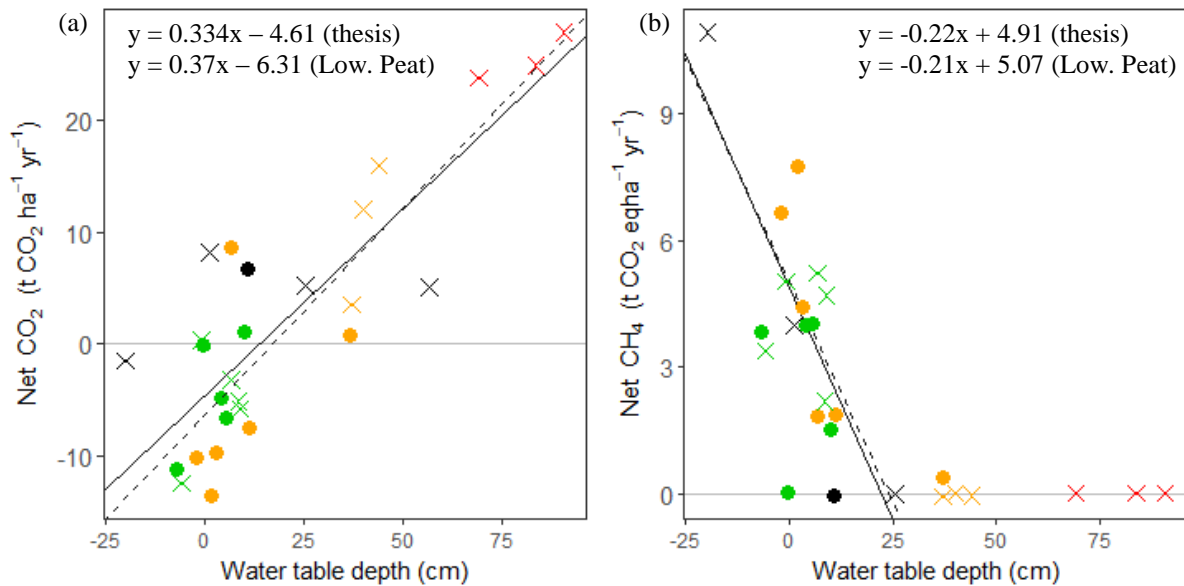
Figure 5.6. Relationships between (a) net CO₂ flux; (b) terrestrial CH₄ flux and mean annual water table depth for study sites measured in this thesis (circles) and the Lowland Peat Project (crosses). Sites are colour-coded as: green = conservation managed fen; orange = grassland; red = arable; black = bare soil/extraction. Linear regression lines are shown using the entire dataset (solid line) and from the Lowland Peat Report (dashed line), and (b) using points with water table < 30 cm, except the bare soil and recolonised plots. Note a positive WTD indicates below the surface.

Given the higher radiative forcing of CH_4, peatlands functioning as a net CO_2 sink may still have a positive GHG balance (i.e. a radiative warming effect) if CH_4 emissions are high. Regression analysis of net GHG flux and mean annual WTD showed a decrease in net GHG flux as the water table approached the surface, however an increase was then observed as the water table rose above the surface (Fig. 5.7), indicating that inundation may lead to an increase in net GHG flux. The regression line did not cross the x-axis suggesting that CO_2 uptake (on average) is offset by CH_4 emissions across all water tables, however the high scatter within the data, particularly within 10 cm of the surface, means that this is far from certain, and that individual peatlands can act as net GHG sinks where conditions are right.
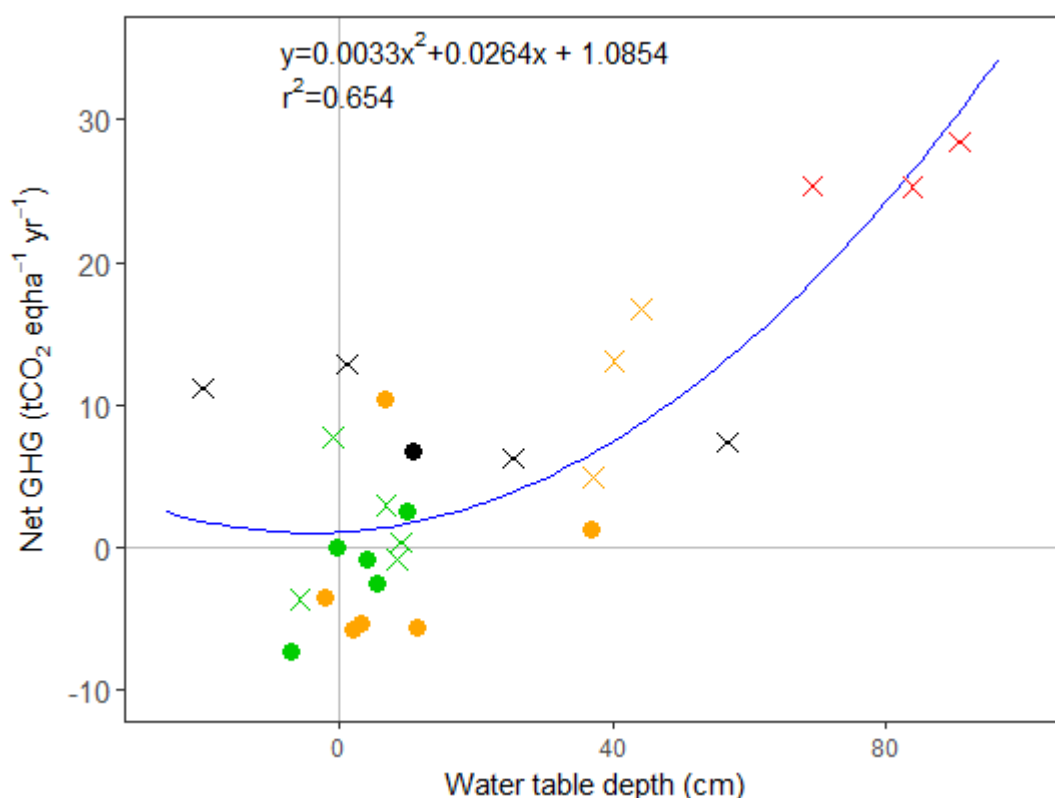
Figure 5.7. Relationship between net GHG flux and mean annual water table depth for study sites measured in this thesis (circles) and the Lowland Peat Project (crosses). Sites are colour-coded as: green = conservation managed fen; orange = grassland; red = arable; black = bare soil/extraction. Note a positive WTD indicates below the surface.

Annual CO_2 and CH_4 balances measured within this study are in good agreement with the draft Tier 2 EFs which address the key differences between UK peatlands and those included in the Tier 1 EFs, namely the oceanic climate conditions and peat condition categories. The near natural fen and bare soil/extraction fluxes measured in this study and the Tier 2 EFs are consistently lower than the Tier 1 EFs (Fig. 5.8), suggesting these could be reduced for UK reporting. Whilst the Tier 2 extensive grassland EFs are lower than the Tier 1 drained grassland EFs, annual fluxes measured in this thesis are notably lower (Fig. 5.8) Although there is some (particularly R_{ECO}) modelling uncertainty which suggests the net CO_2 sink function of rewetted grasslands (in Ch. 4) is not as strong as reported here, the high confidence in GPP modelling and agreement with the regression analysis with WTD (Fig. 5.6a) suggests that a relatively strong net CO_2 uptake is highly likely. Finally, the significant relationships between CO_2 and CH_4 fluxes and mean annual WTD are promising for the development of Tier 3 EFs for UK CO_2 and CH_4 reporting.
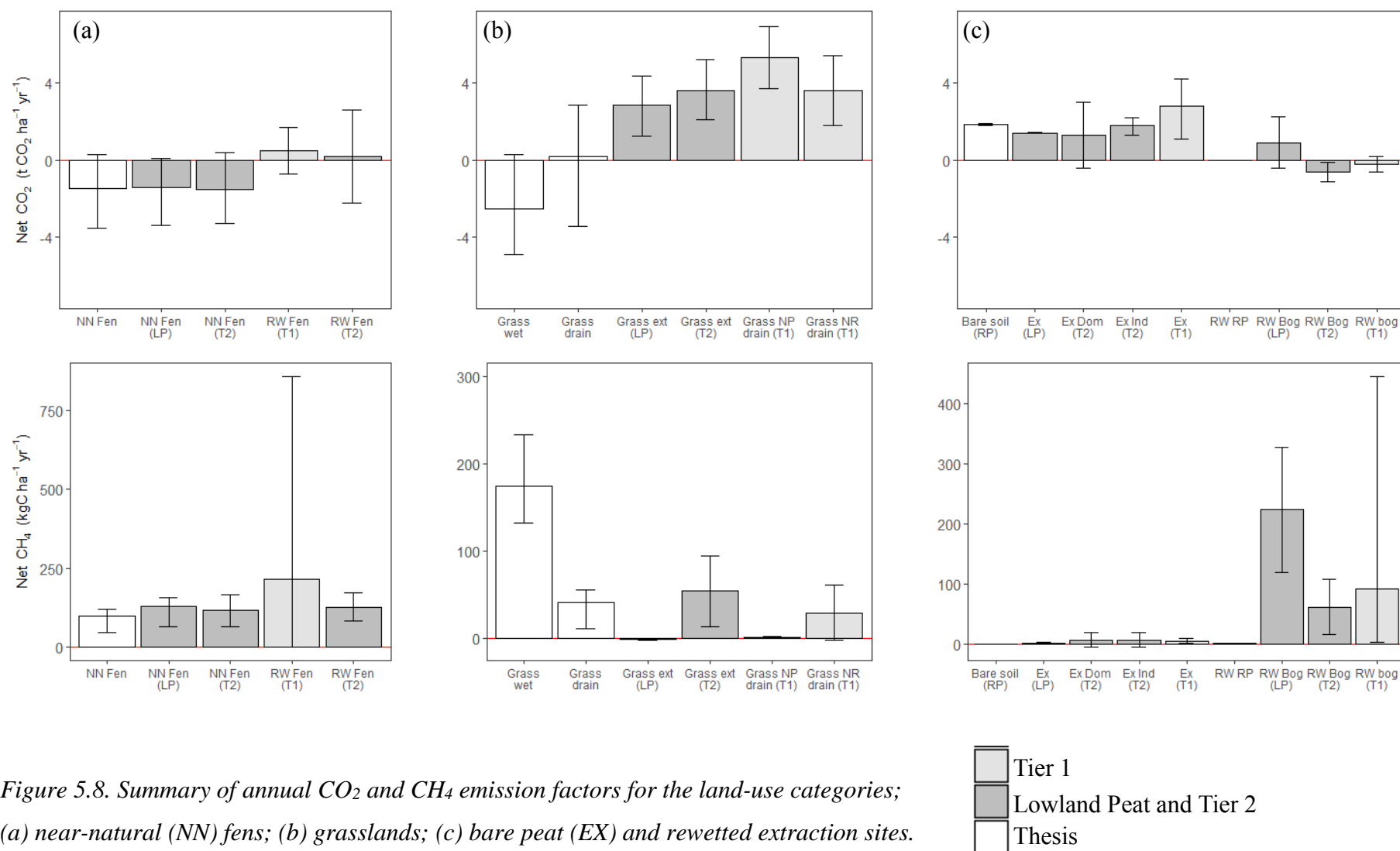
Figure 5.8. Summary of annual CO₂ and CH₄ emission factors for the land-use categories; (a) near-natural (NN) fens; (b) grasslands; (c) bare peat (EX) and rewetted extraction sites.

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5.4. Peatland restoration and policy implication

Numerous global agreements recognise the importance of peatlands as wetlands (Ramsar, EU Water Framework Directive) and with regards to ecosystem services such as climate mitigation (Kyoto Protocol) and biodiversity conservation (Nagoya Protocol, EC Habitats Directive). Thus their conservation and restoration has become increasingly important, particularly given their possible role in meeting stringent emission targets (IPCC, 2014). Whilst there is no overarching UK peatland policy, recent assessments have been carried out on the condition of UK (Bain et al., 2011) and Irish peatlands (BOGLAND, Renou-Wilson et al., 2011). The Commission of Inquiry highlighted the need for a co-ordinated ecosystem based approach to peatland policy, to ensure conservation as well as restoration of peatlands through active habitat management or intervention where severe modification has occurred. In practice, peatland restoration is increasing within Europe and the UK, with at least 120 projects conducted within the UK (Bonn et al., 2016), contributing towards the Inquiry target of having 1 million ha of peatland in good condition or under restoration management by 2020. To further help achieve this target, the UK Peatland Code is under development, as a voluntary standard for peatland restoration projects, with sponsorship provided by businesses through their corporate social responsibility activities. The Code provides guidance to ensure high environmental standards with regards to carbon sequestration, biodiversity and water quality as well as providing assurances to sponsors that they are contributing to measurable, verifiable and long term environmental benefits.

Further potential for peatland restoration has arisen with the announcement of £10 million of government funding for peatland restoration within England, for projects which aim to increase peatland capacity to prevent carbon entering the atmosphere, reduce flood risk and provide habitats for vulnerable wildlife. In order for the success of peatland restoration to be recognised and quantified however, the aims of restoration need to be specified and sufficient monitoring undertaken to assess environmental changes. For example, the vegetation management at Cors Erddreiniog (Ch. 2) contributed to improving botanical condition (and therefore restoring the habitat), but also led to an increase in CO_2 emissions within the short fen, thus having a negative effect on the GHG balance. The overriding control of annual WTD on GHG exchange within UK lowland peatlands (Fig. 5.6 and Fig. 5.7) implies that for restoration to re-establish the climate mitigation function of peatlands, a change in WTD is required. However, the lack of immediate response following rewetting in CO_2 fluxes (e.g. Pant-y-Dwn in Ch. 3) and CH_4

fluxes (e.g. Restored Pasture in Ch.2) suggests that there is a transition phase during peatland restoration in which the gas fluxes observed are not necessarily in equilibrium with the WTD, soil properties or vegetation present. Consequently, the effect of time since rewetting on GHG fluxes is an important area of future research, as peatland restoration increases and the refinement of rewetted peatland EFs is required for national GHG reporting.

Rewetting as a restoration technique is also largely dependent on previous and current land use. At Cors Fochno, two study locations originally part of the same lowland raised bog complex had been subject to rewetting (Ch. 3 and Ch. 4), however the influence of past land use and intrinsic differences resulted in significantly different vegetation compositions. Whilst the hand-cutting at Pant-y-Dwn required drainage and involved the removal of peat, there was a limited effect on the nutrient status and vegetation assemblages, which is reflected by the relatively high current coverage of bryophytes and aerenchymatous species. In contrast, the grasslands on peat which were more deeply drained and used for agriculture, are dominated by wet rush pasture with no defined microtopography. The differences in plant assemblage impact the GHG balance and it would be expected that due to the dominance of more productive vegetation, the grasslands at Cors Fochno would be larger CO_2 sinks compared to Pant-y-Dwn. Whilst annual fluxes could not be calculated for Pant-y-Dwn and therefore the GHG balance cannot be directly compared with the Cors Fochno grasslands, the respective similarity to published literature and Tier 2 EFs suggests the carbon balances would be considerably different. This highlights the importance of clearly defining peatland systems at a (UK) management specific level when reporting to national GHG inventories, as although both are rewetted bogs, the management and vegetation present may result in significantly different annual budgets.

5.5. Recommended further research

Following the discussions in this section and thesis as a whole, the research recommendations below are made:

1. Conduct a functional microbial study of the peat soils to provide an insight into the differences in CH_4 emissions between plant communities (Ch. 2 and Ch. 4) and microtopography (Ch. 3).

2. Quantification of the extent and distribution of lowland peatland restoration projects across the UK, together with environmental monitoring strategies to enable restoration specific GHG emissions to be identified, collated and upscaled.

3. Identification of best available regional examples of peatlands under specific restoration management which can be intensively monitored (ideally with pre-restoration baseline monitoring) which can be used as a point of reference against which other restoration sites can be compared.

4. Field scale measurements over fine time scales to capture short term GHG flux responses to restoration interventions (e.g. during rewetting, immediately after mowing or topsoil removal) when emission pulses may occur. This could be achieved using micrometeorological or automatic chamber methods.

5. Investigation into the effects of grazing (both presence/absence and grazing intensity) on GHG fluxes from peatlands under restoration management, as well as the potential change in vegetation community which may affect GHG dynamics.

6. Water table manipulation experiment to identify the optimal water table conditions for maximum GHG benefits following rewetting; for example the WTD required to balance the restoration of *Sphagnum* spp. and the colonisation of aerenchymatous species e.g. *E.vaginatum*.

7. Investigate the effects of creating inundated areas/open water (not just ditches) during peatland restoration (often for birds) on the GHG balance, specifically if these create hotspots of CH_4 which when scaled up counterbalance CO_2 savings.

8. Full life cycle assessment of restoration management on peatlands, particularly with regards to removal of topsoil or vegetation.

9. Investigations into restoring the hydrological functioning of peatlands; how to keep the peat wet during the summer (especially with potential increases in summer droughts with climate change) and minimise deep inundation in winter.

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Appendix A

Table A1. Ultraportable Greenhouse Gas Analyser performance data (Los Gatos Research Inc.)

GHG	Precision	Measurement range
CH_4	< 2 ppb	0.01 – 100 ppm
CO_2	< 300 ppb	1 – 20000 ppm

Measurement of CO_2 and CH_4 concentration were checked each day in the field to ensure that gas concentrations were ambient (~400 ppm for CO_2 and ~1.9 ppm for CH_4). Prior to monthly measurements instrument checks included the Laser Offset and Signal Ring Down, which were corrected as required. A full instrument calibration was completed on the 28th February 2017 according to the manufacturers instructions and found no significant drift from the factory calibration when the instrument was purchased.

Table A2. Model performance rating criteria from Hoffmann et al. (2015); Mean absolute error (MAE), RMSE-observations standard deviation ratio (RSR), co-efficient of determination (r^2), modified index of agreement (md), percent BIAS (PBIAS) and Nash-Sutcliffs model efficiency (NSE).

Performance rating	MAE	RSR	r^2	md	PBIAS	NSE
Excellent	0 - 5 %	< 0.1	0.9 -1.0	0.9 -1.0	< 15	0.9 -1.0
Very good	5 -15%	0.1 - 0.25	0.75 - 0.9	0.75 - 0.9	15 - 30	0.75 - 0.9
Good	15 - 25 %	0.25 - 0.5	0.5 - 0.75	0.5 - 0.75	30 - 45	0.5 - 0.75
Satisfactory	25 - 50 %	0.5 - 0.75	0.25 - 0.5	0.25 - 0.5	45 - 70	0.25 - 0.5
Unsatisfactory	> 50 %	> 0.75	< 0.25	< 0.25	> 70 %	< 0.25

Table A3. CO$_2$ and CH$_4$ model ratings using criteria defined by Hoffmann et al. (2015). 'Unsatisfactory' model ratings are shown in red, 'excellent' and 'very good' model ratings are shown in green.

Vegetation	Model	adjusted r2	MAE	RSR	MD	PBIAS	NSE
RP control	GPP = Light	0.471	78.90%	0.71	0.62	1.5	0.49
RP recolonised	GPP = Light	0.617	20.90%	0.6	0.66	3.7	0.62
TF burnt	GPP = Light	0.042	62.30%	0.96	0.17	1.4	0.06
TF control *Cladium*	GPP = Light	0.247	77.60%	0.84	0.52	0.7	0.27
TF cut	GPP = Light	0.207	58.70%	0.87	0.4	3.1	0.22
TF control *Phragmites*	GPP = Light	-0.009	78.20%	0.97	0.11	0.4	0.02
SF control Brown moss	GPP = Light	0.504	25.50%	0.68	0.64	0.6	0.52
SF burnt	GPP = Light	0.195	59.20%	0.88	0.47	5.6	0.2
SF cut	GPP = Light	0.548	52.50%	0.66	0.67	3.4	0.55
SF control *Juncus*	GPP = Light	0.138	57.20%	0.9	0.36	0	0.17
SF control *Sphagnum*	GPP = Light	0.127	72.30%	0.91	0.35	3.1	0.15
RP control	GPP = Light * Tsoil	0.603	64.30%	0.61	0.72	3	0.61
RP recolonised	GPP = Light * Tsoil	0.805	13.60%	0.43	0.8	0.3	0.81
TF burnt	GPP = Light * Tsoil	0.578	44.60%	0.7	0.56	12.4	0.5
TF control *Cladium*	GPP = Light * Tsoil	0.714	53.20%	0.53	0.7	6.3	0.71
TF cut	GPP = Light * Tsoil	0.77	35.40%	0.53	0.72	10.9	0.71
TF control *Phragmites*	GPP = Light * Tsoil	0.816	48.60%	0.61	0.6	18.8	0.61
SF control Brown moss	GPP = Light * Tsoil	0.759	16.50%	0.48	0.77	4	0.76
SF burnt	GPP = Light * Tsoil	0.605	45.80%	0.66	0.59	12.2	0.55
SF cut	GPP = Light * Tsoil	0.721	40%	0.52	0.76	5.5	0.72
SF control *Juncus*	GPP = Light * Tsoil	0.799	32.20%	0.54	0.69	12.9	0.7
SF control *Sphagnum*	GPP = Light * Tsoil	0.667	43.70%	0.61	0.62	10.4	0.62
RP control	GPP = Light * ETI	0.524	75%	0.67	0.65	1.4	0.54
RP recolonised	GPP = Light * ETI	0.671	19%	0.56	0.7	3.7	0.68
TF burnt	GPP = Light * ETI	0.339	57.90%	0.87	0.28	8.3	0.22

TF control *Cladium*	GPP = Light * ETI	0.424	69.30%	0.74	0.58	4.6	0.43
TF cut	GPP = Light * ETI	0.366	58.80%	0.85	0.52	31.5	0.25
TF control *Phragmites*	GPP = Light * ETI	0.436	72.20%	0.89	0.22	8.2	0.18
SF control Brown moss	GPP = Light * ETI	0.6	22.60%	0.62	0.67	2.8	0.61
SF burnt	GPP = Light * ETI	0.286	57.80%	0.83	0.44	4.1	0.29
SF cut	GPP = Light * ETI	0.622	49.80%	0.61	0.68	6.4	0.62
SF control *Juncus*	GPP = Light * ETI	0.392	50.70%	0.79	0.44	7.2	0.35
SF control *Sphagnum*	GPP = Light * ETI	0.252	66.50%	0.84	0.38	3.7	0.27
RP control	GPP = Light * WTD	-0.05	118%	1.15	0.35	51.1	-0.39
RP recolonised	GPP = Light * WTD	0.556	13.20%	0.72	0.71	33.1	0.46
TF burnt	GPP = Light * WTD	0.036	72.40%	1.31	0.5	92.5	-0.8
TF control *Cladium*	GPP = Light * WTD	0.504	75.20%	0.73	0.63	19.9	0.45
TF cut	GPP = Light * WTD	0.74	29.80%	0.58	0.78	26.4	0.65
TF control *Phragmites*	GPP = Light * WTD	0.276	63.50%	1.06	0.61	77.8	-0.17
SF control Brown moss	GPP = Light * WTD	-0.035	45.50%	1.4	0.43	91.3	-1.04
SF burnt	GPP = Light * WTD	0.777	21.70%	0.46	0.85	6.5	0.78
SF cut	GPP = Light * WTD	-0.029	66.20%	1.08	0.5	44.9	-0.2
SF control *Juncus*	GPP = Light * WTD	0.549	27.60%	0.71	0.8	21.2	0.48
SF control *Sphagnum*	GPP = Light * WTD	0.578	39.10%	0.66	0.69	14.6	0.55
RP control	GPP = Light * Tsoil * ETI	0.549	67.30%	0.67	0.72	7.9	0.54
RP recolonised	GPP = Light * Tsoil * ETI	0.754	13.90%	0.49	0.8	4.7	0.76
TF burnt	GPP = Light * Tsoil * ETI	0.656	39.50%	0.63	0.63	12.7	0.6
TF control *Cladium*	GPP = Light * Tsoil * ETI	0.765	48.10%	0.48	0.74	4.3	0.77
TF cut	GPP = Light * Tsoil * ETI	0.756	33.50%	0.51	0.74	8.4	0.73
TF control *Phragmites*	GPP = Light * Tsoil * ETI	0.829	43.20%	0.54	0.66	18	0.7
SF control Brown moss	GPP = Light * Tsoil * ETI	0.753	16.40%	0.48	0.78	1.2	0.76
SF burnt	GPP = Light * Tsoil * ETI	0.597	43.40%	0.65	0.63	10	0.57
SF cut	GPP = Light * Tsoil * ETI	0.714	39%	0.52	0.77	2.9	0.72
SF control *Juncus*	GPP = Light * Tsoil * ETI	0.859	25.50%	0.45	0.77	12	0.79

SF control *Sphagnum*	GPP = Light * Tsoil * ETI	0.629	40.30%	0.61	0.67	7.3	0.62
RP control	GPP = Light * Tsoil * WTD	0.471	64.20%	0.72	0.69	16.5	0.46
RP recolonised	GPP = Light * Tsoil * WTD	0.704	12.10%	0.56	0.73	21.6	0.67
TF burnt	GPP = Light * Tsoil * WTD	0	73.80%	1.32	0.49	94.5	-0.8
TF control *Cladium*	GPP = Light * Tsoil * WTD	0.588	73.70%	0.68	0.65	21.1	0.52
TF cut	GPP = Light * Tsoil * WTD	0.806	23.70%	0.47	0.82	16.9	0.77
TF control *Phragmites*	GPP = Light * Tsoil * WTD	0.347	57.40%	0.92	0.62	55.3	0.12
SF control Brown moss	GPP = Light * Tsoil * WTD	-0.03	46.40%	1.41	0.43	93.1	-1.04
SF burnt	GPP = Light * Tsoil * WTD	0.825	19.80%	0.41	0.86	6.2	0.83
SF cut	GPP = Light * Tsoil * WTD	-0.01	70.70%	1.11	0.5	64.5	-0.28
SF control *Juncus*	GPP = Light * Tsoil * WTD	0.591	27.20%	0.7	0.81	25.6	0.5
SF control *Sphagnum*	GPP = Light * Tsoil * WTD	0.692	30.90%	0.57	0.76	15.6	0.66
RP control	GPP = Light * Tsoil * WTD * ETI	0.485	63.10%	0.71	0.7	18.2	0.46
RP recolonised	GPP = Light * Tsoil * WTD * ETI	0.729	11.30%	0.54	0.75	21.1	0.69
TF burnt	GPP = Light * Tsoil * WTD * ETI	-0.042	77.40%	1.32	0.48	99.4	-0.82
TF control *Cladium*	GPP = Light * Tsoil * WTD * ETI	0.598	73.90%	0.68	0.65	22.1	0.53
TF cut	GPP = Light * Tsoil * WTD * ETI	0.81	23.30%	0.46	0.83	16.3	0.78
TF control *Phragmites*	GPP = Light * Tsoil * WTD * ETI	0.282	63%	1.05	0.61	74.8	-0.14
SF control Brown moss	GPP = Light * Tsoil * WTD * ETI	0.427	44.50%	1.31	0.52	66.1	-0.76
SF burnt	GPP = Light * Tsoil * WTD * ETI	0.826	19.80%	0.41	0.86	6.4	0.83
SF cut	GPP = Light * Tsoil * WTD * ETI	0.012	62.20%	1.05	0.54	46.7	-0.14
SF control *Juncus*	GPP = Light * Tsoil * WTD * ETI	0.597	27.90%	0.7	0.8	26.5	0.49
SF control *Sphagnum*	GPP = Light * Tsoil * WTD * ETI	0.695	31.80%	0.58	0.76	17.8	0.65
RP bare soil	RECO = L&T (Tsoil)	0.296	6.34%	0.82	0.63	3.6	0.31
RP control	RECO = L&T (Tsoil)	0.517	26.50%	0.68	0.64	0.2	0.53
RP recolonised	RECO = L&T (Tsoil)	0.465	9.45%	0.71	0.6	0	0.48
TF burnt	RECO = L&T (Tsoil)	0.724	10.90%	0.51	0.75	0.6	0.73
TF control *Cladium*	RECO = L&T (Tsoil)	0.704	19.80%	0.53	0.74	1.6	0.71
TF cut	RECO = L&T (Tsoil)	0.763	11.60%	0.48	0.78	1.4	0.77

TF control *Phragmites*	RECO = L&T (Tsoil)	0.733	16.30%	0.5	0.77	0.9	0.74
SF control Brown moss	RECO = L&T (Tsoil)	0.786	5.70%	0.45	0.79	1.8	0.79
SF burnt	RECO = L&T (Tsoil)	0.65	15.90%	0.59	0.69	3.2	0.65
SF cut	RECO = L&T (Tsoil)	0.581	19.70%	0.64	0.68	1.9	0.59
SF control *Juncus*	RECO = L&T (Tsoil)	0.791	7.97%	0.45	0.8	1	0.8
SF control *Sphagnum*	RECO = L&T (Tsoil)	0.574	16%	0.64	0.69	3.1	0.58
RP bare soil	RECO = L&T (Tsoil) * WTD	0.701	1.42%	0.55	0.76	14.5	0.68
RP control	RECO = L&T (Tsoil) * WTD	0.878	18%	0.49	0.79	23.3	0.75
RP recolonised	RECO = L&T (Tsoil) * WTD	0.686	4.90%	0.67	0.7	32.1	0.52
TF burnt	RECO = L&T (Tsoil) * WTD	0.375	22%	1.07	0.59	50.9	-0.17
TF control *Cladium*	RECO = L&T (Tsoil) * WTD	0.575	29.10%	0.76	0.68	25.8	0.4
TF cut	RECO = L&T (Tsoil) * WTD	0.606	21.70%	0.85	0.67	33.4	0.27
TF control *Phragmites*	RECO = L&T (Tsoil) * WTD	0.645	23.20%	0.67	0.72	25.2	0.54
SF control Brown moss	RECO = L&T (Tsoil) * WTD	-0.01	19.20%	1.47	0.42	97.7	-1.23
SF burnt	RECO = L&T (Tsoil) * WTD	0.756	13.60%	0.54	0.78	14.2	0.7
SF cut	RECO = L&T (Tsoil) * WTD	0.317	29.10%	0.95	0.59	26.8	0.09
SF control *Juncus*	RECO = L&T (Tsoil) * WTD	0.603	13.60%	0.76	0.69	28.6	0.41
SF control *Sphagnum*	RECO = L&T (Tsoil) * WTD	0.551	18.90%	0.72	0.7	17.7	0.47
RP bare soil	CH4 = (ln)Tsoil	0.25	1.20%	0.85	0.52	0.3	0.26
RP control	CH4 = (ln)Tsoil	0.308	86%	0.82	0.53	0.2	0.32
RP recolonised	CH4 = (ln)Tsoil	0.139	2.08%	0.92	0.44	0.2	0.15
TF burnt	CH4 = (ln)Tsoil	0.118	298%	0.93	0.46	0.1	0.13
TF control *Cladium*	CH4 = (ln)Tsoil	0.116	87.80%	0.93	0.46	0.2	0.13
TF cut	CH4 = (ln)Tsoil	0.157	276%	0.91	0.46	0.1	0.17
TF control *Phragmites*	CH4 = (ln)Tsoil	0.15	221%	0.91	0.47	0	0.16
SF control Brown moss	CH4 = (ln)Tsoil	0.352	205%	0.79	0.63	0	0.36
SF burnt	CH4 = (ln)Tsoil	0.175	131%	0.9	0.5	0	0.18
SF cut	CH4 = (ln)Tsoil	0.183	90.20%	0.9	0.5	0.2	0.19
SF control *Juncus*	CH4 = (ln)Tsoil	0.29	140%	0.83	0.6	0.2	0.3

SF control *Sphagnum*	CH4 = (ln)Tsoil	0.504	55.10%	0.69	0.66	0.3	0.51
RP bare soil	CH4 = Tsoil + WTD	0.262	0.65%	0.84	0.52	0.8	0.28
RP control	CH4 = Tsoil + WTD	0.913	35.90%	0.29	0.82	0.1	0.91
RP recolonised	CH4 = Tsoil + WTD	0.525	1.15%	0.67	0.71	0.1	0.54
TF burnt	CH4 = Tsoil + WTD	0.213	280%	0.88	0.54	0	0.22
TF control *Cladium*	CH4 = Tsoil + WTD	0.14	84.90%	0.92	0.49	0.5	0.15
TF cut	CH4 = Tsoil + WTD	0.313	255%	0.82	0.56	0.1	0.32
TF control *Phragmites*	CH4 = Tsoil + WTD	0.214	206%	0.87	0.54	0	0.22
SF control Brown moss	CH4 = Tsoil + WTD	0.362	207%	0.79	0.63	0	0.37
SF burnt	CH4 = Tsoil + WTD	0.183	129%	0.89	0.51	0.5	0.19
SF cut	CH4 = Tsoil + WTD	0.203	87.70%	0.88	0.51	0.1	0.21
SF control *Juncus*	CH4 = Tsoil + WTD	0.319	137%	0.81	0.61	0.1	0.33
SF control *Sphagnum*	CH4 = Tsoil + WTD	0.519	54.10%	0.68	0.67	0	0.53
RP bare soil	CH4 = Tsoil + PAR	0.24	1.33%	0.86	0.47	0.8	0.25
RP control	CH4 = Tsoil + PAR	0.486	78.50%	0.71	0.62	0.3	0.49
RP recolonised	CH4 = Tsoil + PAR	0.236	1.93%	0.86	0.54	0.5	0.25
TF burnt	CH4 = Tsoil + PAR	0.25	219%	0.85	0.53	0.1	0.26
TF control *Cladium*	CH4 = Tsoil + PAR	0.108	72.50%	0.93	0.45	0.6	0.12
TF cut	CH4 = Tsoil + PAR	0.248	224%	0.86	0.48	0.1	0.26
TF control *Phragmites*	CH4 = Tsoil + PAR	0.171	128%	0.9	0.5	0	0.18
SF control Brown moss	CH4 = Tsoil + PAR	0.63	92.40%	0.6	0.73	0.2	0.64
SF burnt	CH4 = Tsoil + PAR	0.322	77.50%	0.81	0.54	0.3	0.33
SF cut	CH4 = Tsoil + PAR	0.293	60.20%	0.83	0.59	0	0.3
SF control *Juncus*	CH4 = Tsoil + PAR	0.27	138%	0.84	0.57	0.2	0.28
SF control *Sphagnum*	CH4 = Tsoil + PAR	0.524	43.30%	0.68	0.67	0.5	0.53
RP bare soil	CH4 = Tsoil + PAR + WTD	0.268	0.62%	0.83	0.54	1	0.29
RP control	CH4 = Tsoil + PAR + WTD	0.97	18.60%	0.17	0.91	0.4	0.97
RP recolonised	CH4 = Tsoil + PAR + WTD	0.532	1.17%	0.67	0.7	0.3	0.54
TF burnt	CH4 = Tsoil + PAR + WTD	0.287	194%	0.83	0.61	0	0.3

TF control *Cladium*	CH4 = Tsoil + PAR + WTD	0.117	73%	0.93	0.46	0	0.13
TF cut	CH4 = Tsoil + PAR + WTD	0.364	198%	0.79	0.58	0.1	0.37
TF control *Phragmites*	CH4 = Tsoil + PAR + WTD	0.198	126%	0.88	0.54	0.1	0.21
SF control Brown moss	CH4 = Tsoil + PAR + WTD	0.664	88.50%	0.57	0.75	0.2	0.67
SF burnt	CH4 = Tsoil + PAR + WTD	0.322	77.20%	0.81	0.54	0.2	0.33
SF cut	CH4 = Tsoil + PAR + WTD	0.293	60%	0.83	0.59	0	0.3
SF control *Juncus*	CH4 = Tsoil + PAR + WTD	0.329	134%	0.81	0.58	0.3	0.34
SF control *Sphagnum*	CH4 = Tsoil + PAR + WTD	0.524	43.30%	0.68	0.67	0.4	0.53
RP bare soil	CH4 = exp(Tsoil)	0.148	1.53%	1.09	0.4	108	-0.2
RP control	CH4 = exp(Tsoil)	0.111	312%	0.94	0.35	10.3	0.12
RP recolonised	CH4 = exp(Tsoil)	0.126	2.13%	0.92	0.4	4.1	0.14
TF burnt	CH4 = exp(Tsoil)	0.099	316%	0.94	0.39	3.1	0.11
TF control *Cladium*	CH4 = exp(Tsoil)	0.095	92.60%	0.94	0.39	2.3	0.1
TF cut	CH4 = exp(Tsoil)	0.123	300%	0.93	0.36	3.7	0.13
TF control *Phragmites*	CH4 = exp(Tsoil)	0.12	233%	0.93	0.38	4	0.13
SF control Brown moss	CH4 = exp(Tsoil)	0.421	206%	0.75	0.63	4.5	0.43
SF burnt	CH4 = exp(Tsoil)	0.137	145%	0.92	0.39	3.8	0.14
SF cut	CH4 = exp(Tsoil)	0.15	95.90%	0.91	0.42	1.9	0.16
SF control *Juncus*	CH4 = exp(Tsoil)	0.262	146%	0.85	0.55	4.7	0.27
SF control *Sphagnum*	CH4 = exp(Tsoil)	0.435	61.60%	0.75	0.58	2.7	0.44

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Appendix B

Table B1. Friedman test summary statistics for comparison of soil temperature between microforms. Bold values show significant levels (p ≤ 0.05)

Test-score χ^2	4.27
P value	**< 0.001**

Post-hoc test results (p values)	
RW-16_ridge – Bund	**< 0.001**
RW-16_trough - Bund	**< 0.001**
RW-15_ridge – Bund	0.710
RW-15_trough - Bund	0.958
RW-16_trough – RW-16_ridge	1.000
RW-15_ridge – RW-16_ridge	**0.032**
RW-15_trough – RW-16_ridge	**0.005**
RW-15_ridge – RW-16_trough	**0.023**
RW-15_trough – RW-16_trough	**0.003**
RW-15_trough – RW-15_ridge	0.978

Table B2. Monthly count of collars inundated during measurement campaigns. Number in brackets represent the total number of collars measured in each microform during each measurement campaign.

	RW-15		RW-16	
	trough	**ridge**	**trough**	**ridge**
July 2015	2 (6)		1 (7)	
September 2015	1 (7)			
October 2015	2 (12)		1 (14)	
November 2015			2 (7)	
February 2016	1 (5)		2 (7)	
March 2016	1 (5)		3 (7)	
April 2016	3 (16)		11 (19)	
May 2016			4 (7)	
June 2016	2 (6)		6 (6)	
July 2016	8 (8)		7 (10)	2 (16)
August 2016	3 (3)		3 (3)	
September 2016			5 (6)	1 (7)
October 2016	2 (6)		6 (6)	1 (7)
Total	**25**	**0**	**51**	**4**

Fig. B1: Manual chamber measurement set up; Clear Perspex chamber (NEE) equipped with fan and pressure equalisation balloon attached to the Los Gatos Ultra Portable GHG Analyser. Picture taken on the bund in RW-15.
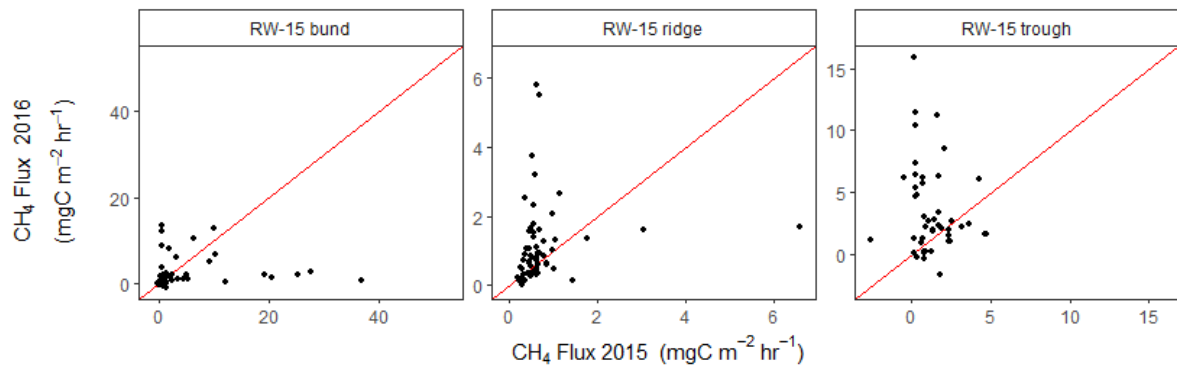
Fig. B2: Measured CH4 fluxes from the same collar and month from 2015 and 2016 for the a) ridge and b) trough microforms within the RW-15 treatment. Months included are June – October. The grey line represents the 1:1 line
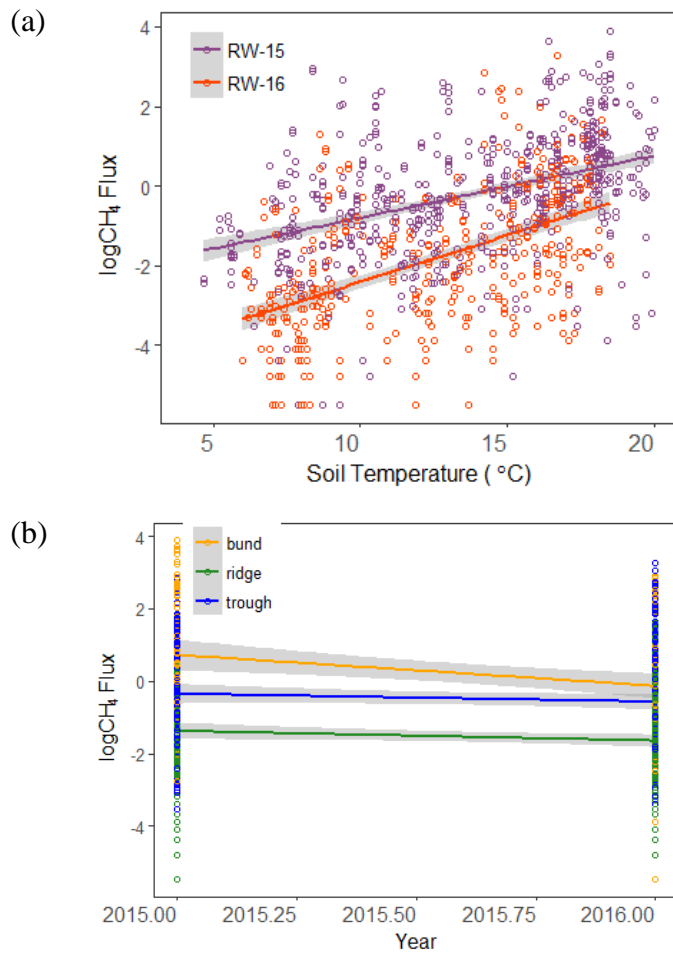
(a)

(b)

Fig. B3. Mixed effect model analysis interaction plot, with logCH₄ flux as the response variable, and interaction between (a) soil temperature and treatment; and (b) Year and microform.

(a)

(b)
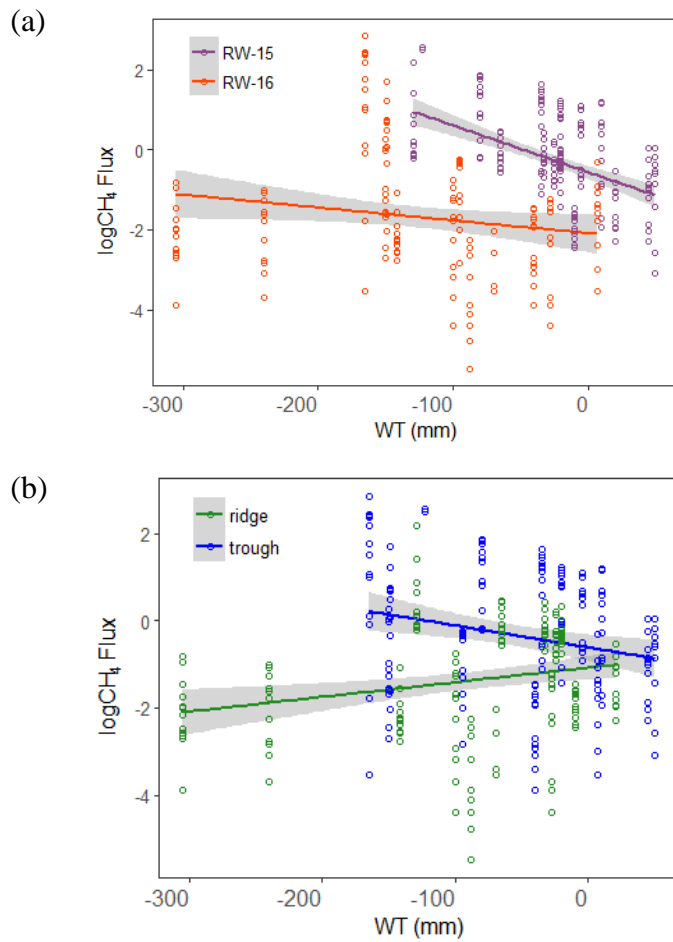
Fig. B4. Mixed effect model analysis interaction plot, with logCH$_4$ flux as the response variable, and interaction between (a) WTD and treatment; and (b) WTD and microform.
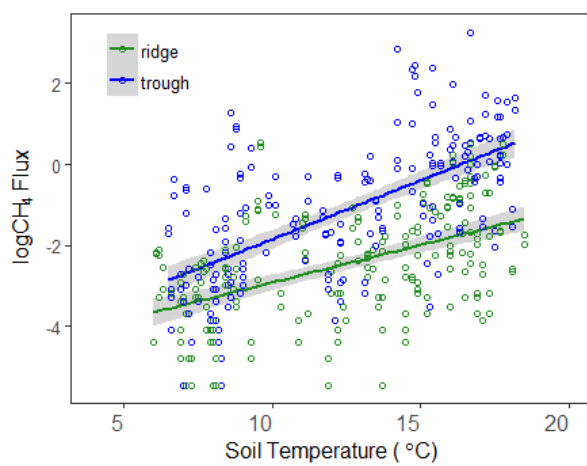

Fig. B5. Mixed effect model analysis interaction plot, with logCH$_4$ flux as the response variable, and interaction between soil temperature and microform.
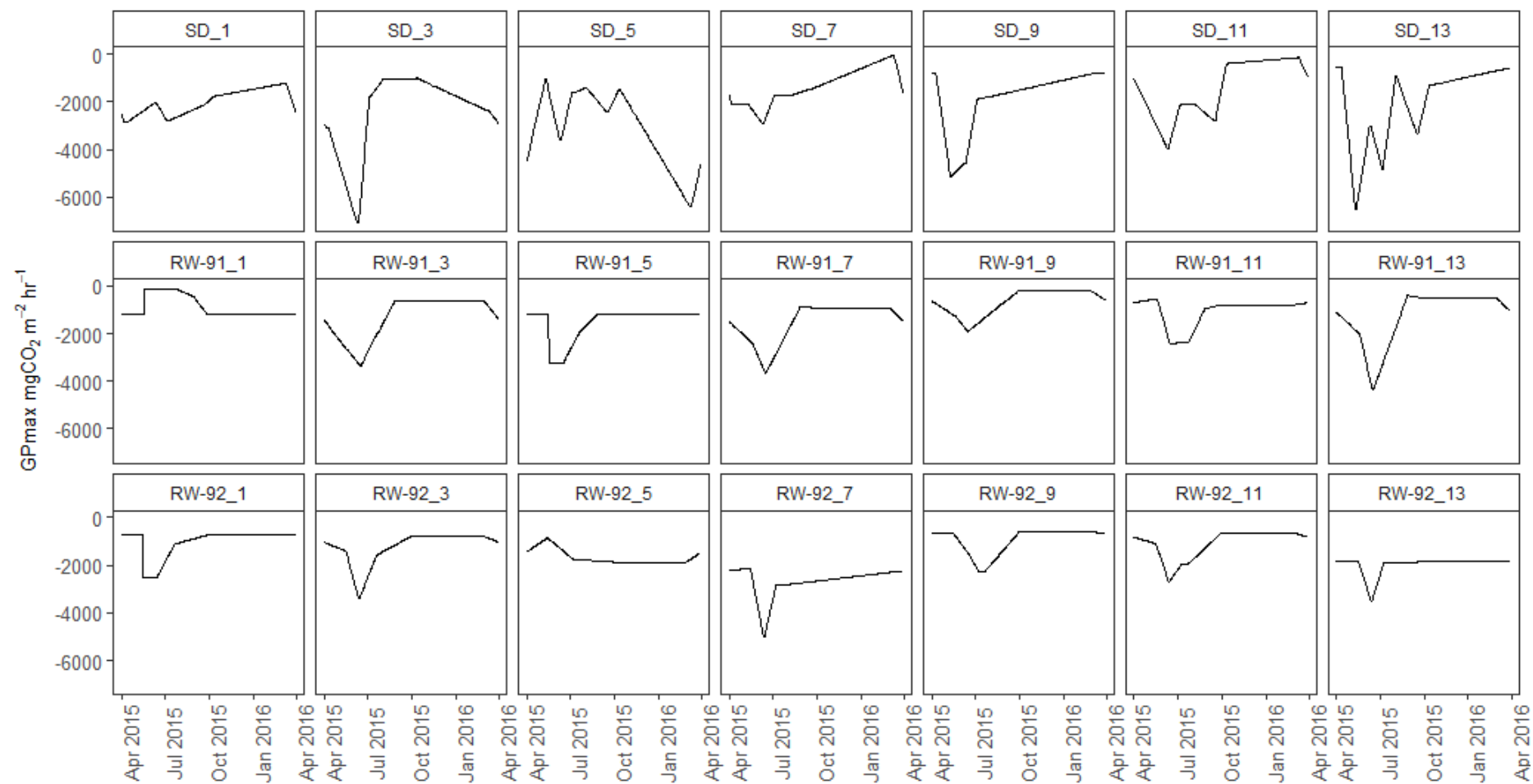
Appendix C

Figure C1a. Interpolated GPmax model parameters for individual measurement collars.
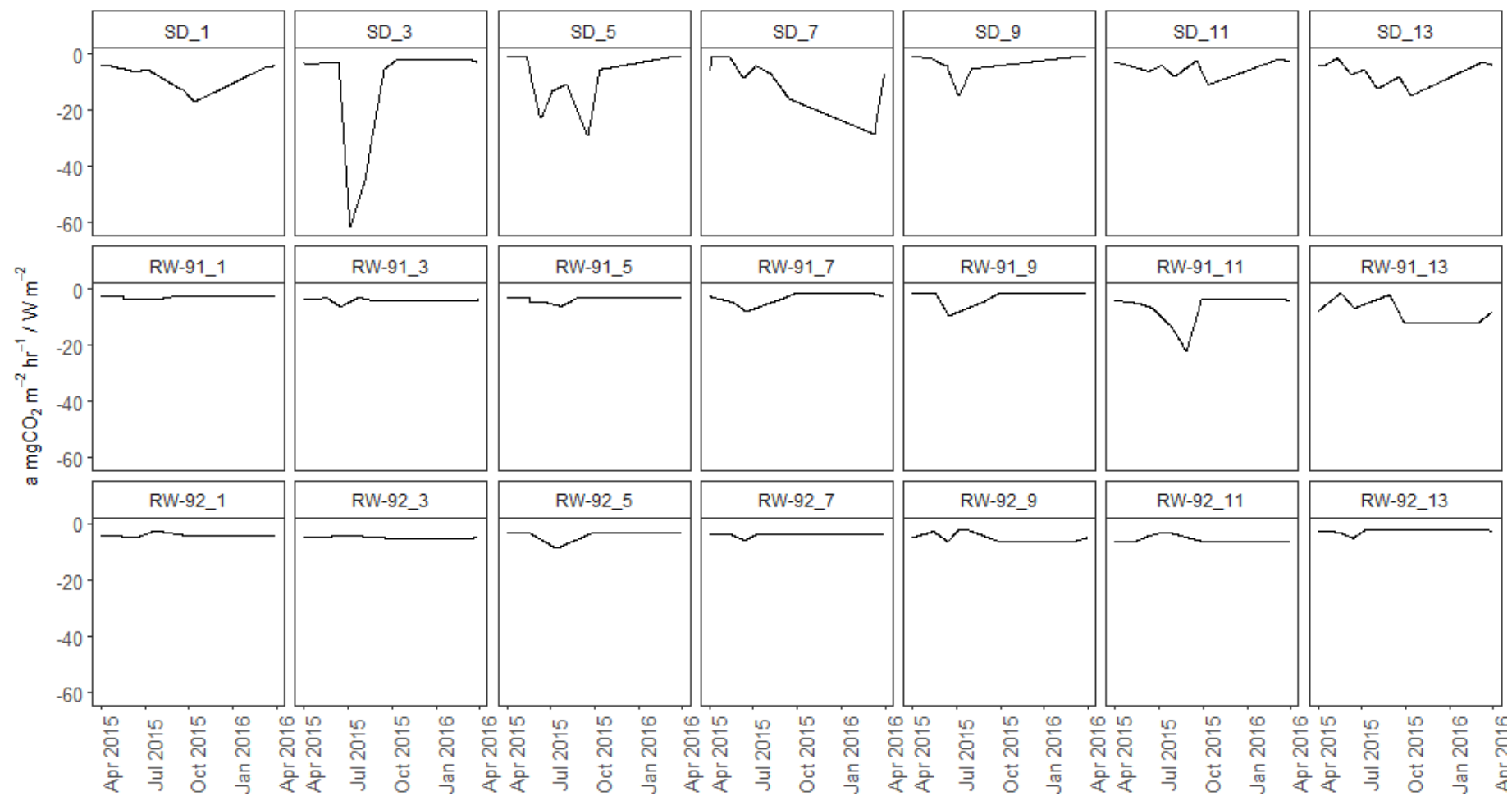
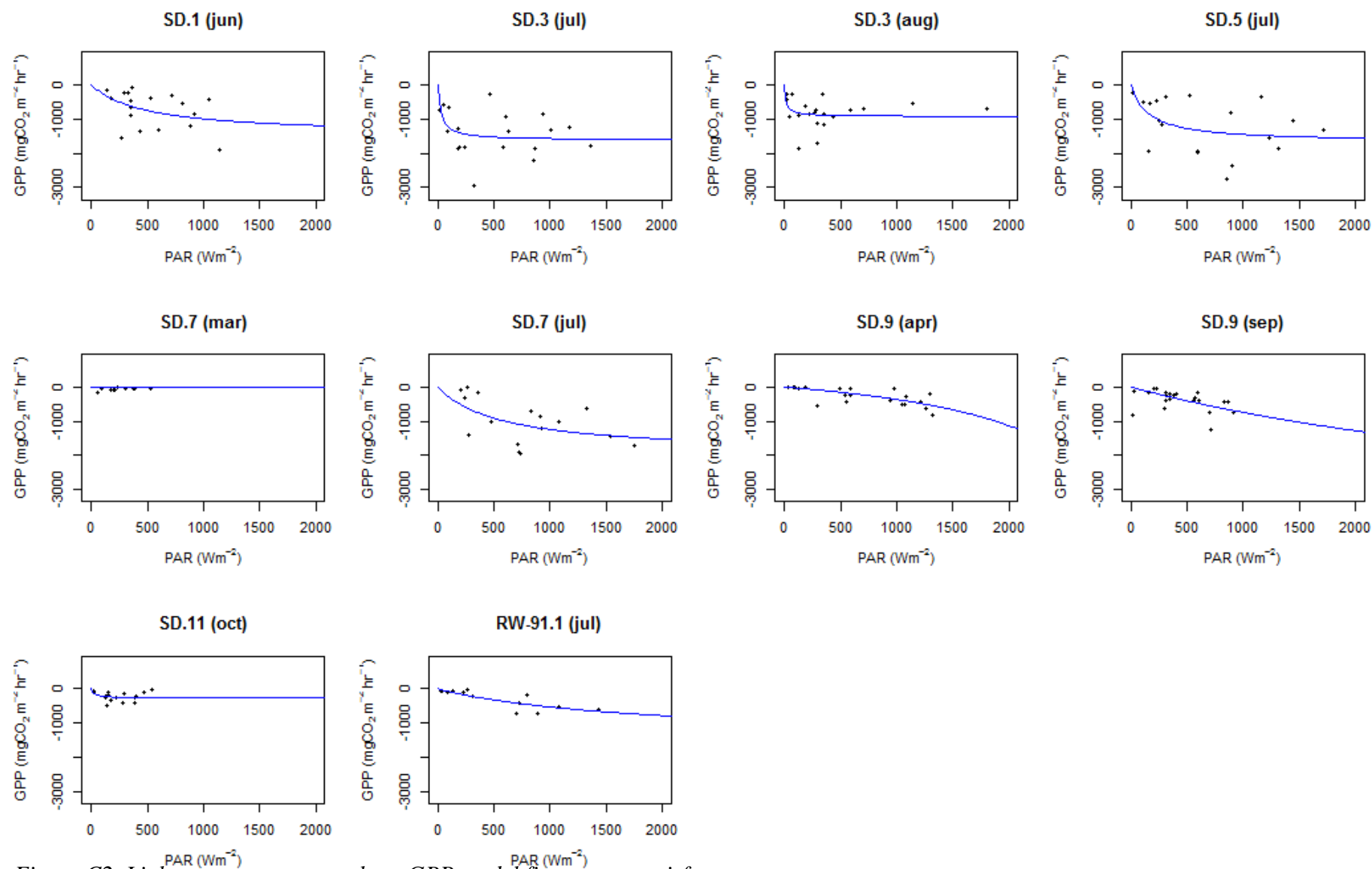*Figure C1b. Interpolated alpha model parameters for individual measurement collars.*

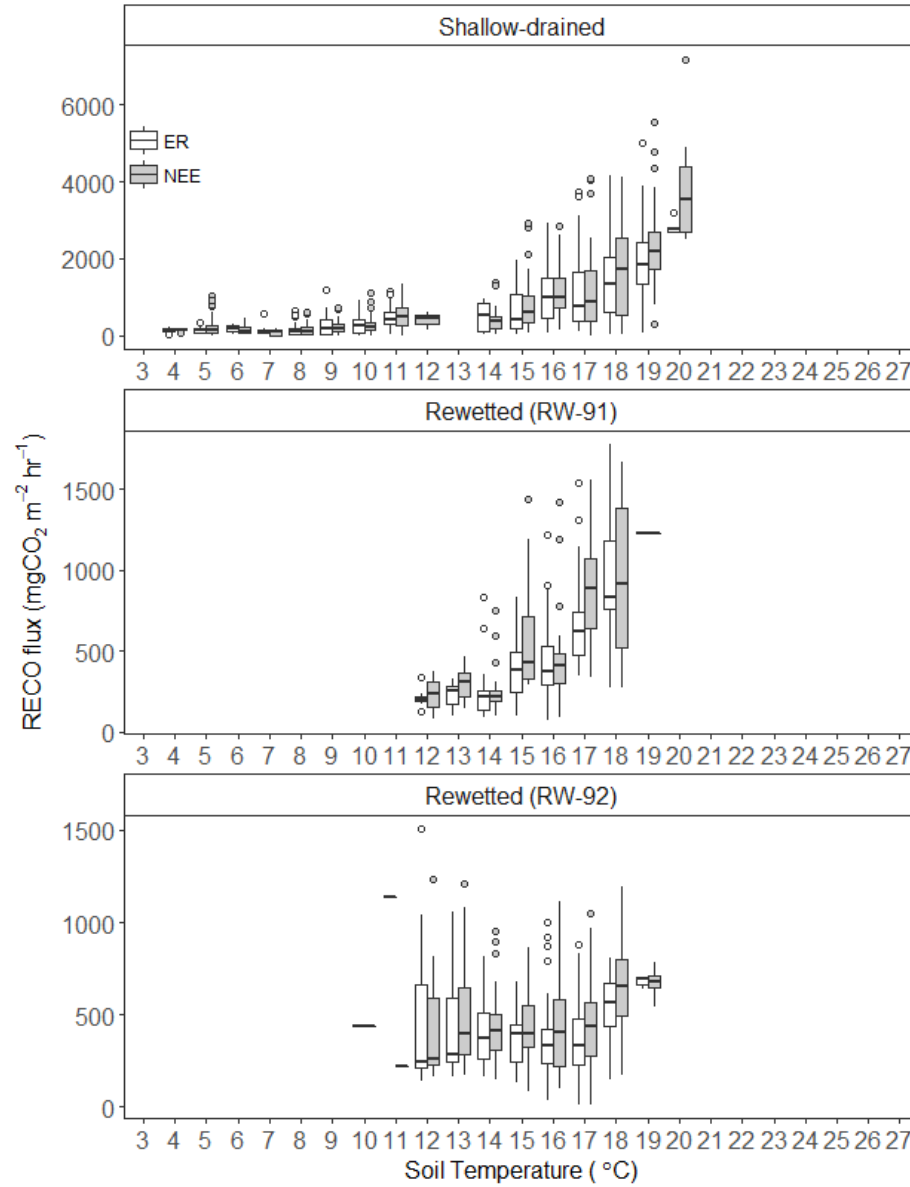*Figure C2. Light response curves where GPP model fits were unsatisfactory.*

Figure C3. Nocturnal R_{ECO} fluxes from transparent (NEE) and opaque (R_{ECO}) chambers with soil temperature.
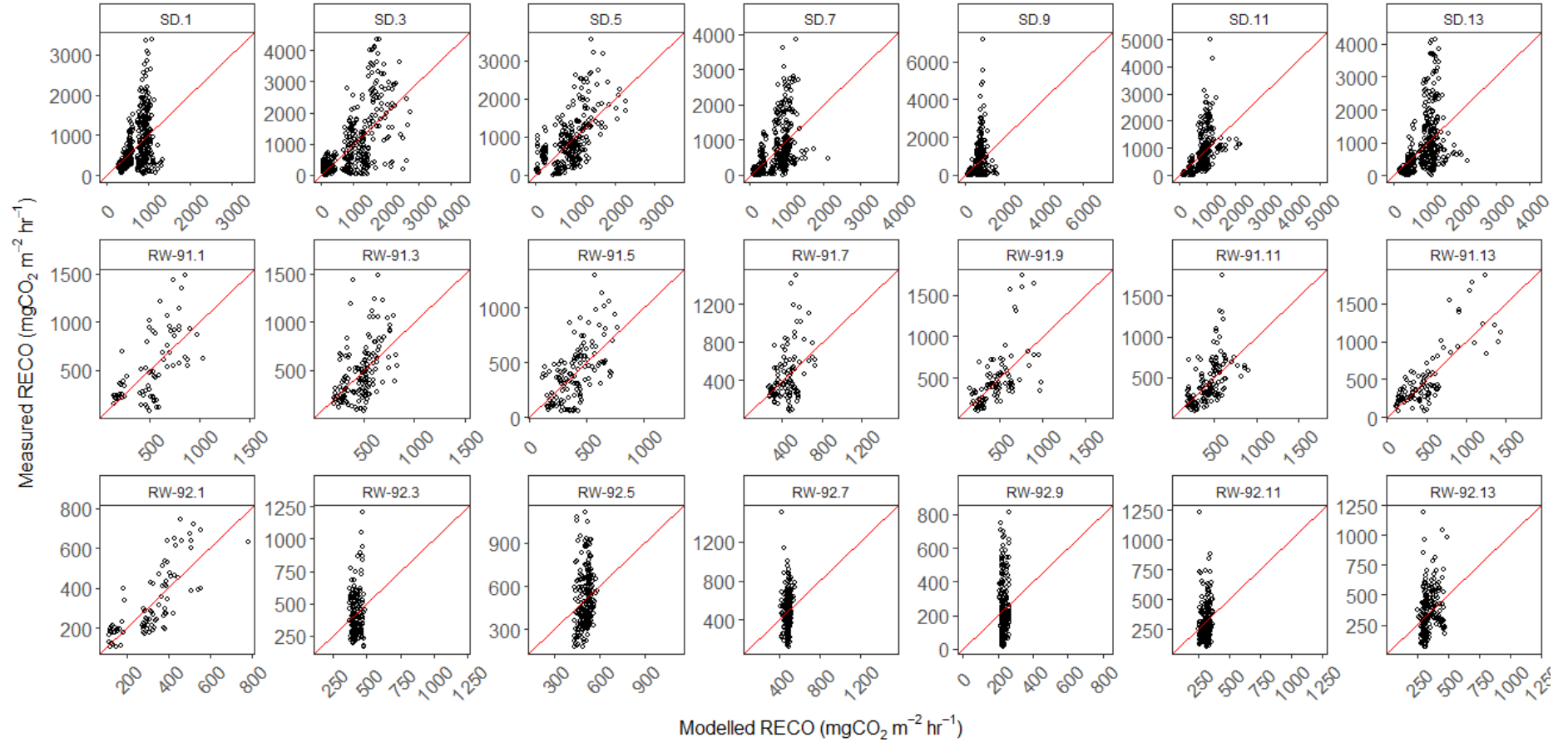
Figure C4. Scatter plot of measured R$_{ECO}$ against modelled R$_{ECO}$ fluxes. Solid line indicates the 1:1 line.
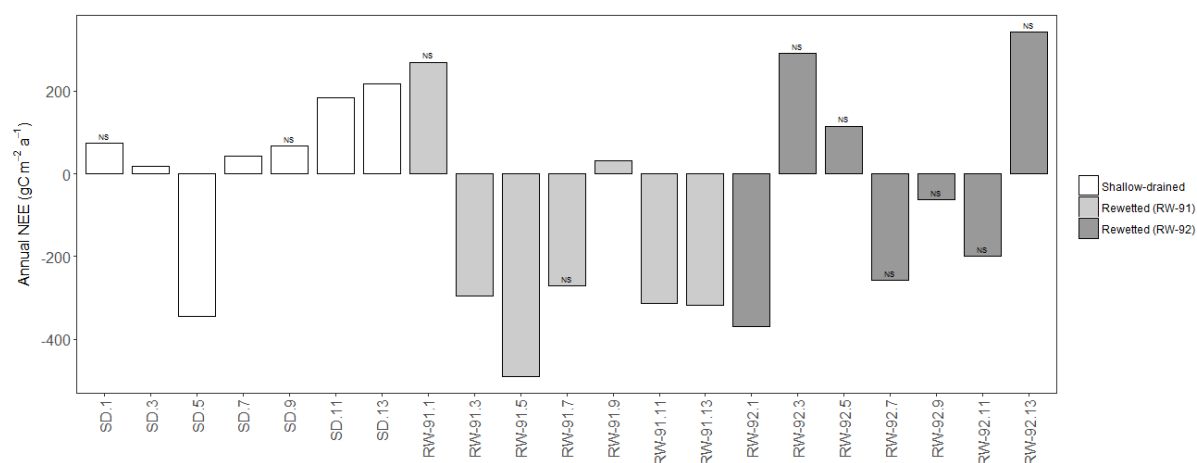
Figure C5. Modelled annual NEE flux values for all collars. [NS] denotes a non-satisfactory model was achieved either for R_{ECO} or GPP modelling.
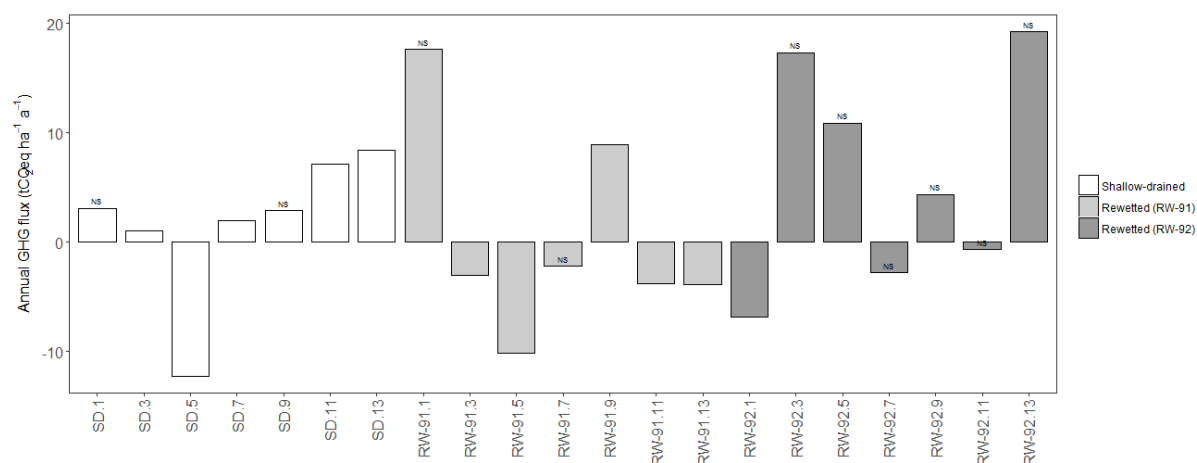

Figure C6. Annual net GHG balance for all collars; [NS] denotes unsatisfactory GPP or R_{ECO} models used.

Appendix D

Table D1. Annual NEE flux estimates for land use categories; conservation managed fens; managed grasslands and bare soil/extraction, from the studies within this thesis, the Lowland Peat project (Evans et al., 2017a) and published emission factors.

Thesis	Lowland Peat Project	Emission Factors
NEE: tCO_2-C ha^{-1} yr^{-1}		
Conservation managed fen		
AF-SF Control -1.30	-1.43 (-3.38 to +0.10)	*Nutrient rich - Rewetted*
Burnt 0.28		+0.50 (-0.71 to +1.71)[1]
AF-TF Control -1.78		
Burnt -3.05		
Managed grassland		
AF-RP Control -2.05	+2.85 (+1.23 to +4.33)	*Nutrient rich SD*
AF-TF Hand-cut -2.66		+3.6 (+1.8 to +5.4)[2]
AF-SF Hand-cut 2.35		+4.6 ±3 (Tiemeyer et al., 2016)
Cors Fochno		*Nutrient rich SD - Rewetted*
SD +0.23 (-3.45 to +2.18)		+0.26 (Wilson et al., 16)
RW -2.93 (-4.90 to +0.31)		
		Nutrient poor Drained
		+5.3 (+3.7 to +6.9)[3]
		+3.5 ±2.5 (Tiemeyer et al., 2016)
		Nutrient poor D - Rewetted
		-0.33 (Wilson et al., 2016)
Bare soil/extraction		
AF-RP	*Extraction*	*Extraction*
Bare soil +1.84	+0.91 (-0.41 to +2.23)	+2.8 (+1.1 to +4.2)[2]
Recolonised (rewetted) -		
0.02	*Extraction - Rewetted*	*Nutrient poor – Rewetted*
	+1.40 (+1.38 to +1.43)	-0.23 (-0.59 to -0.09)[4]
		Extraction – Rewetted
		-0.33 (Wilson et al., 2016)

Table D2. Annual NEE flux estimates for land use categories; conservation managed fens; managed grasslands and bare soil/extraction, from the studies within this thesis, the Lowland Peat project (Evans et al., 2017a) and published emission factors.

Thesis	Lowland Peat Project	Emission Factors
CH4 flux: kg CH4-C ha-1 yr-1		
Conservation managed fen		
AF-SF Control +119	+129 (+66 to +157)	*Nutrient rich - Rewetted*
$\quad\quad$ Burnt + 46		+216 (0 to +856)[1]
AF-TF Control +120		
$\quad\quad$ Burnt + 115		
Managed grassland		
AF-RP Control +56	-1 (-2 to 0)	*Nutrient rich SD*
AF-TF Hand-cut +130		+29 (-2 to +61)[2]
AF-SF Hand-cut + 55		+79.8 ±104 (Tiemeyer et al.,
Cors Fochno		2016)
$\quad$ SD +11.9		*Nutrient rich SD - Rewetted*
$\quad$ RW +217 (+200 to +233)		+236 (Wilson et al., 2016)
		Nutrient poor Drained
		16.3 ±24.4 (Tiemeyer et al., 2016)
		Nutrient poor D - Rewetted
		+90 (Wilson et al., 2016)
Bare soil/extraction		
AF-RP	*Extraction*	*Extraction*
$\quad$ Bare soil +1	+224 (+119 to +328)	+4.6 (+1.2 to +10.5)[2]
$\quad$ Recolonised (rewetted) +1		
	Extraction – Rewetted	*Nutrient poor – Rewetted*
	+0.7 (-0.4 to +1.8)	+92 (3 to +445)[4]
		Extraction – Rewetted
		+30 (Wilson et al., 2016)

1: Temperate nutrient rich rewetted organic soil (Table 3.1, IPCC, 2014)

2: Shallow drained nutrient rich grassland (Table 2.1, IPCC, 2014)

3: Nutrient poor grassland, drained (Table 2.1, IPCC, 2014)

4: Temperate, nutrient poor rewetted organic soil (Table 3.1, IPCC, 2014)