# Defra Lowland Peat 2: Managing agricultural systems on lowland peat for decreased greenhouse gas emissions whilst maintaining agricultural productivity. Report to Defra for Project SP1218

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March 2023

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# Contents

# 1. Policy Summary

I. Around one third of English peatland is located in the lowlands. The majority of this area has been converted to drainage-based agriculture, much of it for high-value horticulture and arable farming, along with intensive beef and dairy production. While contributing significantly to the UK's economy and food security, lowland peat drainage has resulted in peat oxidation, long-term land subsidence, the depletion of a major carbon store, and large and continuing $CO_2$ emissions. As a result, agriculturally drained peatlands are thought to be responsible for around half of UK peatland greenhouse gas (GHG) emissions, and 2% of all UK GHG emissions.

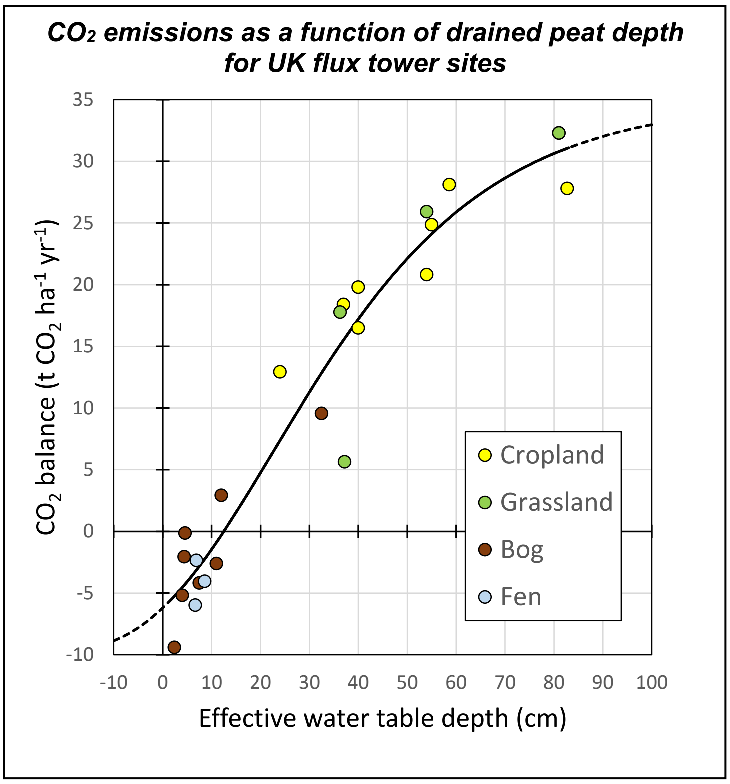
II. This report describes the results of the 'Lowland Peat 2' project, which ran from 2019 to 2023. Following the earlier Lowland Peat 1 project (2014-2017) which generated fundamental data on GHG emissions from a range of contrasting lowland peat sites across England and Wales, the project focused primarily on developing and testing options to mitigate these emissions, whilst maintaining the agricultural productivity of lowland peatlands. The project coincided with a number of major peatland-related policy initiatives across the UK, including the Committee on Climate Change's 6th Carbon Budget and Net Zero Strategy, the England Peat Action Plan, and the Defra Lowland Agricultural Peat Task Force. Interim results from the project have fed into many of these initiatives, as well as to an update of emission factors for cropland and grassland on peat in the UK's National Atmospheric Emissions Inventory.

III. A detailed review was undertaken of the opportunities and challenges for paludiculture (wetland-based agriculture) as a potential emissions mitigation measure. The review concluded that while a wide range of paludiculture options exist, their efficacy and economic viability remain largely untested at scale in the UK, and further work is needed to develop markets and supply chains. The most promising options include production of reed for thatch (for which existing demand is currently met via imports), incorporation of wetland biomass in building materials, and production of *Sphagnum* moss as an alternative to peat in horticulture. Given that these are all non-food crops, it is important that any expansion of paludiculture does not displace the GHG emissions and other environmental impacts of food production to other regions or countries. At present there are few proposed food crops that could be grown via paludiculture, although wetlands are not intrinsically unsuitable for food production, as illustrated by the global importance of rice as a staple crop. The recommendations of the report for further field-scale trials to identify and overcome barriers to large-scale paludiculture have since been addressed through the implementation of Defra's Paludiculture Exploration Fund. Until these barriers have been overcome, there remains an urgent need to mitigate GHG emissions from agriculturally drained peatlands.

IV. A second review scoped out the societal impacts of lowland peat drainage, with a focus on the impacts of long-term subsidence. These include damage to roads, pipelines, communication and energy supply networks, houses and archaeological records, as well as the costs of building and maintaining drainage and flood defences in areas that are now below river and sea-level. In many cases above the level of the land. At present many of these costs are hidden, for example in wider local authority maintenance budgets, but they are may be in the £10s to £100s of millions per year (not including the societal costs of GHG emissions). Reducing these costs and associated risks would require either removing the hazard (e.g. raising water levels to halt subsidence) or planning future infrastructure to reduce exposure (e.g. not building in areas of active subsidence) or vulnerability (e.g. designing infrastructure that is resilient against subsidence).

V. The project, along with aligned work for BEIS (now DESNZ) and UK Research and Innovation has maintained and expanded what is believed to be the largest network of flux towers on agricultural peatlands globally, providing near-continuous measurements of $CO_2$ emissions, and in some cases $CH_4$ and $N_2O$ emissions. A synthesis of these data was published in the journal Nature in 2021, which has since been updated with new data that reinforce the importance of drainage depth as the primary control on peatland emissions. The analysis suggests that, in general terms, every 10 cm reduction in annual average water table depth within the peat would reduce $CO_2$ emissions by ~5 t $CO_2$ $ha^{-1}$ $yr^{-1}$.

VI. Higher $CH_4$ emissions may be a concern in some circumstances, such as nutrient-rich sites with extensive standing water. However high emissions from agricultural fields are unlikely unless average water tables rise to within 20 cm of the peat surface, and will not outweigh the benefits of reduced $CO_2$ emissions until water table depth is < 10 cm.




VII. There is a risk of high $N_2O$ emissions from agricultural peatlands (both conventionally managed and with higher water table management) where crops are heavily fertilised and irrigated. Fully re-wetted peatlands are unlikely to emit $N_2O$.

VIII. Field-scale high water table trial experiments undertaken at two sites during the project, based on paired (intervention versus control) flux towers, support the inference from the wider flux network that raising water levels within cultivated peatlands will effectively reduce $CO_2$ emissions. However, implementing high water level management at a field scale during the severe 2022 drought and heatwave was challenging, and it appears that a sudden reduction in water levels at one of the sites led to a substantial (~25%) reduction in wheat yield. Nevertheless, these initial results suggest that raising water levels could help to mitigate peat GHG emissions, and if correctly optimised may not lead to yield declines.

IX. A plot-scale trial of the emissions mitigation potential of increased surface irrigation did not show clear benefits, with both $CO_2$ and $N_2O$ emissions increasing from the irrigated plots. The experiment was undertaken at the peak of the 2022 heatwave, when soils were exceptionally dry, and may therefore have alleviated moisture limitations on microbial processes, rather than wetted the soil sufficiently to constrain emissions. This interpretation was supported by higher lettuce yields in the irrigated plots, suggesting field moisture levels were sub-optimal during the experiment. Nevertheless, the data currently available do not show clear evidence for emissions mitigation via surface irrigation

X. A range of other agricultural mitigation measures were considered during the project, and as part of linked PhD research. In general, measures that reduce base soil exposure and disturbance during dry periods, such as cover crops, reduced tillage and soil stabilisers, can be expected to reduce peat loss via wind erosion. Incorporation or crop residues appears to offer limited long-term benefits in terms of retaining carbon in the soil, but equally there is little evidence that it leads to accelerated decomposition ('priming') of existing peat organic matter. Incorporation of carbon in more resistant forms such as biochar holds more promise as a mitigation measure, but requires further testing. In general, few if any of these 'regenerative' farming measures are likely to reduce overall peat GHG emissions unless water levels are raised. Where water levels *are* raised, they may deliver additional mitigation, as well as wider environmental benefits.

XI. Based on flux data from multiple sites, collected over multiple years, we found limited evidence that crop type (including a range of salad and root vegetables, cereals and ley grass) directly affects the amount of $CO_2$ released at any given water table and peat depth. Thinner ('wasted') peat soils have lower (but still substantial) $CO_2$ emissions compared to deeper peat, and permanent grasslands may have somewhat lower $CO_2$ emissions, but further measurements are needed (and the impact of livestock $CH_4$ emissions needs to be considered). In general, crop type may be more important in determining the amount of mitigation that could be achieved in future; for example leafy salad crops could be grown at higher water levels than root vegetables, and some cereal crops may be more tolerant of wetter conditions than others.

XII. We did not observe a strong impact of the 2022 heatwave on $CO_2$ emissions. This is likely because most agricultural peatlands are managed to similar water levels in all years (often below the base of the peat), so the drought had a limited additional impact. It is also possible that soils became so dry that the microbial processes driving $CO_2$ emissions became moisture-limited near the peat surface, offsetting increased emissions from depth.

XIII. The practical challenges for mitigating emissions from lowland agricultural peatlands remain substantial. A survey of farmers from four lowland peat regions of England indictaed that mitigation measures deemed most effective by peatland experts (typically those involving raised water levels) tended to be viewed as the least practical or economic by farmers, whereas those measures favoured by farmers typically delivered limited mitigation. Some measures aimed at reducing wind erosion or improving nitrogen use efficiency to reduce $N_2O$ emissions could deliver modest mitigation whilst also having economic benefits, but in general there are few environmental and economic 'win wins' for farm businesses at present.

XIV. A follow-on focus group study indicated that a substantial proportion of farmers would consider implementing wetter management practices if it were financially viable for them to do so. However they also highlighted the practical challenges, particularly for thinner peat soils, and the need for better water storage and distribution infrastructure to enable different management to take place. They also highlighted the need for long-term policy commitments to providet financial security sufficient to outweigh the risks of management transition, such as the need to invest in new farm machinery to operate on wetter soils.

XV. An initial economic analysis of different management options for thick and wasted peat confirmed that, given the income streams currently available for farmers, there are few if any financially appealing alternatives to drainage-based horticulture and cereal production, largely because the environmental costs (including GHG emissions) are external to the farm business. Implementing solar farming on re-wetted peat offers some potential for generating comparable levels of income without accompanying emissions, and could also support food production via controlled environment agriculture. However, to date all solar farms on peat operate with continued land-drainage, and thus offer no direct mitigation of peat $CO_2$ emissions.

XVI. A qualitative assessment of the environmental, societal and financial costs and benefits of alternative land-management options for lowland peat highlighted the difficulty of avoiding a trade-off between agricultural incomes and food production on the one hand, and environmental benefits (or reduced environmental impacts) on the other. Halting food production on peat soils risks displacing these impacts elsewhere. However a range of options do now exist to maintain food production within more sustainable, lower-emitting peat landscapes via improved water management, and a range of traditional and innovative land-use options, from paludiculture for reed production to controlled environment agriculture supported by renewable energy production. Achieving these more diverse and sustainable lowland peat landscapes will require new policies, a re-assessment of current regulatory and subsidy regimes, and a combination of public and private-sector investment.

# 2. Introduction

Of the 670,000 ha of peat in England, almost half are in the lowlands. Approximately 248,000 ha of lowland peat, representing over a third of all English peatlands, is utilised for agriculture (Figure 2.1). Of this, 183,000 ha (74%) is managed as cropland, including a mixture of cereals and horticultural products, and 65,000 ha (26%) as grassland, primarily for beef and dairy production.

The productive agricultural use of peat almost always involves drainage, which first took place at scale in England during the 17th century and has become increasingly intensive over time as technology has advanced from wind and then to electric pumping systems. Consequences of the long-term drainage of peatlands include peat wastage, or subsidence, typically in the region of 10 to 30 mm $yr^{-1}$ under cropland, and somewhat lower under grassland.

In some areas, this process has led to over 4 m of cumulative peat loss, leading ultimately to the formation of 'wasted' peat soils, in which the original thick peat has been reduced to a thin plough layer comprising a mix of remnant peat and mineral soil. Over England as a whole, approximately 72% of the cropland area is on wasted peat, and 45% of the grassland area. Bearing in mind that the extent of wasted peat was largely mapped during or prior to the 80s (e.g. Burton and Hodgson, 1987), the current area of wasted peat is almost certainly under-estimated (Holman and Kechavarzi, 2011).

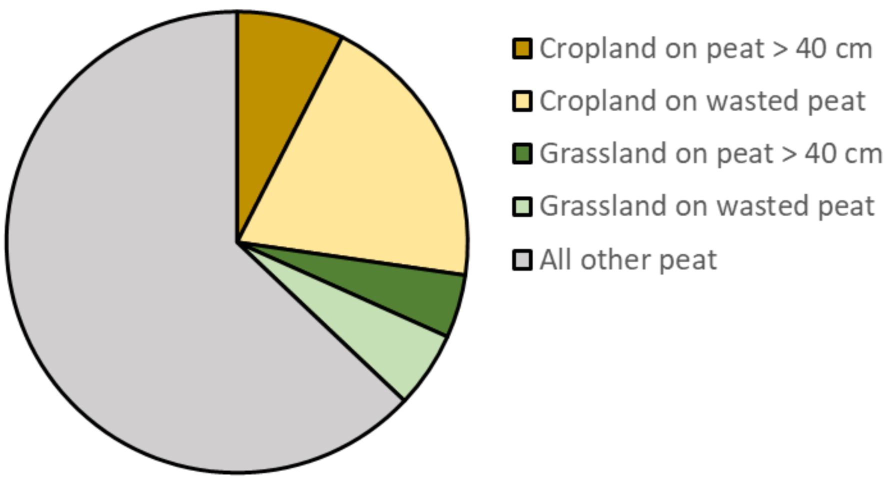



***Figure 2.1**. Cropland and grassland on lowland peat as a proportion of the total English peat area (data from Evans et al., 2017).*

A direct consequence of agricultural peat drainage, and a major contributor to long-term subsidence, is oxidation of peat organic matter. Peat forms because organic matter (net primary productivity) produced by plant photosynthesis and deposited into wetland environments is protected from decomposition under waterlogged conditions. When exposed to oxidation by drainage, this organic matter can decompose rapidly, giving rise to high rates of $CO_2$ emissions. Although drainage can reduce natural wetland methane ($CH_4$) emissions, the creation of drainage ditches creates new 'hotspots' of $CH_4$ emission, while the application of nitrogen-rich fertilisers and manures can give rise to high rates of nitrous oxide ($N_2O$) emission. In work undertaken to support the inclusion of managed peatlands in the UK's National Atmospheric Emissions Inventory (Evans et al., 2017), it was estimated that at a UK scale, cropland on peat is responsible for a 6.9 Mt $CO_2$e $yr^{-1}$ of total GHG emissions (where $CO_2$ equivalent emissions, $CO_2$e, are calculated as the sum of $CO_2$, $CH_4$ and $N_2O$ emissions based on 100 year Global Warming Potentials, calculated at the time using IPCC AR4 GWP values). Grassland on peat was estimated to contribute a similar total emission at a UK scale, although this emission is distributed across all four UK countries, whereas cropland emissions are almost entirely produced in England. While the values reported in the UK inventory have been revised since this work was originally done – partly as a result of work undertaken during this project – it remains the case that cropland and grassland on organic soils are the major sources of overall GHG emissions from peat in England, and significant contributors to total GHG emissions at both an England and a whole-UK scale.

The UK Government's 25 Year Environment Plan identifies drainage-based agriculture on peat as inherently unsustainable, and includes strategies to reduce these emissions. The land-use mitigation scenarios that underpin the UK's Net Zero Strategy as part of the 6th Carbon Budget (CCC, 2021) include ambitious targets to restore 25% of lowland peat to wetland, implement paludiculture (wetland-based farming) on 15%, and deploy higher water table management on a further 35%. However, such ambitious targets present major practical and economic challenges, not least because England's drained agricultural peatlands include some of the highest-value farmland in the UK; farming in the Fenland region alone is estimated to have an economic value of £3 billion $yr^{-1}$, and to employ around 80,000 people across the overall supply chain (NFU, 2019). Lowland peat is particularly important for the fresh produce sector, accounting for the majority of UK production of some salad crops (NFU, 2019, Rhymes et al., 2022). In an era of decreasing food security, rising food costs and declining real-terms incomes, mitigating GHG emissions from England's lowland peatlands without causing further price rises, loss of employment or increased reliance on imports (with the associated risk of simply displacing GHG emissions from food production to other regions) represents a major challenge. This was recognised in the UK's National Food Strategy review (Dimbleby, 2022), and in the formation and work of the Defra Lowland Agricultural Peat Task Force.

For the current project, '*Managing agricultural systems on lowland peat for decreased greenhouse gas emissions whilst maintaining agricultural productivity*' ('Defra Lowland Peat 2') we have sought to build on the existing evidence base, developed through research funded by Defra, BEIS, UKRI and others, to identify and evaluate a range of options for the future productive management of England's lowland agricultural peatlands, in order to reconcile the currently competing demands of climate change mitigation and food production, whilst also taking account of other issues such as biodiversity and water supply. In the first part of the project, we undertook a detailed review of the options, opportunities for and barriers to paludiculture (Mulholland et al., 2020), which is summarised in Section 3. A second shorter 'scoping study' (Page et al., 2020) evaluated the societal impacts of peatland drainage, such as the effects of peat subsidence on roads and other infrastructure, which are often omitted from economic assessments of the costs and benefits of agricultural land use on peat; this review is summarised in Section 4. The field component of the project, described in detail in Section 5, consisted of a new, plot-scale field experiment to evaluate the effect of potential mitigation measures on GHG fluxes and crop production; the operation, expansion and synthesis of data from the UK's lowland peat flux tower network, including new field-scale water table manipulation trials; and an extensive survey of $CO_2$ emissions from a broader range of locations. Finally, in Section 6 we worked with the farming sector across England to evaluate the effectiveness, practicality and economic viability of a wide range of mitigation options for agricultural peatlands; to identify key barriers and opportunities for change; to assess the economic impact and cost-effectiveness of a range of different mitigation measures; and to review the implications of these measures for a wider range of environmental impacts and ecosystem services affected by peatland management.

We recognise that the field element of the project has mainly focused on the East Anglian Fenland region, as England's largest and most intensively managed area of lowland peat. However, our field programme and farmer consultations have extended to other areas including the Somerset Levels and Moors, Humberhead Levels, Lancashire Mosses and Norfolk and Suffolk Broads, and the work as a whole is intended to inform the management of all of England's lowland peatlands, as well as those of the other UK countries and beyond. We acknowledge that the project has not been able to answer all questions – for example, more work is needed on the sustainable management of grasslands – and it has also raised new questions, which we hope to address through ongoing and future work. As the following report makes clear, there are no easy answers to the challenges of environmentally and economically sustainable lowland peat management.

# 3. Review of paludiculture

A full report on this work package was published earlier in the project (Mulholland et al., 2020) and is available for download at https://lowlandpeat.ceh.ac.uk/. It will also be published on Defra R&D pages as an annex to the project: Managing agricultural systems on lowland peat for reduced GHG emissions - SP1218. The following provides a brief summary and update of the findings.

Paludiculture involves the productive use of wet and re-wetting peatlands in a way that preserves their carbon stocks and minimises GHG emissions. Activities that fall within this definition range from traditional land-uses such as reed-cutting for thatch, through to new forms of management to create novel products such as insulation materials or new food crops. It commonly involves the cultivation of native wetland species, but can extend to non-native or wetland-tolerant crops. The prospect of raising water levels to reduce emissions in peatlands managed for production demands new ways of growing existing crops, or new crops capable of thriving with elevated water tables.

At the time that the report was produced, the number of field-scale paludiculture trials in the UK was quite limited in terms of both spatial extent and the scope of measurements undertaken. Where trials have been undertaken, findings suggest that paludiculture has the potential to reduce $CO_2$ (and overall GHG) emissions relative to conventional drainage-based agriculture or peat extraction. This mitigation potential largely takes the form of avoided present-day $CO_2$ emissions from deep-drained peat cropland, which can be as high as 25-30 t $CO_2e$ $ha^{-1}$ $yr^{-1}$ (see subsequent sections). Some studies suggest that paludiculture sites could become net $CO_2$ sinks, thereby helping to sequester GHGs from the atmosphere, although there is inevitably some trade-off between the amount of biomass that can be harvested from a site in paludiculture products versus the amount of net primary production (i.e. organic matter) that remains in the system to support new peat formation.

Emissions reductions and new $CO_2$ capture by adoption and uptake of paludiculture techniques have the potential to make an important contribution to achieving the UK's commitment to net zero GHG emissions by 2050. However, it is also important to consider methane ($CH_4$) emissions, which are typically higher from wetlands (including sites managed by paludiculture) compared to drained cropland, and may therefore partly offset the climate mitigation benefits of reducing $CO_2$ emissions. The highest $CH_4$ emissions occur when water levels are above the peat surface, and/or where nutrient levels are high, which represents a risk in the case of re-wetted farmland. Some potential crops, such as *Sphagnum* bog moss, have the capacity to oxidise $CH_4$ before it is released while others such as *Typha latifolia* (bulrush) may facilitate its transport from depth. However, with careful crop selection and optimised water management it should be possible to minimise $CH_4$ emissions, whilst also sequestering $CO_2$.

Contrary to the widespread assumption that peatlands need to be drained in order to enable high productivity, undrained wetlands are among the most productive ecosystems in the world (e.g. Rocha and Goulden, 2009). As a result, fen peatlands have the potential to be used to grow biomass crops for bioenergy (and in theory at least Bioenergy with Carbon Capture and Storage, BECCS) including native species such as *Phragmites australis* (common reed) or high-yielding wetland tolerant non-natives such as some varieties of

*Miscanthus*. Some fenland species such as *Typha* can also be used to produce building materials such as insulation boards, or even included in clothing. Coppicing of wet (carr) woodland on fen peat provides another potential source of biomass for bioenergy or building materials. On bog peat, the most developed paludiculture option is *Sphagnum* cultivation for horticulture, as a direct substitute for the use of extracted peat.

In situations where paludiculture replaces farmland used for food production, there is a risk of displacing emissions and other environmental impacts either to other areas of the UK, or overseas. Producing food via paludiculture is therefore an appealing option, but as yet there are few wetland-adapted food crops that could be grown under UK conditions. However, some paludiculture trials of food crops are ongoing, including high water-table celery cultivation and *Glyceria fluitans*, a grain crop. A range of other food and medicinal crops can be grown on re-wetted peat, but at present the markets for most are either undeveloped or limited in terms of their potential scale.

More generally, there remain significant practical, economic and societal challenges for the large-scale implementation of paludiculture, including the need to support rural economies, maintain national food security, develop markets and supply chains, manage water within complex and heavily modified landscapes, and avoid displacement of emissions associated with food production to other areas. Facilitating the wider adoption of paludiculture is likely to require the development of new financial incentive schemes for farmers, landowners and investors, new regulatory approaches and investment in supporting infrastructure. This is turn requires a stronger evidence base, both to develop viable paludiculture systems and to accurately quantify the associated benefits and trade-offs. Compared to conventional crops grown under higher water tables, paludiculture crops may offer lower but more reliable economic yields, and by protecting the soil from ongoing loss may help to maintain the productive lifetime of the soil. The high water demand of paludiculture crops presents some challenges in water-scarce regions, but well-designed areas of paludiculture within farmed landscapes could also provide effective water storage within the landscape, holding flood water during winter and releasing some of this to adjacent farmland during summer. The incorporation of paludiculture areas within farmed landscapes may therefore enhance their overall resilience to climate change.

Overall, we concluded that, although there is considerable potential, paludiculture does not yet offer an economically viable, large-scale or immediately implementable solution to the challenge of high GHG emissions from cultivated lowland peats. However, this should not preclude continued research and development into the potential of high-water table crops, or to the development and expansion of paludiculture trials with the aim of scaling these up where successful. Since the publication of Mulholland et al. (2020) there have been significant steps forward in this regard, including the 'Paludiculture Roadmap' developed as part of the Defra Lowland Agricultural Peat Task Force, a growing number of individual trials, and the commissioning of additional work via the Defra/Natural England 'Paludiculture Exploration Fund'. Several major projects have also been funded to explore the potential to use re-wetted peatlands for carbon capture and storage, in support of the government's Net Zero strategy, including the UKRI Peat Greenhouse Gas Removal (GGR) Demonstrator project and BEIS Reverse Coal project. These projects are evaluating the extent to which 'carbon farming' can be used to sequester additional carbon in peatlands, for example by growing biomass crops to produce and/or store biochar in re-wetted. Although similar to paludiculture in terms of both concept and land-management, these

GGR projects differ in that the carbon capture and GHG removal itself is the primary aim (and marketable 'product') of the management, rather than a by-product of wetland management to produce a harvestable product. However, the recently published Peatland Code 2.0 does not yet include paludiculture or peat GGR as mitigation options, due to a continued lack of field-scale demonstrations and GHG flux data. These data are required in order to generate robust emission factors and resulting carbon credits, and their collection across a representative range of projects is therefore a priority, along with the development of commercially viable wet farming systems, supply chains, markets and financial mechanisms. Until and unless paludiculture becomes a viable large-scale proposition, it remains essential to mitigate emissions from UK peatlands remaining under drainage-based arable and horticulture cultivation.

# 4. Societal impacts of lowland peat drainage

A full report on this work package was published earlier in the project (Page et al., 2020) and is available for download at https://lowlandpeat.ceh.ac.uk/water-level-management. It will also be published on Defra R&D pages as an annex to the project: Managing agricultural systems on lowland peat for reduced GHG emissions - SP1218.The following briefly summarises the key conclusions.

Peat is an organic material that contains very little solid matter and is around 90% water by volume when saturated. Drainage of previously saturated peat soils sets in motion a series of events resulting in reduction in peat volume and lowering of the land surface. Peat subsidence is a function of several processes, namely peat consolidation, compaction and shrinkage, and the oxidation of previously water-saturated organic material under aerobic conditions. The first three processes lead to an increase in peat bulk density over time and concomitant changes in peat hydrology. Oxidation, acting alone, does not increase peat bulk density, but does result in greenhouse gas emissions, thereby connecting peat subsidence to climate change. Other processes can also contribute to lowering of the peat surface, including erosion by wind and water, peat off-take during crop harvest, peat extraction, and burning. Contemporary rates of subsidence for drained lowland fen peatlands under arable agriculture in the UK are typically in the range 1-2 cm $yr^{-1}$ (Evans et al., 2019). At Holme Fen in Cambridgeshire, 128 years of drainage has resulted in total subsidence of around 4 m (Hutchinson 1980). Wind erosion makes a smaller contribution to peat loss and subsidence. In the Fens, wind erosion typically occurs during the spring months when the soil has been ploughed but is without a crop cover, and can be particularly high when fields are prepared for late season crops, such as sweetcorn (*Zea mays*), which is widely grown on peat for biogas production. Estimated losses via wind erosion translate into a peat surface lowering of 0.03 to 0.27 mm $yr^{-1}$ (Cumming, 2018; Newman, 2022).

Land subsidence resulting from the drainage of lowland peatlands can result in an array of negative impacts for infrastructure. While some of these have been previously recognised, most emphasis to date has been on identifying and addressing the *symptoms* of subsidence, rather than addressing the *causes* or gauging the associated economic or social *costs*.

The most direct consequence is a change in hydrology, since subsidence brings the peat surface within the reach of local river flood levels or, in coastal areas, of high tide levels. Large areas of the Fens are below sea level (40% of Lincolnshire; 50% of Cambridgeshire) but drainage has provided some of the most fertile agricultural land in the UK, producing a third of England's fresh vegetables (NFU, 2019). Maintaining agricultural production, whilst also ensuring protection from flood risk, has necessitated significant investment in embankments and coastal flood defences, drainage pumps and sluices, which are managed by a combination of Internal Drainage Boards (IDBs), the Environment Agency and local authorities. During 2015-16, IDBs in England invested £61 million in water level management work, with additional investment from the Environment Agency to maintain fluvial and coastal flood defences (ADA, 2017). An unknown portion of costs associated with maintenance of watercourses and flood defences are attributable to peat subsidence, including repairs to embankments that have slumped or deformed and deepening/clearance of drains.

Peatland drainage and associated subsidence also have consequences for maintenance of other categories of infrastructure. Peat shrinkage affects thousands of kilometres of the UK's road network, as well as sections of the rail network. Roads crossing peat soils in the Fens suffer regular deformation, cracking and pot-holing, resulting in high repair costs for local authorities. Even where Fenland roads are located on silty ridges, subsidence of the peat either side of the ridge has left roads well above the adjacent landscape, necessitating investment in crash barriers to improve road safety. Several railway lines cross lowland peatlands. Reported issues include track deformation, resulting in reduced engine power, increased journey times and regular repairs of the track bed, and ground vibration boom from high speed trains, which requires investment in mitigation measures to reduce dynamic amplification.

Where houses and other buildings have been constructed on peats, subsidence can cause cracks, tilting and differential settlement. Compared to the Netherlands, there has only been limited urban and rural development on lowland peat soils in the UK, thus subsidence damage to properties appears to be a relatively minor problem. In the Fens, most settlements are located on mineral islands or ridges, rather than on peat, and have relatively stable foundations. Communication and energy supply networks are also at risk of damage from peat subsidence, as evidenced by tilting of telegraph poles and the differential movement of energy supply pipelines.

In addition to direct impacts on infrastructure, current water and land management practices on lowland peatlands incur a range of other societal benefits and costs. In England, around 2400 $km^2$ of drained lowland peatland are farmed for food production which brings with it benefits for the rural economy, employment and food security. It is estimated that Fenland agriculture and food-related industries employ 80,000 people and generate around £3 billion a year for the regional economy (NFU, 2019).

Lowland peatlands contain a wealth of archaeological interest, but drainage and peat wasting have exposed buried artefacts to aerobic decay, degradation and loss. Examples of peatland archaeology include the world's oldest surviving trackway in the Somerset Levels as well as human remains (so-called bog bodies). It is estimated that as many as 10,000 archaeological monuments (74% of the total resource) have been destroyed completely in the last 50 years as a result of peatland drainage and peat loss (Van de Noort et al. 2002). Mitigation measures to prevent further loss will require landscape-scale maintenance of high water levels.

Peatland drainage and land use change have also resulted in the demise, or in some cases the transformation, of peatland cultural values. Drainage of the Fens led to the loss of a unique cultural heritage associated with the exploitation of the former wetland's rich natural resources. Nevertheless, for today's communities, the unique drainage history of the Fens, along with their important farming and food production history, provide a strong sense of tradition and place.

Peat drainage and loss also result in loss or reduction of other valued ecosystem services – carbon storage and biodiversity support. Currently reported greenhouse gas emissions from English peatlands are estimated to be around 11 Mt $CO_2e$ $yr^{-1}$, with lowland peatlands drained for agriculture contributing 80% of this emission. Halving the drainage depth across all peatland under intensive agricultural use in the UK, most of which is in England, could reduce emissions by around 70% (Evans et al. 2021). A large proportion of remaining, undrained lowland peatlands are protected as Sites of Special Scientific Interest and both lowland fens and bogs are included as priority habitats in the UK Biodiversity Action Plan. The main threats to their biodiversity interests are water management, including drainage and excessive water abstraction from underlying aquifers, and pollution from agricultural run-off. In the Fens, peat subsidence has left areas set aside for nature conservation isolated as ‘wet’ islands perched several metres above adjacent drained fields. This incurs management costs and challenges for maintaining an appropriate wetland hydrology.

Mitigating the risks posed by current water management regimes in lowland peatlands will require consideration of appropriate actions to reduce hazards, reduce exposure, and reduce vulnerability.

Measures to reduce *hazards* focus on raising the peatland water table to counteract subsidence. This would deliver benefits in terms of reduced greenhouse gas emissions, reduced maintenance costs for transport routes and other infrastructure, protection of archaeological heritage, and improved hydrological security for wetlands managed for nature conservation, but would have major impacts for current drainage-based agricultural production, and potentially place additional pressures on regional water supplies.

Measures to reduce *exposure* could include diverting traffic away from roads without strong foundations, strengthening transport routes that cross peatlands, limiting further infrastructure development on peat soils, and wider uptake and implementation of on-farm soil conservation measures to reduce erosion losses.

Measures to reduce *vulnerability* include designing future infrastructure to take account of both the low load bearing capacity and subsidence of peat substrates and the increased risks of fluvial and coastal flooding under future climate change scenarios. The magnitude of risks will be determined by the characteristics of a particular location (e.g. elevation, proximity to river/coast); vulnerability of assets and people (e.g. presence of high value agricultural land, infrastructure, future impacts of climate change); and the mitigation and adaptation measures already in place, and their effectiveness.

Implementing appropriate mitigation measures will reduce risks but it will not be possible to offset or eliminate all of them. Measures need to be judged according to their specific costs and benefits (social, economic, environmental) over appropriate timescales. For example, the rate of peat subsidence could be reduced or even stopped by raising water levels. This would provide benefits in terms of reduced costs for water management, reduced greenhouse gas emissions and so on, but would challenge various agriculture-related functions and interests. Taking all lowland peatlands out of agricultural production would significantly impact on UK food production, as well as having implications for livelihoods and regional economies.

Climate change also needs to be considered in any assessment of the costs and benefits associated with peatland drainage. Climate change projections indicate that the UK is likely to experience hotter, drier summers (such as that of 2022) and wetter, warmer winters (UK Climate Change Risk Assessment 2017). These conditions will promote and possibly enhance

current rates of subsidence; they could also increase the risk of peat loss by wind erosion and, during extended droughts, increase the risk of damage to infrastructure. In addition, lowland peatlands located at or below sea level, such as the Fens, the Somerset Levels and the Norfolk Broads, could be at increasing risk of coastal flooding and saline intrusion and incursion, both as a result of sea level rise and the increased risk and height of storm surges. This level of increased risk could incur additional costs for the IDBs, the Environment Agency and local authorities with responsibility for land drainage and flood risk management.
The scoping study of Page et al. (2020) provided a broad assessment of the principal environmental, economic and social impacts arising from the drainage of lowland peatlands in England and Wales. There remain some key uncertainties and knowledge gaps which could lead to underestimation of the total scale of the impacts. In view of this, it would currently be difficult to model the returns (costs and benefits) delivered from implementing most of the proposed mitigation measures. Nevertheless, we can confidently conclude that the costs associated with drainage are largely 'hidden' and/or are not directly connected to drained peatlands and their management.

Key uncertainties relate to costs associated with infrastructure, both in terms of maintenance and higher initial costs associated with construction on soft and subsiding substrates, and on society, particularly in terms of the costs of providing and maintaining land drainage and flood defences. While some infrastructure impacts arising from peatland drainage have been recognised in previous studies, most of the emphasis has been on identifying and addressing the symptoms of subsidence and little consideration has been given to addressing the causes.

A more detailed assessment would allow: i) an improved understanding of the effect of alternative water and land management measures on subsidence and greenhouse gas emissions; ii) an insight into the key financial values, enabling an accurate cost-benefit analysis; and iii) an understanding of what will happen, for example in terms of damage to infrastructure or loss of high value agricultural soils, if nothing is done, thereby providing the basis for a business as usual scenario against which to compare various policy options.

**The full report on which this summary is based is available here:**
***https://lowlandpeat.ceh.ac.uk/water-level-management.*** It will also be published on Defra R&D pages as an annex to the project: Managing agricultural systems on lowland peat for reduced GHG emissions - SP1218.

# 5. Assessment of the greenhouse gas impact of potential mitigation measures

## 5.1 Introduction

This section comprises the main field-based elements of the project, which aimed to expand on existing knowledge regarding the relationships between GHG emissions and agricultural peatland management, to test prospective mitigation measures aimed at reducing these emissions, and to evaluate the extent of any trade-offs with agricultural production and other ecosystem services. The work initially comprised: 1) a plot-scale experimental mitigation trial; 2) continuation and expansion of the lowland peat flux tower network; and 3) extensive measurements across a wider range of Fenland agricultural soils to examine the extent to which results from the flux towers and plot experiment are representative of the wider farmed lowland peat landscape. A number of factors resulted in amendments to the original work plan, notably restrictions on access to field sites during the Covid pandemic, which affected implementation of the field experiment and measurement programmes in particular, although most of the flux towers continued to collect data during this time, and a number of new sites were brought online despite the challenges of Covid. Subsequently, we experienced several practical challenges with the plot-scale trial, the most significant of which was that it proved impossible (despite several modifications) to hold water levels continuously high within the experimental plots, which necessitated a change in the original experimental design as discussed below. More positively, additional funding for capital equipment, and closely related work for the BEIS Wasted Peat and UKRI Peat GGR projects, significantly expanded the flux tower measurement programme, and additional flux tower data were provided by the Fenland SOIL group. With farmer support we were also able to establish two full field-scale water level manipulation trials with paired flux towers, an option which had not been available when the project was first planned and which realised a long-term ambition of our work. The results of each trial are described below.

## 5.2 Plot-scale irrigation trial

### Introduction

There is now a substantial body of evidence to show that average $CO_2$ and $CH_4$ fluxes, and to an extent also $N_2O$ fluxes, are correlated with water table depth (e.g. Couwenberg et al., 2011; Tiemeyer *et al.,* 2016; Evans *et al.,* 2021a). However, these relationships are largely derived from comparisons between sites with differing long-term water table depths (WTDs). Most assessments of emissions mitigation assume (explicitly or implicitly) that a change in mean WTD will lead to a commensurate change in $CO_2$ and other GHG emissions, in line with these published studies – i.e. that temporal changes in emissions at one location from WTD = A to WTD = B will correspond to the observed difference in emissions between two different sites, one with WTD = A and one with WTD = B. Although this assumption is supported by most mechanistic understanding and likely to hold true in the long term, there is less confidence about the emissions that may occur during a transitional phase, for example from a drained agricultural peatland converted back to a wetland, during which emissions of $CO_2$, $CH_4$ and/or $N_2O$ may be elevated. Legacy effects of land-use could be prolonged in some cases, as a result of nutrient enrichment, physical changes in the soil, or loss of original vegetation, and may in part help to explain why $CO_2$ and $CH_4$ emission factors for rewetted fen and bog tend to be higher than those for near-natural fen and bog (Evans et al., 2021a).

To date, there have relatively few studies that have measured changes in GHG emissions under realistic field conditions following a change in management. In particular, there have been few studies in which water levels have been raised within agricultural peatlands. In addition, engagement with farmers during the project, and as part of the work of the Defra Lowland Agricultural Peat Task Force, highlighted the profound challenges of raising water levels in some areas of peatland, and particularly in those areas where peat wastage has reduced the remaining peat depth and modified the surface topography to the extent that raising and holding water levels uniformly high across an entire field may not be possible. This problem has been made more acute by the expansion of fields and removal of some ditches (partly to reduce the proportion of land lost to protective buffer strips around watercourses, i.e. ditches) which makes precise management of water levels more difficult. For this reason, consideration has been given to the option of surface-irrigating some areas to maintain soil moisture levels where subsurface irrigation via the ditch network and subsurface drains is not possible.

The experiment established for this task was intended to provide a flexible, highly controllable and replicated, plot-scale facility for testing the impacts of different hydrological management regimes on GHG fluxes, crop yield and condition, and other metrics of crop and soil health. A key aim of the experiment was to examine the extent of trade-offs between wetter agriculture and crop yields, in order to identify an appropriate balance between these often-competing objectives.

## Methods

### *Field site and methodology*

The Rosedene Farm site was chosen for this study. The site has thick peat soils, supporting high-value production of a range of horticultural crops. A flux tower has been present on the farm since 2012 providing long-term context and a baseline understanding of the $CO_2$ balance. A field was chosen with close proximity to a building with mains electrical power for the instrumentation, which would go through rotations of crops suitable for an automated chamber study (i.e. not tall crops, like maize). The instrumentation was installed along the field boundary next to a ditch so that there would be ample access to water for filling/irrigating the plots.

The original experimental design centred around the installation of piling to hydrologically isolate individual peat blocks which would be gravity fed with water from elevated water tanks at each end of a transect. Drainage would then be controlled at desired water level depths, providing control of water levels within individual plots. However, following piling installation, it was not possible to control water level in plots due to apparent leakage though or (more likely) below the piling. Several attempts were made to seal the piling, after which further investigation showed the presence of a thin sand layer at around 1 to 1.2 m depth, which appeared to be stopping the piling from sealing into the underlying clay. To address this, the peat around each block of three experimental plots was excavated using a digger, and trenches were dug down to the clay subsoil. The trenches were then backfilled with clay in an attempt to seal each set of plots. This, unfortunately, also failed to hydrologically isolate the experimental plots from the rest of the field, most likely due to porous subsoil below the plots.

In the autumn of 2020 (following the end of access restrictions due to Covid), GHG fluxes were measured but without hydrological manipulation of the plots, which were managed for winter wheat in line with the rest of the field. In 2022, following the unsuccessful attempt to seal the plots, and in recognition of the growing interest in surface irrigation as a potential alternative mitigation measure, the experiment was adapted to allow controlled surface (overhead) irrigation. The plots were then planted with lettuce, again in line with the management of the rest of the field, and GHG fluxes were measured with varying rates of irrigation for the duration of the cropping cycle.

### *Crop management and experimental design*

Measurements were made over 13 sample plots, along a 20 m transect following the field margin. In 2021 all 13 plots received the same treatment which involved plot preparation (tillage), planting and harvest of the wheat crop, but no differential water management. No additional fertilisers or pesticides were applied to the plots (Table 5.1). The wheat crop was managed as closely as possible to that in the rest of the field, and was harvested in August 2021.

| Plot | 1 | 2 | 3 | 4 | 5 | 6 | 7 | 8 | 9 | 10 | 11 | 12 | 13 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Treatment | BAU | High | High | High | High | Medium | Medium | Medium | Medium | BAU | BAU | BAU | BAU |
| No. Lettuce | 7 | 14 | 16 | 16 | 8 | 15 | 16 | 15 | 8 | 16 | 16 | 15 | 8 |

***Figure 5.1***. *Plot layout for the 2022 irrigation trial (above) and schematic and photograph below showing typical lettuce planting within each treatment plot.*

In 2022, a first surface irrigation trial was abandoned due to failure of the crop (Chinese cabbage) within the experimental plots. A second surface irrigation trial was initiated in July, this time with a lettuce crop. Plots were split into three different treatment groups, with clusters of four plots along the transect each receiving the same surface irrigation treatment (Figure 5.1). Although not ideal from an experimental design perspective, this approach was more practicable in terms of controls on water application per plot, and also avoided the risk of lateral seepage between adjacent plots receiving different levels of irrigation. Irrigation was either based on standard ('business as usual, BAU) irrigation in line with the rest of the field, or increased to medium (MI) or high (HI) irrigation levels. Plots 1 and 10-13 were managed according to BAU field management, receiving irrigation of 25 mm roughly once a week over the period of crop growth. Plots 2-5 were heavily irrigated (HI) with roughly 55 mm applied every day over 20 minutes, from July 7$^{th}$ to September 16$^{th}$. Plots 6-9 were moderately irrigated (MI) with roughly 0.7 mm being applied daily over 10 minutes from 7$^{th}$ – 13$^{th}$ July and thereafter 2.2 mm, again over 10 minutes, which reduced to 1.7 mm due to pump blockages during the remainder of the experiment.

Measurement plots were 1.5 by 1.5 m (2.25 m$^2$). Plots 1, 5, 9 and 13 were smaller, at 0.75 by 1.5 m (1.13 m$^2$), due to previous groundworks during attempts to seal the piling. Although GHG flux measurements were made for these smaller plots, they were excluded from the final analysis. Under the guidance of the farm manager, the management strategy applied to the crop on the plots followed that of the crop in the rest of the field, this is summarised in Table 5.1 and includes regular treatments of herbicides (applied by knapsack) and more frequent heavy irrigation due to drier than usual conditions throughout summer 2022. Lettuce seeds were germinated in peat plugs in greenhouses, and then transplanted (in the plugs) to the field. The early seeding, cotyledon and seedling stages therefore happened before outdoor planting. Plugs were planted with a density of 14 plants per square meter, again reflecting the wider field management. Fertiliser and irrigation were applied across all plots as per Table 5.1. Lettuce was harvested following the grower's advice and outer leafy material typically left on field was left to decompose on the surface of the plots.

### *Instrumentation*

GHG fluxes were measured using a Skyline 2D system (Earthbound Scientific, York, UK) to undertake automated, high-frequency measurements of GHG fluxes from multiple plots along the transect (Figure 5.2). The Skyline comprises a cableway, along which moves a dolly unit supporting a gas sampling chamber along the transect. The dolly unit stops at pre-determined points to make measurements with a chamber with an inner diameter of 0.38 m and height of 0.7 m. The chamber was lowered onto permanent soil collars with a matching diameter, using guide wires to ensure that the chamber aligned and sealed with the collars. The gas density within the chamber was sampled through recirculating gas lines connected to a greenhouse gas analyser. The chamber was set to measure on each plot for 5 minutes, before being lifted and transported to the next plot. Gas lines were purged for 2 mins before repeating the flux measurement process. Plots were measured in sequence from 1-13, with measurements made two-hourly on each plot when conditions allowed (during high winds, the system shut down to avoid damage to the equipment or crop). The system ran throughout the day and night, thereby collecting very high-resolution data throughout the experimental periods.

***Table 5.1.*** *Agricultural management information for the Skyline experimental plots during periods of measurement.*

| **Date** | **Management** |
|---|---|
| **Wheat crop – no hydrological manipulation** | |
| 15th October 2020 | Plots weeded |
| 21st October 2020 | Plots ploughed and disked by hand |
| 30th October 2020 | Wheat sewn by hand |
| 19th August 2021 | Wheat crop harvested |
| **Lettuce crop – surface irrigation** | |
| 7th July 2022 | Plots weeded |
| 7th July 2022 | Plots weeded by hand and irrigation started |
| 12th July 2022 | Herbicide: Pendemethlin 0.2 ml $m^{-2}$<br>Fertiliser: N.P.K: 6-6-12 36 ml $m^{-2}$ |
| 13th July 2022 | Plots ploughed to a depth of 0.35 m and then surface disked in preparation for planting<br>Romaine lettuce planted (density of 12 $m^{-2}$, 3 per collar)<br>Irrigated with 25 l $m^{-2}$ |
| 15th July 2022 | Irrigated with 25 l $m^{-2}$ |
| 20th July 2022 | Herbicide: Kerb flo 0.1875 ml $m^{-2}$, stomp 0.05 ml $m^{-2}$<br>Pesticide: Movento 0.05 ml $m^{-2}$, Hallmark 0.0075 ml $m^{-2}$, Switch 0.08 ml $m^{-2}$<br>Fertiliser: $MnSO_4$ 0.2 g $m^{-2}$, $MgSO_4$ 0.2 g $m^{-2}$, Headland Complex (Nutrient mix @0.3 g $m^2$) |
| 21st July 2022 | Irrigated with 25 l $m^{-2}$ |
| 27th July 2022 | Herbicide: Kerb flo 0.1875 ml $m^{-2}$, stomp 0.05 ml $m^{-2}$<br>Pesticide: Movento 0.05 ml $m^{-2}$, Decis Protech 0.042 ml $m^{-2}$, Revus 0.06 ml $m^{-2}$<br>Fertiliser: MnSO4 0.2 g $m^{-2}$, Mg $SO_4$ 0.2 g $m^{-2}$, Headland Complex (Nutrient mix @0.3 g $m^{-2}$) |
| 28th July 2022 | Irrigated with 25 l $m^{-2}$ and manually weeded plots |
| 4th August 2022 | Manually weeded plots |
| 10th August 2022 | Irrigated with 25 l $m^{-2}$ |
| 16th August 2022 | Irrigated with 25 l $m^{-2}$ and manually weeded plots |
| 24th August 2022 | Lettuce crop harvested |

Within the chamber, meteorological variables were recorded using a Quantum sensor (Skye Instruments Ltd., UK) which measured photosynthetic photon flux density (PPFD, µmol photons $m^{-2}$ $s^{-1}$), and a thermistor (LI-COR Inc., USA) for air temperature. Tubing ran from the chamber to a Los Gatos benchtop Fast Greenhouse Gas Analyser (Los Gatos Research Inc. USA), which measures $CO_2$ and $CH_4$, during the 2021 experimental period. In 2022 this was switched to a Picarro G2508 (Picarro, California, USA) measuring $CO_2$ and $N_2O$. Gas densities were sampled and logged as dry mole fractions (mol $mol^{-1}$) at a resolution of 1 Hz. Both analysers were housed in a trailer for which mains electrical power was supplied.

Volumetric soil water content (VWC) and soil temperature were recorded in each plot using a Time Domain Transmissometer (TDT) sensor (Acclima, Idaho, USA) with internal temperature sensor. Soil water level was recorded using a CS451 pressure transducer (Campbell Scientific Inc, USA). PPFD was also recorded outside of the chamber using a second SKYE Quantum sensor, and rainfall was recorded using an SBS500 tipping bucket rain gauge (Environmental Measurements Limited, UK). All meteorological and soil physical variables were recorded on a CR1000 (Campbell Scientific Inc, USA) data logger, averaged once a minute and telemetered back to a secure UKCEH data server.

***Figure 5.2.*** *The Skyline plot-scale irrigation experiment at Rosedene, with recently planted lettuce crop (above) and the plot treatment layout (below)*

### *GHG flux data processing*

Raw gas concentration measurements of $CO_2$ and $CH_4$ (Los Gatos analyser) and $CO_2$ and $N_2O$ (Picarro analyser) were split by timestamp into the sampling periods of each chamber closure using the R Statistical Language (R Core Team, 2023). Flux densities (fluxes) were then calculated using the 'flux' package (Jurasinski, et al. 2022, v0.3-0.1) for R. A number of quality assurance tests were applied to the data by individual plot to remove outliers and implausible flux measurements. Negative $CO_2$ fluxes measured during night-time (PPFD < 20 µmol photons $m^{-2}$ $s^{-1}$) were considered implausible for these crops and were excluded, and a restricted plausible range of measurement values was applied for $CO_2$ of -4500-8000 mg $CO_2$ $m^{-2}$ $hr^{-1}$. An additional quality control test adapted from eddy covariance based on functional responses to light and temperature was applied to the $CO_2$ fluxes as described by Elbers *et al.* (2011). In this approach, fluxes were split into 20 equal bins of temperature (at night) and PPFD (during the day). Bin averages and standard deviations of $CO_2$ fluxes were calculated. $CO_2$ fluxes were then flagged and excluded from further analysis when they fell outside of ± 2 standard deviations of the mean within each bin. Gap filling of the dataset was performed using the random forest approach, using the 'randomForest' package for R (Liaw & Wiener, 2002) with soil variables (VWC, temperature), PAR, chamber temperature, growth stage and irrigation used as predictors.

A single average hourly dataset was derived for the 2021 wheat crop. This dataset was treated as a bulk sample covering all plots, because no plot-scale treatments were applied. In 2022, data collected during the lettuce crop were averaged within treatments. As noted earlier, the smaller plots 1, 5, 9 and 13 were excluded from the analysis during this period.

### *Crop assessment*

Crop performance was evaluated for the same three plots from each treatment as the GHG fluxes. Lettuces were assessed for marketable yield (wet and dry weights), disease incidence and severity on the 27th August 2022. This was aligned with the harvest window for the commercial crops grown in the same area by the same grower.

**Marketable yield** was measured for all plants in each plot. At harvest, plants were cut at the base and outer leaves trimmed as would be done during commercial harvest. For those plants that were growing outside of the test collar, the trimmed marketable heads were counted and weighed. The trimmed leaves were combined by plot and weighed. All samples were taken to the laboratory for dry weights to be calculated.

The three plants per plot that were growing within the collar were harvested and trimmed in the same way, but the trimmed basal leaves were left on the ground as would be done during a commercial harvest. The marketable heads were weighed separately from those that were growing outside of the collar, due to the possibility that crowding within the collar and damage from the Skyline chamber had reduced yields compared to normal field conditions.

**Disease incidence** was evaluated by visually inspecting each marketable head for the presence of disease and if present scored as follows:

- Downy mildew: Percent disease severity per plant looking at the underside after it is cut using the guide from EPPO standard PP 1/65 (4) Downy mildew of vegetables.
- *Botrytis cinerea*: Each individual plant assessed using 0-5 scale:

    0 = no attack;
    1 = slight attack, infection of basal petioles only;
    2 = moderate attack, stem lesion not girdling stem;

3 = heavy/severe attack, stem lesion girdling stem, or upper leaves infected, lettuce unmarketable (including plants completely destroyed by Botrytis during the trial).
4 = total plant collapse/dead

- *Sclerotinia*: Each individual plant assessed using 0-4 scale:

0 = no attack;
1 = slight attack, plant wilted, mycelium of Sclerotinia spp. present on lower leaves
2 = moderate attack, infection of upper leaves
3 = heavy/severe attack,
4 = total plant collapse/dead

Diseases were identified according to their symptoms. Samples were not taken for culturing to confirm the cause of the visual symptoms.

**Root core analysis** was undertaken by manually collecting soil cores using a 2-6 cm diameter borer to a depth of 60 cm. Four cores were taken from 2 plots per treatment from the area immediately after lettuce harvest. High irrigation (plots 2 and 3), medium irrigation (plots 7 and 8), BAU (plots 11 and 12). Cores were separated into 20 cm horizons and amalgamated for each plot by depth.

The two deepest horizon samples (20-40cm and 40-60cm) for each quadrant and field were thawed and washed using a Delta-T root washing system with 550 micron filters to separate soil and organic material from roots. Each horizon and quadrant was washed separately. Crop debris and non-root material was removed from samples. Clean roots were placed into containers with water for scanning using WinRHIZO root analysis package software (Regent Instruments Ltd. Quebec City, Canada) and a flatbed scanner. Root measurements (total length, mean diameter and surface area) were calculated.

After scanning roots were placed into tins and weights recorded. Roots were dried in an oven at 80°C for 48 hours or until no further weight loss. Dry weights were then recorded.

## Results

### *2021 Wheat (no irrigation)*

Daily gap filled net ecosystem exchange of $CO_2$ (NEE) data (Figure 5.3) show that the sample transect was a source for most of 2021, when a wheat crop was present (until August) and no irrigation treatments were applied. All plots within the transect were a consistent daily source throughout January to February, despite a young crop being present on the field, which appears to have led to some net uptake in the hourly data (upper plot). Crop growth peaked in April-May, with a maximum measured daily net $CO_2$ sink of -5 g $CO_2$-C m$^{-2}$ d$^{-1}$, before becoming a steady source of around +5 g $CO_2$-C m$^{-2}$ d$^{-1}$ as the crop ripened, which continued after harvest before declining to lower but still positive values in autumn.

When the wheat crop had grown over 0.6 m, we started to experience interference from the chamber not landing cleanly on the collars, flattening the crop in the process. This caused damage to the crop within the collar, but did not affect the crop within plots but outside of collars. We therefore harvested all wheat from within the collars (at the same time as the rest of the field), and also harvested representative areas within the plots that were not affected by chamber closure. The average dry harvested weight (including straw) of the samples from outside the collars averaged 12.5 t ha$^{-1}$ which compares well to similar winter wheat crops measured at other sites. The dry harvested weight (including straw) of the collar samples, however, was just 27% of the sample from the rest of the plot, confirming that repeated chamber closures had significantly reduced the growth of plants inside the collar.

Annual gap-filled NEE measured for the wheat crops with the Skyline 2D was 9.3 t $CO_2$-C ha$^{-1}$ yr$^{-1}$. This figure does not include carbon exported in harvested biomass, which was estimated to be just 1.8 t C ha$^{-1}$. This gives an annual Net Ecosystem Production (NEP) of 11.1 t $CO_2$-C ha$^{-1}$ yr$^{-1}$, equivalent to a net gaseous $CO_2$ emission of 40.3 t $CO_2$ ha$^{-1}$ yr$^{-1}$.

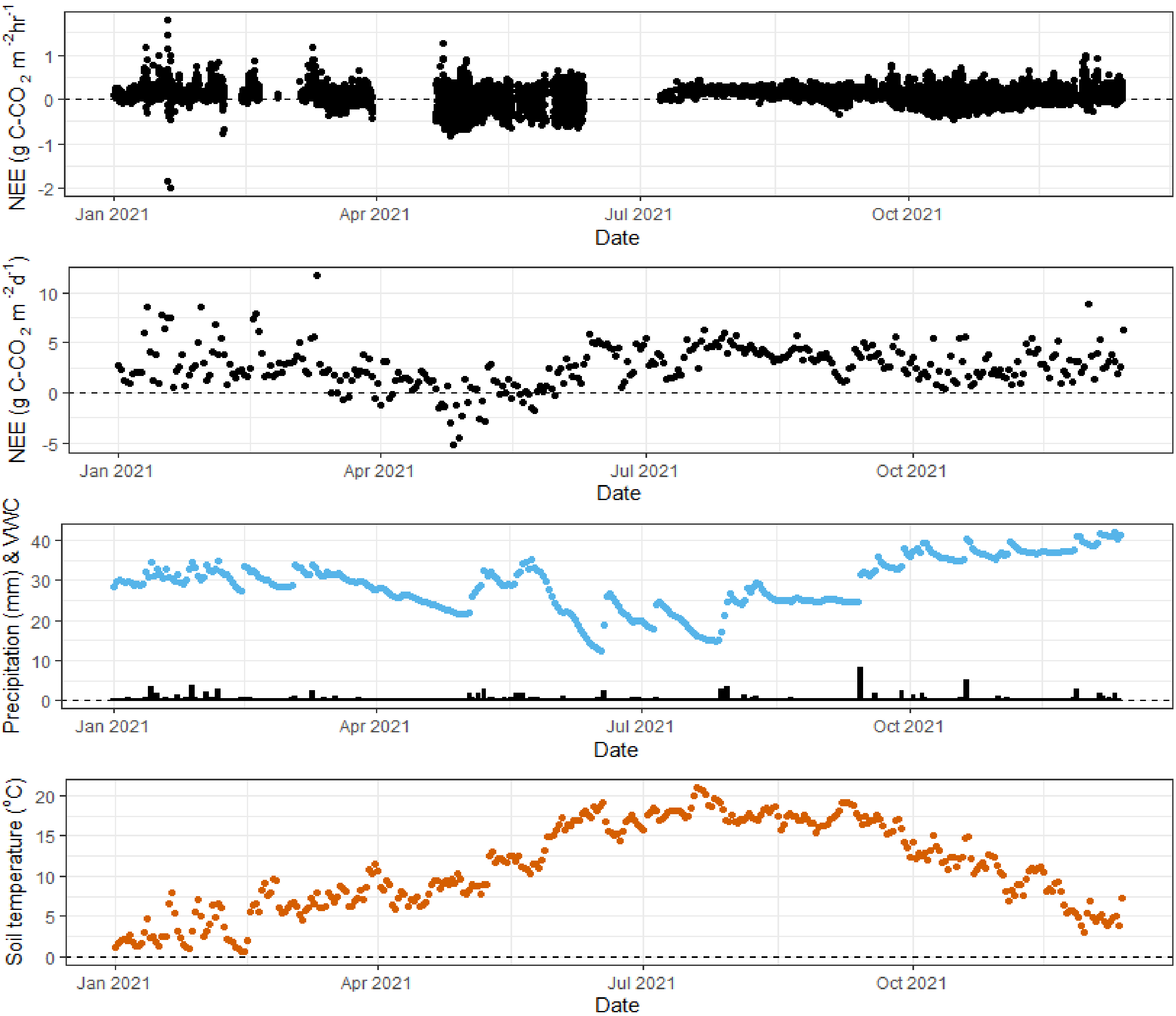



***Figure 5.3.*** *Data from the 2021 Skyline experiment with a wheat crop present, and no irrigation treatment applied. Plots show a) Mean hourly quality controlled NEE of C-$CO_2$, b) mean daily gap filled NEE of C-$CO_2$, c) mean daily soil moisture (VWC) in blue with precipitation as black bars, and d) soil temperature*

## 2022 Lettuce (surface irrigation experiment)

### *Hydrological and meteorological measurements*

The lettuce irrigation experiment began in July 2022, during severe drought and heatwave conditions, and (as shown in the upper panel of Figure 5.4) minimal rainfall during the crop period. The only significant rain event during the lettuce trial occurred just after harvest, adding around 2 mm of precipitation. The BAU irrigation added the equivalent of around 0.2 m of cumulative rain over the crop growth period. The two irrigation treatments added considerably more water, up to the equivalent of 3 m of rainfall in the HI treatment. This is clearly not a realistic level of irrigation in practice, but was intended to test the effects of raising soil moisture content at a time when the field around the plots was exceptionally dry, and the ditch level and water tables very low.

For all plots, the ploughing, planting and manual BAU irrigation applied on the 13th July led to an increase in average VWC, from < 10% to around 20-30%. Thereafter the largest changes in VWC were seen on days of manual BAU irrigation (approximately once a week) which led to a series of 'sawtooth' peaks. For the experimental irrigation treatments, daily automated irrigation resulted in a diurnal variation in soil moisture content, with VWC increasing by ~2-4% for the MI treatment, and ~5-10% for the HI treatment. Much of this increase in VWC rapidly dissipated, suggesting that most of the excess water was gravity-draining to the water table, returning to field capacity later in the day. A small diurnal variation in VWC in the BAU treatment of <1% (in the absence of irrigation) could indicate water movement between plots, and/or diurnal variation in evaporation rates.

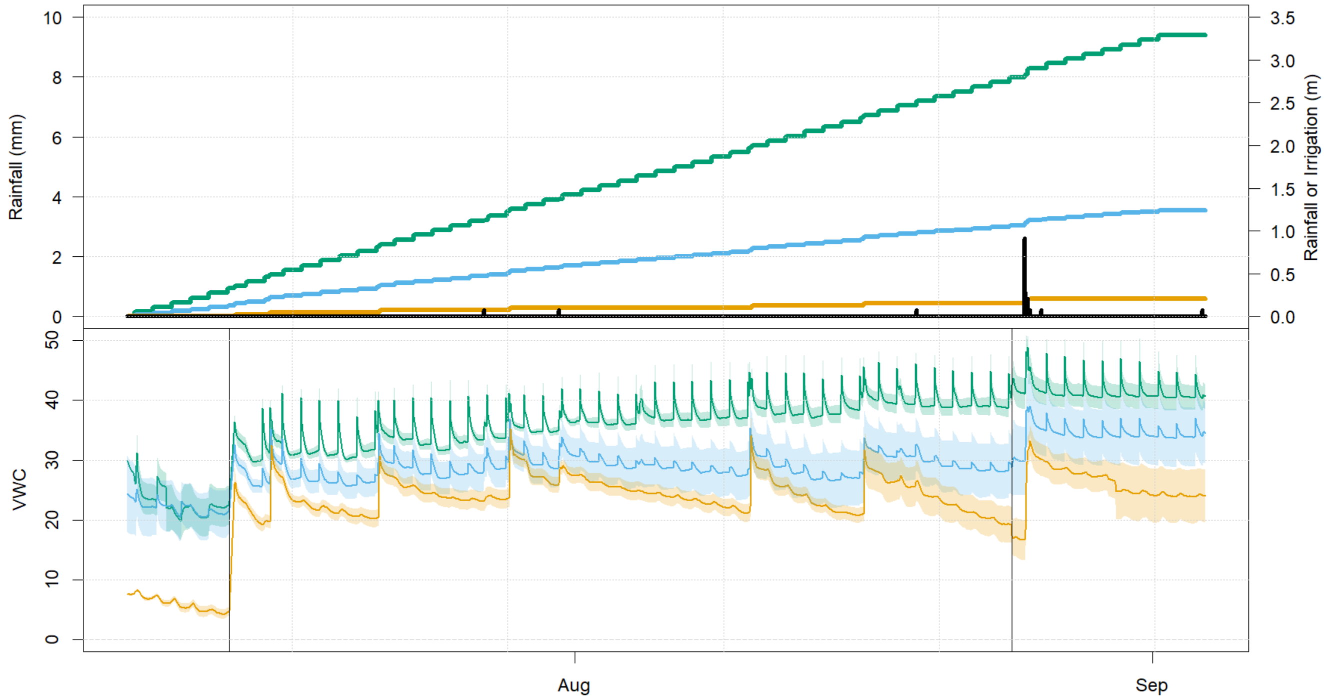



***Figure 5.4**. Top: cumulative water management including irrigation treatments and rainfall. Bottom: Time series of hourly VWC for the observation period averaged within treatment, . Shaded areas show the standard error of the mean. HI treatment is the green, MI treatment is the blue and BAU is orange.*



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Following manual (BAU) irrigation events, VWC in both the BAU and MI treatments trended steadily downwards between manual irrigation events, likely due to a combination of evapotranspiration and seepage. The downward trend was minimal in the MI plots which received daily automated irrigation. Automated irrigation of the HI plots led to a stabilised baseline of soil moisture, which likely represents a return to field capacity for this soil type after a period of infiltration and drainage though the soil immediately after irrigation. Despite the evidence of rapid gravity-drainage of much of the additional water applied to the irrigation plots, clear and sustained differences in VWC between treatments emerged during the experiment, with VWC fluctuating within the range 20-25% in the BAU plots and 25-30% in the MI plots (Figure 5.5). In the HI plots, VWC rose steadily throughout the trial, from 30% at planting to 40% at harvest.

Irrigation also resulted in clear differences in soil temperature between treatments. The influence of variable soil water levels was particularly strong during exceptionally hot conditions at the start of the experiment, when the variance between BAU and irrigated plots was most pronounced. The highest soil temperature recorded was prior to any irrigation on the BAU plots, which was also when these plots were at their driest (Figure 5.4). The irrigated treatments had similar daily soil temperatures between planting and harvest, but the BAU plots were 1-2 ˚C warmer except following manual irrigation. As the lettuce canopy closed and shaded the soil, and as air temperatures decreased through August, differences in soil temperatures between the two irrigation treatments were minimal.

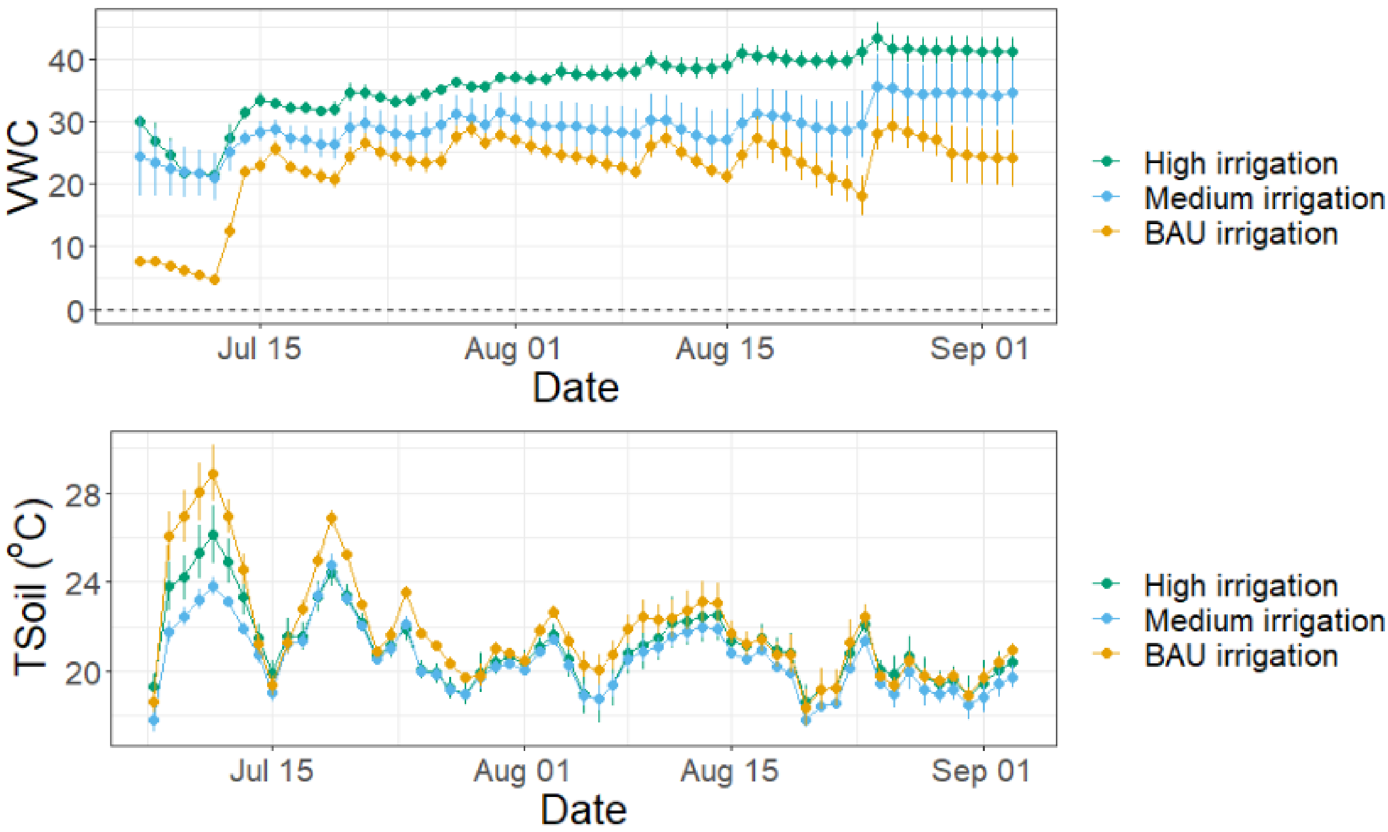



***Figure 5.5**. Daily mean volumetric moisture content (top) and soil temperature (bottom) by treatment with standard error bars.*

The groundwater level was kept at roughly 0.65 m below the field surface between planting and harvest, which is typical for a lettuce crop on thick peat in this region (Figure 5.6). It dropped by 100 mm in late July, and then was immediately dropped post-harvest by over 200 mm. Despite this drop in water level the VWC remained high in all treatments, possibly due to the rain event that followed harvesting.

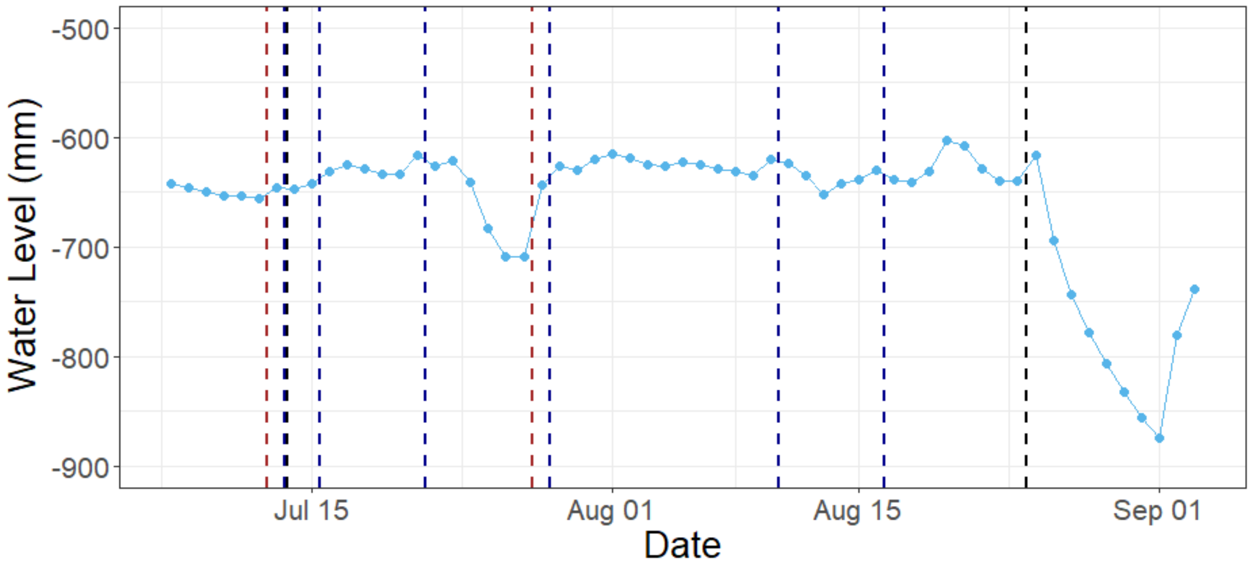



***Figure 5.6****. Field water level beneath measured within HI treatment*

*$CO_2$ fluxes*

Average daily $CO_2$ fluxes per treatment and cumulative $CO_2$ fluxes per individual plot are shown in Figure 5.6. Daily $CO_2$ emission (positive NEE) peaked at 11.3 g $CO_2$-C m$^{-2}$ d$^{-1}$ in the HI treatment the day after the plots had been ploughed and plugs planted. NEE then declined steadily in all treatments as fertiliser was applied and the lettuce crop developed leaf area, becoming briefly negative in the plots receiving additional irrigation during the first week of August. This period of net uptake coincided with soil temperatures dipping below 20°C for the first time. This brief period was almost the only time during the entire cropping cycle when $CO_2$ uptake via photosynthesis exceeded $CO_2$ losses from ecosystem respiration. NEE in the BAU treatment also became more negative at this time but remained a net source of around 2 g $CO_2$-C m$^{-2}$ d$^{-1}$. The BAU treatment did not become a net daily sink until another dip in soil temperature, occurring just before harvesting in late August when leaf area and photosynthesis were high.

Clear differences in NEE between treatments emerged immediately after crop establishment. This was notable for the HI treatment, which had daily emissions approximately 50% greater than the other two treatments at this time, whereas NEE was similar in the BAU and MI treatments. Two further peaks in $CO_2$ emission occurred in the HI treatment towards the end of July and in early August, in both cases following manual irrigation events. After the irrigation event on August 10$^{th}$, systematic differences in mean NEE emerged between all treatments, in the order HI > MI > BAU. These differences persisted after harvesting, when the treatments were stable sources of roughly 9.5, 7 and 4 g C m$^{-2}$ d$^{-1}$ for HI, MI and BAU treatments, respectively. Decomposition of wasted biomass following harvest may have contributed to elevated emissions during this period.

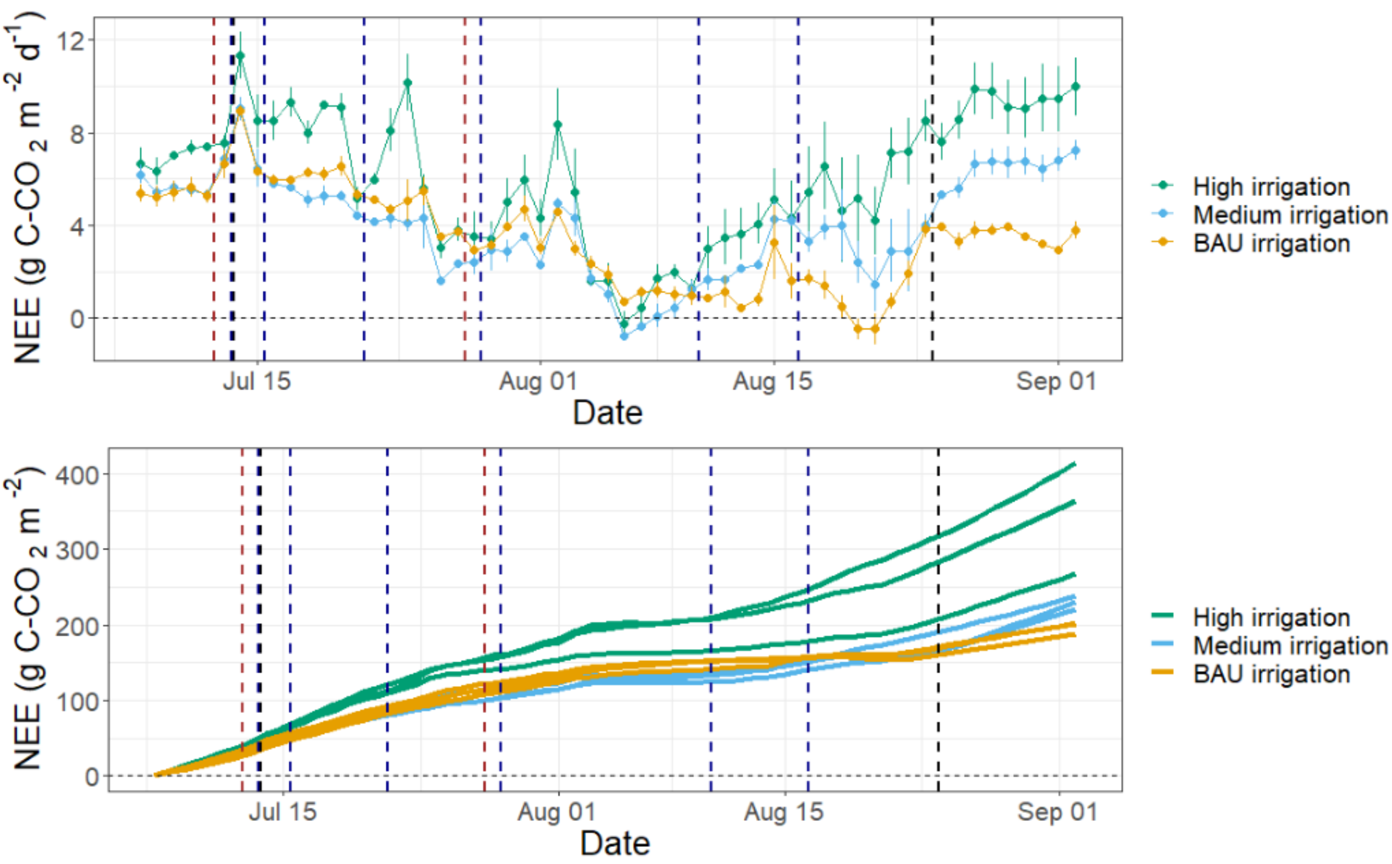



***Figure 5.7***. *Daily mean (top) $CO_2$ fluxes for each treatment and daily cumulative (below) $CO_2$ fluxes for each plot within treatment. Black vertical lines represent planting and harvest, brown lines represent fertiliser application events, and blue manual irrigations.*

### *$N_2O$ fluxes*

Daily mean and cumulative $N_2O$ exchange data are shown in Figure 5.7. Similar to $CO_2$, $N_2O$ emissions were highest on average from the HI treatment, and in the periods following planting and harvest. Prior to fertiliser application and ploughing/planting, $N_2O$ emissions were around 5 mg $m^{-2}$ $d^{-1}$, and this flux doubled for the HI treatment once fertiliser was applied. In general, $N_2O$ emissions increased following planting, and after all manual irrigation and fertilisation events, but showed an overall decline over the period of crop growth in all treatments. However, between-plot variability was high, and cumulative emissions differed considerably from individual plots within same irrigation treatment. It appears that management activities alone cannot explain all observed variance in $N_2O$ emissions. Following the harvest, $N_2O$ emissions rose again in the HI treatment, becoming twice as high on average as emissions from the MI and BAU treatments.

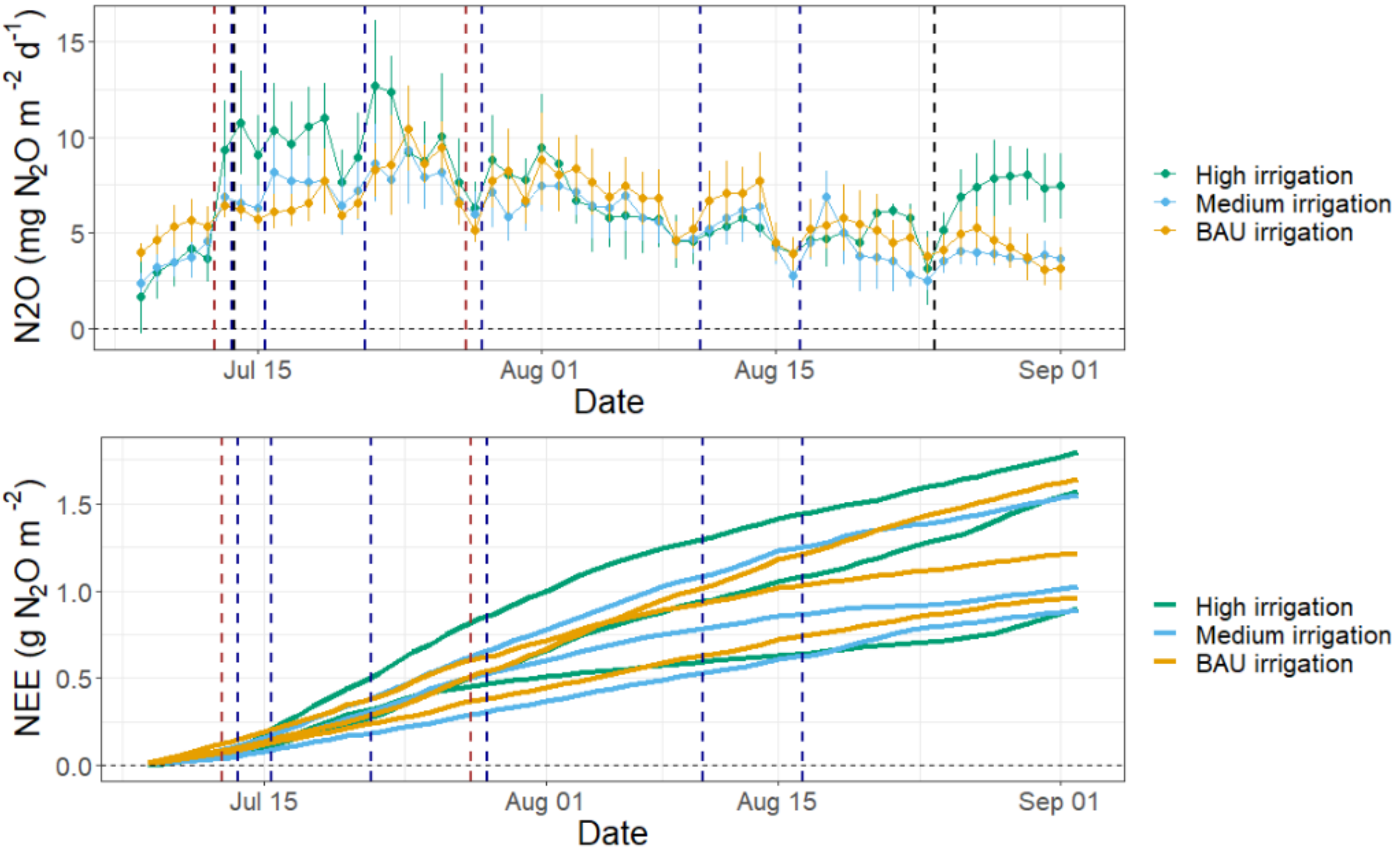



***Figure 5.8**. Daily mean (top) $N_2O$ fluxes for each treatment and daily cumulative (below) $N_2O$ fluxes for each plot within treatment. Black vertical lines represent planting and harvest, while brown lines represent fertiliser application events, and blue manual irrigations.*

### *Crop yield and condition*

The following is a summary of the results of the assessment undertaken by ADAS. See separate project report (Eyre et al. 2023) for a more complete description.

At harvest there was a visible difference between different treatments, with the BAU plants being smallest and plants in the MI treatment largest (Figure 5.9). These visible differences were confirmed by the fresh and dry weights of the marketable yield (Figure 5.10). The BAU plots had substantially lower marketable yield, approximately half that of the HI and MI plots. Plants harvested from within the collar area (inner) were generally smaller for all treatments than those outside of the collars, but the pattern between treatments remained consistent with the MI plots having the largest plants. Trimmed leaves followed a similar pattern to fresh weight but with the high irrigation appearing to require more leaves to be trimmed to bring the head to marketable specification.


***Figure 5.9.*** *Example of lettuce head growth in different treatment plots at harvest.*

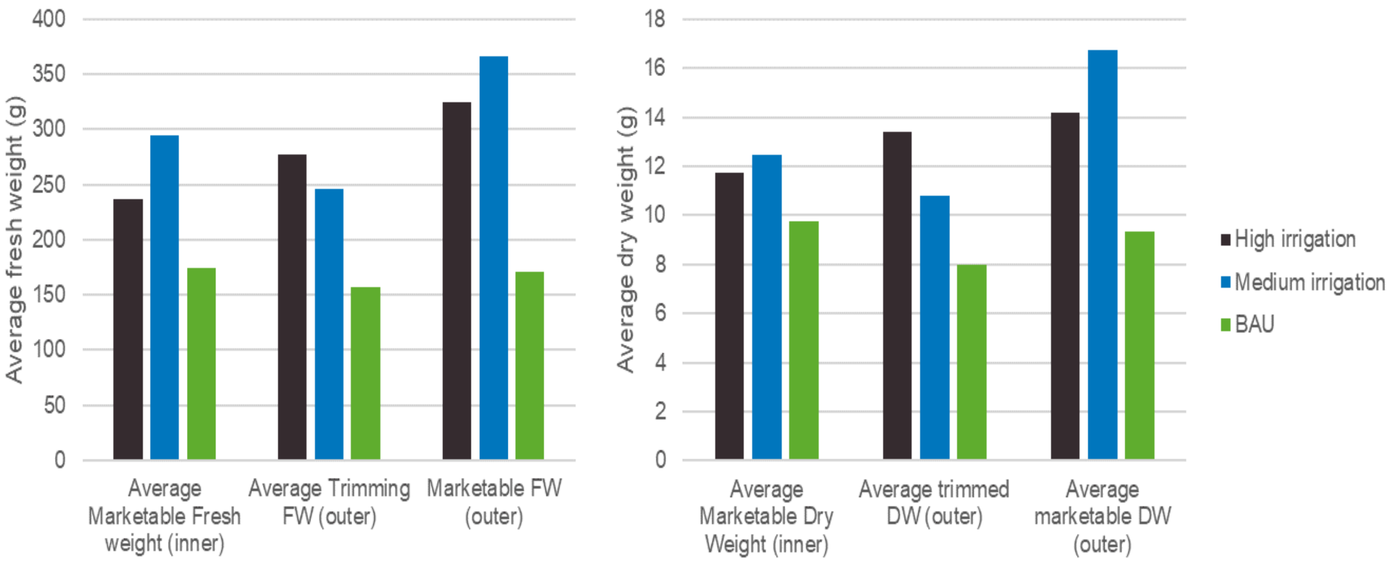



***Figure 5.10**. Average fresh weight of lettuces (left) and dry weight (right) of lettuces at harvest for the three irrigation treatments.*

Very little disease incidence was recorded in the plots, with no downy mildew or botrytis observed in any plots. Very low levels of *Sclerotinia* were detected, with no plants scoring more than a 1 on the disease severity scale, and with no more than 1-5% coverage of leaf symptoms for any one plant. Each of the three plots with high irrigation had some *Sclerotinia*-affected plants (1 to 5 per plot), but the majority of the plants were unaffected. A single plant in plot 8 with medium irrigation had some symptoms, and a single plant in plot 11 with BAU.

Results of the root analysis are shown in Table 5.2 and discussed fully in Eyre et al. (2023). In summary, root density was higher in the 40-60 cm horizon compared to the 20-40cm horizon, but with no clear differences between treatments. Mean dry weight followed a similar pattern (Figure 5.11), with highest average root dry weight in the HI plots. Mean root diameter was relatively consistent between horizons and treatments, but the specific root length (m $g^{-1}$) was higher in the 40-60 cm horizon for all treatments, where it also increased from the HI to MI to BAU treatments. The size distribution of roots was similar across all treatments with the highest proportion of roots in the smallest size category.

***Table 5.2.*** *Root analysis results for soil cores taken for lettuce plots immediately after harvest.*

| Treatment | Depth (cm) | RLD (cm/cm3) | Root DW (mg/cm3) | Mean Diam. (mm) | Specific root length (m/g) | 0 < L <= 0.5 | 0.5 < L <= 1.0 | 1.0 L <= 1.5 | 1.5 < L <= 2.0 |
|---|---|---|---|---|---|---|---|---|---|
| High irrigation | 20-40 | 0.21 | 0.01 | 0.20 | 202 | 86 | 3.68 | 0.037 | 0.000 |
| | 40-60 | 1.209 | 0.046 | 0.173 | 249 | 502 | 7.82 | 3.79 | 0.198 |
| Medium irrigation | 20-40 | 0.13 | 0.01 | 0.22 | 126 | 54 | 2.61 | 0.000 | 0.000 |
| | 40-60 | 0.876 | 0.028 | 0.162 | 309 | 368 | 3.80 | 0.00 | 0.000 |
| BAU | 20-40 | 0.28 | 0.01 | 0.18 | 210 | 116 | 0.85 | 0.000 | 0.000 |
| | 40-60 | 1.220 | 0.035 | 0.167 | 344 | 511 | 6.55 | 0.05 | 0.000 |

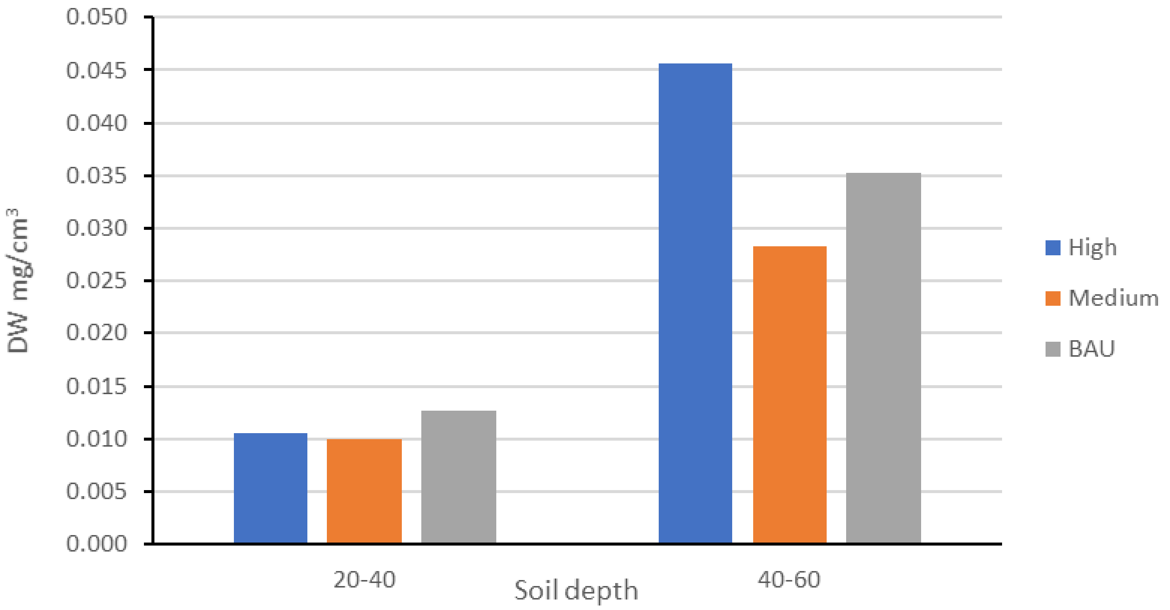



***Figure 5.11.*** *Mean dry weight (mg/cm$^3$) of roots sampled from lettuce plots with high and medium irrigation or BAU for 20-40 and 40-60 cm horizons of soil cores.*

## Discussion

### *GHG fluxes*

The NEP of 11.1 t C ha$^{-1}$ yr$^{-1}$ for the wheat crop in 2021 is at the high end of emissions measured by eddy covariance from croplands on lowland peat in the UK. It is similar to that obtained for a maize crop recorded on a thinner peat site in the Fens (11.9 t C-$CO_2$ ha$^{-1}$ yr$^{-1}$), but greater than those from the nearby Rosedene flux tower, which has recorded emissions ranging from 7.3 to 10.4 t C ha$^{-1}$ yr$^{-1}$ over a range of crops (including sugar beet, potatoes, lettuces and other salad crops, but not wheat) since 2012. The NEP measured by the Skyline 2D experiment is therefore within, but at the upper end of, the range of values obtained from flux towers in comparable locations. While this figure is therefore plausible, the annual flux estimate could have been affected by gaps in the Skyline 2D data during April and June 2021 (which were gap-filled, but errors are possible given that wheat growth peaked in May). Observed damage to the wheat within the chambers by the Skyline 2D chamber may also have influenced results, although lower resulting uptake via photosynthesis may have been largely cancelled out by the removal of less biomass C during harvesting. Previous disturbance of the plots during installation of the piling and subsequent attempts to seal the plots could also have affected fluxes, although there was no disturbance during the measurement period. Notwithstanding these uncertainties, it seems clear that annual net $CO_2$ emissions from the Skyline experimental site were high. The consistency of these results with the flux tower data, given that they are based on completely different methods (autochambers versus eddy covariance), theoretical foundations (gas accumulation versus turbulent transport), spatial scales (plots versus fields), at different locations and under different crops, provides a useful cross-validation and lends weight to the robustness of annual flux estimates for lowland agricultural peatlands. The results also add to the existing evidence base that remaining areas of thick peat drained for intensive crop production are among the most intense sources of $CO_2$ emissions in the UK land-use sector.

Due to the experimental nature and sub-annual duration of the 2022 lettuce dataset it is not possible to use the data collected to derive a full annual NEP value, or to compare results with other annual crop cycles monitored using eddy covariance. It is possible, however, to compare the experimental results to those for other lettuce crop growth periods measured at the nearby Rosedene flux tower (which is also on thick peat), and from the Redmere flux site which is roughly 10 km away on a thinner peat soil (Table 5.2). Mean values of soil temperature and VWC have also been calculated for these periods at these sites, though it is noted that different soil instrumentation was used at Rosedene (CS616, Campbell Scientific Inc., USA) compared to the Redmere and the Skyline experiment. Furthermore, the soil moisture sensors in this experiment were installed into recently ploughed soil, whereas those at the flux towers (excepting Rosedene in 2012) were all installed in more compacted peat soil and have been in stable (e.g. non-tilled) soils for several years. These differences in installation and instrumentation and the local soil environment could have influenced VWCs, although the comparatively low mean value for the BAU plots in the Skyline experiment is consistent with the severe drought conditions experienced during the study period. However, the comparatively low VWC values recorded in the plots receiving additional irrigation are surprising given the volumes of water added, and are hard to explain.

One striking difference between the Skyline experiment period and previous lettuce crops is the markedly higher soil temperatures experienced (21.6 °C compared to 12 to 17 °C in all previous lettuce growth periods). This likely drove higher rates of evapotranspiration, and could have contributed to lower than expected VWC in the irrigation treatments. However, the rapid drop in VWC following daily irrigation (Figure 5.3) indicates that much of the added water was being rapidly lost via gravity drainage rather than via evapotranspiration. This could simply mean that water was being added faster than the peat could absorb it, with excess water draining directly to the water table. However, given the extreme drought that preceded the experiment, and continued throughout it, it is also likely that the soil had become cracked and hydrophobic, and was therefore less able to retain the additional water. Since VWC was < 10% before the crop was planted, this explanation seems plausible. It could also account for the progressive increase in VWC observed in the HI treatments, as antecedent drought conditions had influenced the capacity of the soil to retain moisture recovered from an initial hydrophobic state. If this was indeed the case, the timing of the irrigation trial may have been sub-optimal, coinciding with a period when the peat had very little capacity to retain the additional irrigation water, let alone reach moisture levels that might begin to constrain peat decomposition rates.

***Table 5.2**. Net ecosystem exchange of $CO_2$, with mean VWC and soil temperature for crop growing period, for eddy covariance observations and skyline experiment over lettuce crops*

| **Site - variety (planting – harvest)** | **C-$CO_2$ exchange (g $CO_2$-C $m^{-2}$)** | **VWC** | **Tsoil (°C)** |
|---|---|---|---|
| **Eddy Covariance** | | | |
| Rosedene - iceberg (26-06-2012 – 12-08-2012) | 2.9 (48d) | 0.57 | 17.0 |
| Rosedene - iceberg (02-05-2014 – 27-06-2014) | 23.4 (60d) | 0.35 | 14.6 |
| Rosedene – mini romaine (13-04-2016 – 23-06-2016) | -48.6 (71d) | 0.27 | 14.3 |
| Rosedene - iceberg (26-04-2018 – 21-06-2018) | 85.6 (56d) | 0.49 | 17.5 |
| Redmere - mini romaine (21-03-2022 – 13-06-2022) | -3.2 (84d) | 0.32 | 12.1 |
| **Skyline Chamber (13-07-2022 to 24-08-2022) Romaine** | | | |
| High irrigation | 218.9 | 0.37 | 21.0 |
| Medium irrigation | 133.6 | 0.29 | 20.6 |
| BAU irrigation | 130.9 | 0.24 | 21.6 |

Comparing NEE data from the Skyline 2D study to previous eddy covariance data, it is clear that $CO_2$ emissions from the plots were unusually high. This cannot be explained by differences in soil moisture levels alone; the two previous periods with negative measured NEE had similar VWC (0.27 and 0.32 vs 0.24 for the Skyline BAU treatment). However, both of these periods were much cooler on average, at 14.3 and 12.1 °C, respectively, and the growth period was correspondingly longer. If we assume a $Q_{10}$ value of 2 (proportional change in decomposition rates for a 10 °C temperature increase, see Section 5.3), we would expect heterotrophic respiration rates to have been 50 to 90% higher under the lettuce crop in 2021 compared to these periods, if all other factors remained constant. Abnormally high temperatures in 2022 therefore probably contributed to the higher $CO_2$ emissions in all three treatments.

With regard to treatment effects, it is clear that irrigation caused some evaporative cooling of the MI and HI treatments, by an average of 1.0 and 0.6 °C respectively. This did not translate into higher $CO_2$ emissions from the BAU treatment, indicating either that very high temperatures had a negative impact on decomposition processes, or that factors other than temperature influenced the carbon balance. In a previous mesocosm study, Kechavarzi et al. (2010) found in some cases that a decreasing temperature could result in an increased respiration rate. However, once the lettuce canopy had closed (around the 10$^{th}$ of August) and limited direct solar heating of the soil surface, between-treatment differences in soil temperature largely disappeared, whereas differences in NEE became steadily greater. The

HI plots emitted roughly 2 g $CO_2$- C $m^{-2}$ $d^{-1}$ more than the MI treatments at this time, and the MI treatments emitted roughly the same amount more than BAU treatments. Post-harvest, a lowered water table and a large labile carbon source from crop residues likely contributed to higher $CO_2$ emissions from all treatments, but differences between irrigation treatments persisted and even increased during this period.

Given that differences in NEE were largest when differences in soil temperature were smallest, the thermal regime of the soil was clearly not the main controlling variable. Similarly, given that crop growth was greater in the HI treatment, lower emissions in the BAU treatment versus the HI and MI treatments cannot be attributed to differences in photosynthetic uptake. We therefore conclude that observed differences in NEE are due to higher heterotrophic respiration rates, resulting from higher soil water levels from surface irrigation. This was clearly not the desired outcome of the irrigation trial, but at this stage we cannot be sure whether it represents a generalised response to surface irrigation, or a specific result of the unusually warm and dry weather conditions during which the trial took place. Based on our interpretation of the soil moisture response above, it appears that surface irrigation likely alleviated pre-existing moisture constraints on microbial decomposition under drought conditions, possibly by reducing hydrophobicity of the soil as a result of extreme drying in the preceding heatwave. Despite applying a large volume of water in the irrigation treatments, most of this water seems to have passed rapidly through the peat to the water table (and ultimately back out to the ditch) without effectively re-wetting the soil surface beyond the field capacity of the peat soils. Even in the HI treatment, where VWC did progressively rise through the experiment, it still did not reach levels observed under ambient conditions during some of the previous lettuce crops at the Rosedene flux tower. It certainly never reached levels at which oxygen availability would be expected to constrain microbial activity, so there was no suppressive effect of irrigation on $CO_2$ emissions. In other words, it appears that additional irrigation under extremely dry conditions was sufficient to raise soil moisture *into* the range at which conditions were optimal for peat decomposition (which could then occur rapidly given the very high temperatures) but not increase *beyond* this range to the point where decomposition became constrained by oxygen availability.

For $N_2O$ emissions, results were somewhat similar to those for $CO_2$. While $N_2O$ fluxes are known to be highly temporally and spatially variable, the frequency of Skyline 2D autochamber measurements appears to have been sufficient to capture much of the spatial variation, and fluxes were generally large enough to permit fluxes to be measured accurately given the sensitivity of the analyser and size of the chamber. Certainly, hourly measurement is more frequent than most previously reported studies of $N_2O$ fluxes from peatlands (e.g. Leppelt *et al.,* 2014, Tiemeyer, *et al.,* 2016), and we have captured significant fluxes 5-10 mg $N_2O$ $m^{-2}$ $d^{-1}$ that are within the range of those reported for a potato crop in the Fenland region using the eddy covariance technique on a wasted peat (UKCEH, unpublished data for the BEIS Wasted Peat project). Furthermore, despite considerable between-plot variability, we observed clear and consistent differences between treatments during some time periods, which appeared consistent with site conditions and management activities.

Total $N_2O$-N emissions measured during the experiment are 3.3, 3.1 and 3.6 kg $N_2O$-N $ha^{-1}$ over 42 days for the BAU, MI and HI treatments, respectively. As partial annual values, these are obviously less than the annual Tier 2 emission factors for intensive grasslands and croplands on peat, which are 7.39 and 16.28 kg $N_2O$-N $ha^{-1}$ $yr^{-1}$, respectively (Evans *et al.,* 2022a), but greater than the other land uses on peat soils due to the addition of N fertiliser. While it would not be realistic to simply extrapolate measurements over the short experimental period out to a full year, our high measured emissions over one short cropping cycle (bearing in mind that the field was double-cropped during 2021) suggest that the Tier 2 emission factors may be broadly realistic, although the current emission factor for cropland does appear high.

The time series data (Figure 5.7) suggest that differences between irrigation treatments were small during the period of active crop growth, but larger (particularly for the HI treatment) in the periods immediately after initial planting, and post-harvest, when fluxes were highest from all treatments. These observations suggest that higher soil moisture levels may have influenced $N_2O$ emissions during periods of excess N availability and limited N demand from the crop, i.e. when there was likely to have been free nitrate in soil water. These findings are broadly consistent with those for $CO_2$ in suggesting that irrigation during the 2021 drought period was sufficient to raise soil moisture to levels that enabled denitrification of nitrate to $N_2O$ to occur, but not high enough to limit oxygen availability to the extent that all available nitrate was denitrified to $N_2$.

### *Crop performance*

Additional surface irrigation increased the marketable yield of the lettuce crop, with over two times higher fresh weight in both the MI and HI irrigation treatments compared to BAU irrigation in lettuces collected from outside the measurement collars. Yield increases were smaller when measured as dry weight, i.e. part of the enhancement in fresh weight was due to a higher water content. The level of yield enhancement did not increase from the MI to the HI irrigation levels. These results indicate that supplemental irrigation enhanced crop productivity, likely as a result of the exceptionally warm and dry 2022 growing season; soil moisture levels were exceptionally low, growers were using all of their available water, and the supplementary reservoir supplying irrigation water at Rosedene Farm was almost empty by the end of the season. It is therefore likely that BAU irrigation levels were suboptimal (as it appears that they were for soil microbial activity) and that moderate irrigation (i.e. the MI treatment) was sufficient to alleviate moisture limitations on crop growth.

Despite a slightly increased incidence of *Sclerotinia* in the high irrigation treatment plots the overall disease level was very low. This is in part due to the very hot, dry growing season that was experienced which was not conducive to disease development, with growers elsewhere experiencing a similarly good year for low disease incidence. The growing season for lettuce from planting to harvest was also relatively short so there was very little time for disease development to occur.

The root analysis was perhaps as expected for a lettuce crop, with roots developing relatively shallowly and spreading laterally. The higher root density in the 40-60 cm horizon vs the 20-40 cm horizon in all treatments indicates that these deeper roots were seeking water in an exploratory habit. Roots that are in an exploratory phase tend to be lower diameter as the plant puts less energy into producing thicker roots in favour of sending out longer roots to seek water. This effect was not seen between horizons, perhaps because the need to seek water was not so great given the surface irrigation. Variation between treatments was not consistent and may have been due to natural variability rather than a response to irrigation.

The main finding of the crop analysis is that the higher irrigation level was beneficial in terms of marketable yield, but that this result may have been a result of the prevailing drought conditions. In a wetter year, in which growth was not moisture-limited, further irrigation might not boost yields, but conditions might be more conducive to disease development. If water levels were raised in the field (e.g. to less than 40 cm from the surface) this would clearly impact on the typical rooting depth of the lettuce, but might require the plants to allocate less resources to exploratory roots to access water.

**Conclusions**

High-resolution autochamber measurements made using the Skyline experimental system recorded similar net $CO_2$ emissions (from a wheat crop) to those obtained using field-scale eddy covariance systems, suggesting that (despite some limitations) this method provides a reasonable measure of ecosystem-level GHG fluxes. Although it proved impossible to control water levels within the experimental plots (see following section for field-scale water level manipulation trials), we were able to test the hypothesis that surface irrigation offers an alternative means of mitigating $CO_2$ and $N_2O$ emissions in circumstances where raising water levels below the peat surface is not feasible. Based on measurements over a full lettuce crop cycle, undertaken during a period of severe drought, this hypothesis was not supported, and indeed irrigation appears to have increased both $CO_2$ and $N_2O$ emissions during some periods (whilst also enhancing crop yields). We cannot therefore recommend surface irrigation as a mitigation measure at this stage. However, it does appear that our results were strongly influenced by the extended drought conditions during the trial period. Prior to crop establishment soil moisture levels were exceptionally low, to the extent that microbial processes were likely limited by soil moisture, and the peat may have become hydrophobic. Irrigation under these conditions appears to have been relatively ineffective at raising soil moisture, because most of the water simply passed through the topsoil back into groundwater. Water that was retained was sufficient to activate soil microbial processes, including those leading to $CO_2$ and $N_2O$ emissions (and to increase lettuce growth and yields) but insufficient to limit oxygen availability and therefore the production of either $CO_2$ or $N_2O$. Repeating the experiment with high antecedent soil moisture levels and/or higher groundwater levels would indicate whether irrigation might be more effective at reducing GHG emissions under these conditions, and provide a better overall understanding of the interactions between soil moisture, GHG emissions and crop yields.

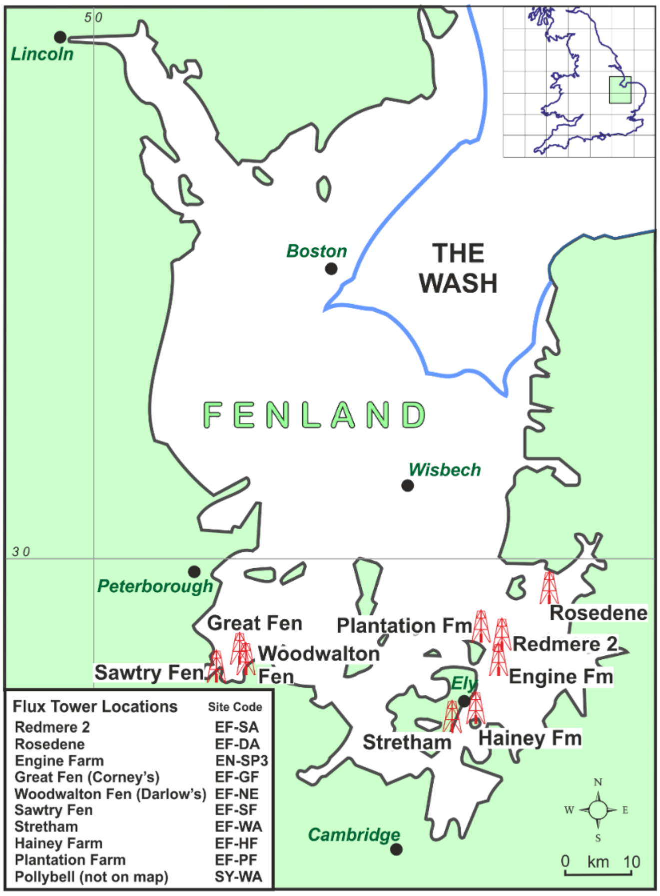
## 5.3 Analysis of flux tower data

The lowland peatland flux tower measurement programme has continued to expand during the Defra SP1218 project period. In addition to the long-term agricultural monitoring sites at the Redmere (2012- present, but split across three locations on the farm), Rosedene (2012-present) and Engine (2018 - present) farm sites, new agricultural peatland flux sites have been established at Sawtry (2020 - present), Stretham (2021 - present) and Pollybell (2021-present). An additional SP1218 flux tower was also deployed at Stretham during the summer of 2022, providing flux tower measurements of high water arable (winter wheat) management for comparison against business as usual (BAU) arable management. Note that all sites are in the Fenland region apart from Pollybell, which is in the Isle of Axholme area of north Nottinghamshire, adjacent to the Humberhead Levels. Site locations for the Fenland sites are shown in Figure 5.12, and soil profile descriptions in Figure 5.13.

Four independently-funded flux towers have also been established by the Fenland SOIL farmer group during the course of the Defra project, on sites at Hainey Farm (2021 - present), Plantation Farm (2021 - present), Chatteris (2022 - present), and Yaxley (2022 - present). Data from Hainey (thin peat) and Plantation (wasted peat) were kindly made available to the project by Fenland SOIL and included in the flux tower data synthesis undertaken to support the development of an emission factor for cropland on wasted peat (Evans et al., 2022b). The Plantation site has continued to operate well during 2022, while the Hainey site experienced instrument failure during the summer of 2022, so it will not be possible to derive a new annual flux for this site (although it is now running again). Chatteris and Yaxley began operating in October 2022, so 2023 will be the first year for which annual fluxes can be produced.

New flux towers have also been established at areas of lowland peat managed as grassland. Extending previous flux tower studies at Corney's at Great Fen (2017 - 2021), Bakers Fen at Wicken Fen, and Tadham Moor on the Somerset Levels (2012 - 2021), new grassland sites were established at Woodwalton Fen (2020 - present) and a new location within the Great Fen project area (June 2022 - present). However, these two new grassland sites are scheduled to be relocated to a new paludiculture and restoration trial (both on former arable land) within the Great Fen project area during spring 2023. This new measurement programme will target the first commercial scale *Typha* grown as a dedicated paludiculture crop in the East Anglian Fens, together with a transition from arable to restoration management on deep peat. This new work will be funded by the WTBCN-led Peatlands Progress project, but will mean that grasslands again become a gap in the lowland peatland monitoring network.

***Figure 5.12****. Map of flux tower sites on the Fenland region.*



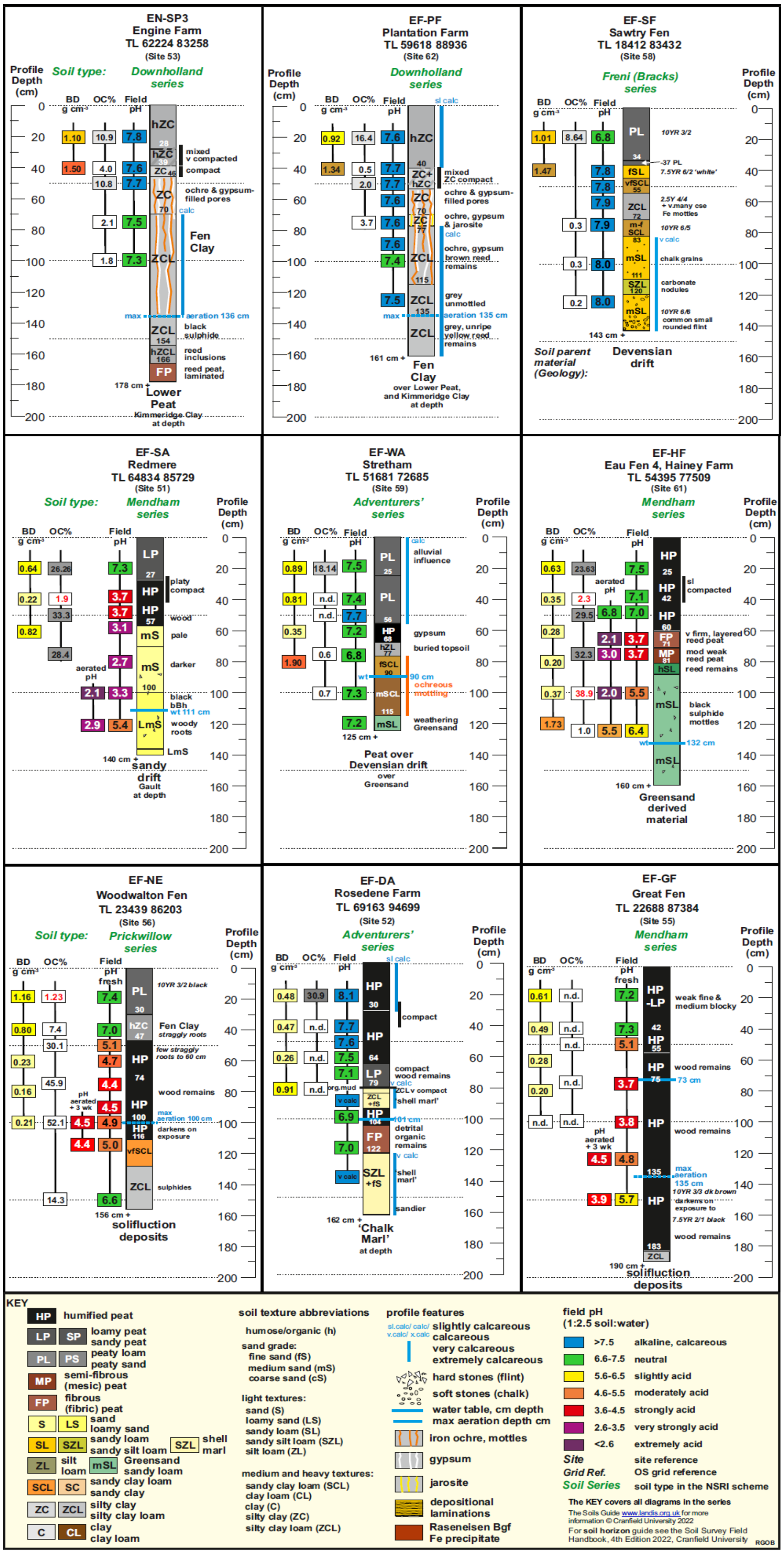



***Figure 5.13**. Soil profiles for nine Fenland flux towers sites (including sites operated by UKCEH, the University of Leicester and Fenland SOIL) ranked from least to most peaty. Profile analyses and illustrations by Rodney Burton*

## Site level hydrology

As well as measuring $CO_2$ fluxes (see below), all flux tower sites measure the main components of the peatland water balance (Figure 5.14). Hydrological measurements at flux sites include fluxes of water via precipitation (P, mm) and evapotranspiration (ET, mm), as well as changes in water storage reflected by the position of the water table relative to the peat surface (m) and the volumetric water content (VWC, $m^3$ $m^{-3}$) of peat at different depths (not shown). In particular, soil hydrology represents an overriding control on greenhouse gas emissions from peatlands (Evans et al., 2021). Beyond the simple water budgets presented below, there is considerable scope to use these data to better understand the interactions between meteorological conditions, land-management, soil hydrology and GHG emissions. A full analysis is beyond the scope of the current project but could inform decision-making on how to balance emissions mitigation options with issues of regional water supply and demand. This issue is briefly considered in Section 5.5.

Examples of time series plots of water fluxes and changes in storage are shown in Figure 5.14 for each of the lowland peat flux tower sites, along with summary data in Table 5.3. Annual rainfall totals vary across the spatially distributed network of lowland peat sites. Irrigation (not shown in Figure 5.14) represents a further component of the water balance during the cultivation of some crops, particularly horticultural salad crops and potatoes which receive overhead irrigation. ET follows clear seasonal trends across sites, with low daily values in the winter and attaining maximum values during the peak growth stage for each crop due to seasonally high transpiration rates. For cereals (wheat at Stretham and Sawtry, barley at Pollybell), ET showed a rapid and sharp decline during the senescence stage prior to harvest. ET rates are subsequently dominated by evaporation from soil (and intercepted rainfall) for the remainder of the season at agricultural sites, when soils are bare or sparsely vegetated. The decline in ET was particularly strong for cereal crops during the extended drought conditions during summer 2022.  By contrast, other crops (potatoes, peas, broccoli) had higher rates of ET later in the season, reflecting the different growth pattern of these crops (and hence transpiration rates) relative to cereals. Grasslands have a more consistent pattern of ET, but with large between-year variability. For example, ET at Woodwalton was notably lower during the 2022 drought relative to the previous year.

Temporal variation in water levels is controlled by the balance between P and ET, as well as drainage and irrigation management practices (e.g. for potatoes at Stretham in 2022, lettuce and celery crops at Redmere during 2022). The impact of the summer 2022 drought was particularly evident at Sawtry and Stretham and at the Woodwalton grassland, when water levels were notably lower than during previous growing seasons. By contrast, water levels at Pollybell did not decline as far during 2022 compared to the previous growing season. This is explained by the regional drainage regime, specifically by higher levels of pumping during the early spring of 2021 that were not compensated for by rainfall for the remainder of the season. Similarly, at Redmere, water levels were maintained at higher levels for lettuce and celery cultivation in 2022 compared to maize production the previous year. A defining characteristic of the lowland agricultural peatland sites is that water levels are typically located within the subsoil underlying the peat. At Woodwalton, water levels were within the peat layer during the winter months, but dropped into the subsoil during the growing season.

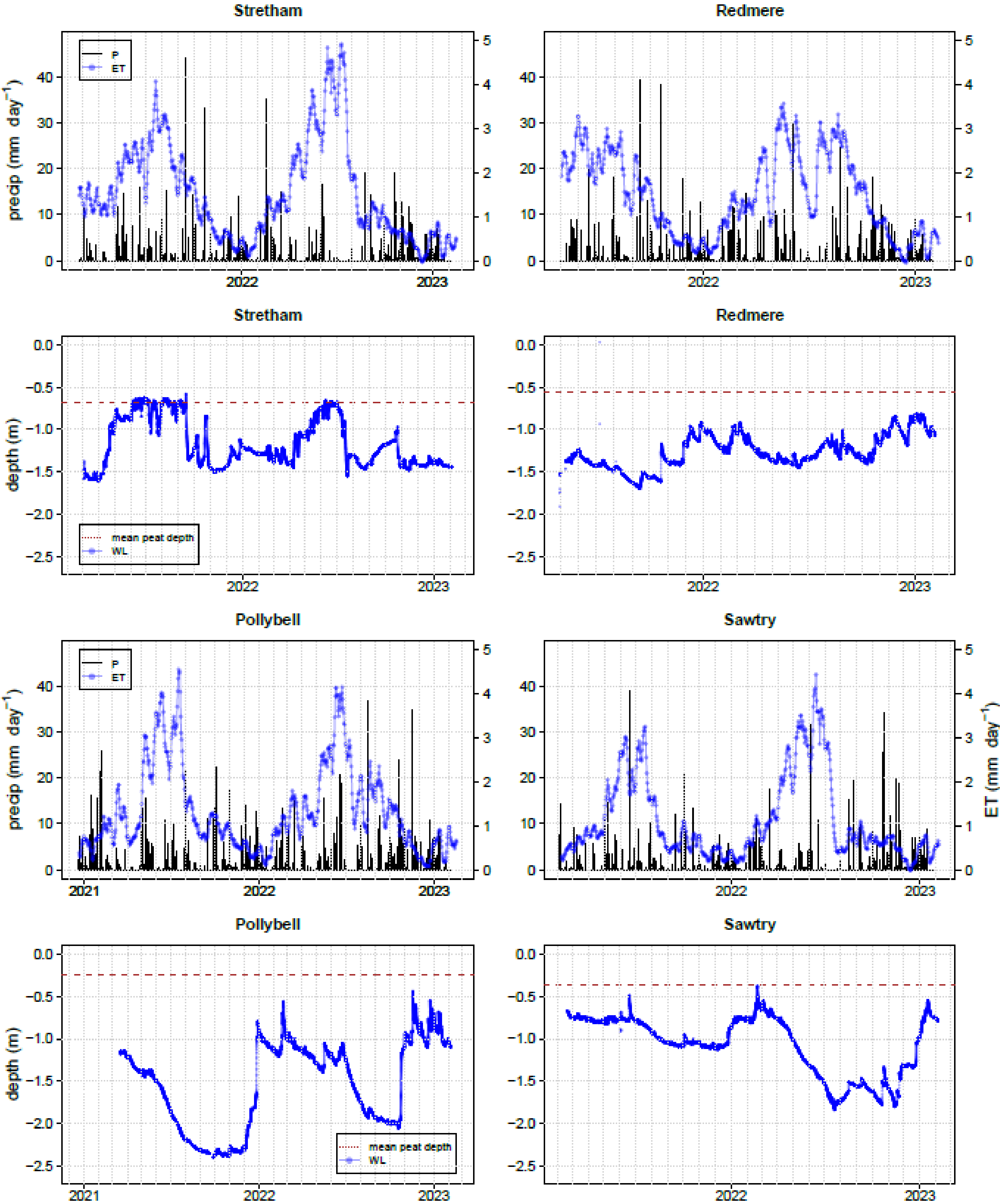



***Figure 5.14***. *Examples of the main components of the peatland water balance measured at lowland peatland flux tower sites. Upper plots for each site show daily precipitation and evapotranspiration, lower plots show water level relative to the ground surface. Horizontal lines in the water level plots show estimated mean depth of the peat layer for that site.*

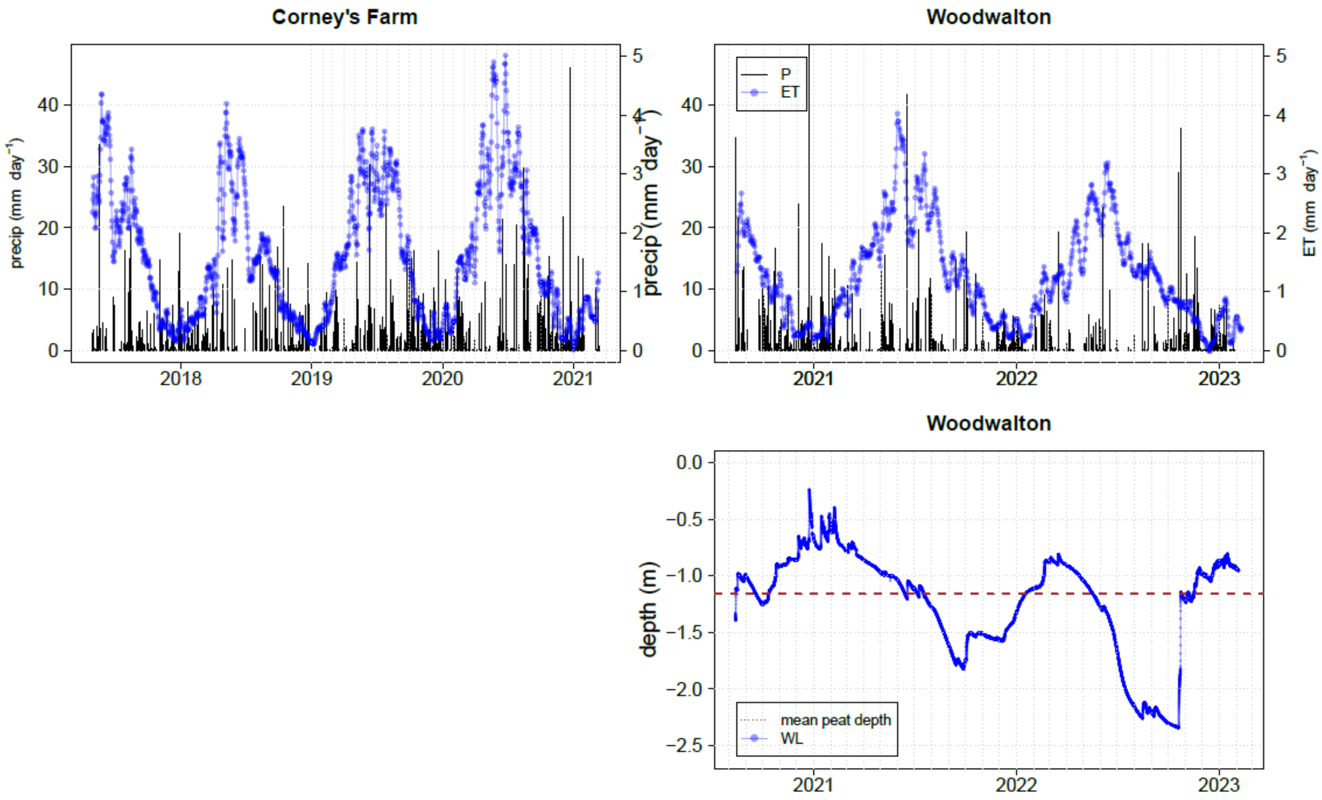



***Figure 5.14 (continued)***. *Note no water level data for Corney's Farm.*

***Table 5.3***. *Summary of main components of the water balance measured at Lowland Peatland flux tower sites. P = precipitation, ET = evapotranspiration, ET std = standard deviation of evapotranspiration, Mean WL = mean water level relative to field surface. Note no water level data for Corney's Farm.*

| Site | Year | Crop | P | ET | *ET std* | Mean WL |
|---|---|---|---|---|---|---|
| | | | mm $H_2O$ year$^{-1}$ | | | m |
| Stretham | 2021 | Potato | 503 | 546 | *2.3* | -1.11 |
| | 2022 | Winter wheat | 418 | 553 | *2.1* | -1.22 |
| Sawtry | 2021 | Peas | 418 | 359 | *1.6* | -0.90 |
| | 2022 | Winter wheat | 544 | 458 | *1.7* | 1.26 |
| Pollybell | 2021 | Sping barley | 596 | 504 | *2.3* | -1.89 |
| | 2022 | Broccoli | 641 | 509 | *2.3* | -1.31 |
| Redmere | 2022 | Lettuce/Celery | 542 | 555 | *2.5* | -1.20 |
| Corney's Farm | 2018 | Grass (grazing) | 477 | 498 | *1.9* | - |
| | 2019 | Grass (grazing) | 659 | 596 | *2.3* | - |
| | 2020 | Grass (grazing) | 670 | 688 | *2.1* | - |
| Woodwalton | 2021 | Grass (grazing/hay) | 599 | 521 | *1.7* | -1.17 |
| | 2022 | Grass (grazing/hay) | 493 | 450 | *1.5* | -1.42 |

## Site level carbon budgets

Carbon fluxes and annual carbon budgets (representing annual net ecosystem production, NEP) are now available for multiple site years across multiple lowland peatland sites, including sites under both arable and grassland management, and covering gradients of peat depth and condition. Here, new data collected over the duration of the Defra SP1218 Lowland Peatlands Phase 2 project are reported (Table 5.4). These new flux tower data are placed into the wider and longer-term context of the Lowland Peatlands research programme in the synthesis section below. These data now allow annual $CO_2$ emissions for arable sites to be differentiated according to different peat depth categories, namely thick peat (≥ 1 m), thin peat (0.4 m to 1 m) and wasted peat (≤ 0.4 m). Annual carbon budgets are currently available for grasslands on thick (Woodwalton, Tadham Moor) and wasted peat (Bakers Fen).

## Cropland sites

Daily and cumulative $CO_2$-C flux terms for the lowland peat flux tower sites were calculated for both NEE and NEP. The NEE term represents the dynamic balance between the $CO_2$ assimilated during photosynthesis (gross primary productivity, GPP), and its release back to the atmosphere via the combined respiration (total ecosystem respiration, TER) of plants (autotrophic respiration, Ra) and soils (heterotrophic respiration, Rh). The sign of the NEE therefore determines whether a site is functioning as a net *in situ* sink (negative flux) or source (positive flux) for atmospheric $CO_2$ over a given period of summation (e.g. daily, annual). Step changes shown on the cumulative graphs represent lateral imports and exports of C from the sites in the form of seeds and peat plugs, and harvested biomass (produce and residues), respectively. To construct site-level C budgets, it was assumed that all C leaving the fields is ultimately respired back to the atmosphere in the form of $CO_2$, and that the NEP of the site represents the magnitude of the annual $CO_2$ sink or source strength, or annual C balance. Whilst this assumption holds at site scale (which is the focus of this report) it should be noted that this does not represent the full life cycle emissions associated with these crops.

***Table 5.4***. *Summary of new annual carbon budgets measured at lowland peatland eddy covariance sites during the Defra SP1218 Lowlands Peatlands Phase 2 project.*

| Site | Year | Crop | NEE | *St Dev* | GPP | TER | IMPORT | EXPORT | NEP | ET | *St Dev* |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | tonnes C ha$^{-1}$ year$^{-1}$ | | | | | | | mm $H_2O$ year$^{-1}$ | |
| Stretham | 2021 | Potato | 2.31 | *0.07* | 11.57 | 13.88 | 0.27 | 3.30 | 5.34 | 546 | *2.3* |
| | 2022 | Winter wheat | -0.85 | *0.07* | 17.49 | 16.64 | 0.00 | 6.60 | 5.75 | 553 | *2.1* |
| Sawtry | 2021 | Peas | 2.82 | *0.04* | 7.46 | 10.29 | 0.00 | 0.78 | 3.61 | 359 | *1.6* |
| | 2022 | Winter wheat | -4.29 | *0.07* | 17.91 | 13.61 | 0.00 | 6.97 | 2.68 | 458 | *1.7* |
| Pollybell | 2021 | Spring wheat | -2.11 | *0.08* | 14.53 | 12.42 | 0.00 | 5.13 | 3.02 | 504 | *2.3* |
| | 2022 | Broccoli | 3.64 | *0.06* | 9.14 | 12.78 | 0.87 | 1.37 | 4.13 | 509 | *2.3* |
| Redmere | 2022 | Lettuce/Celery | 6.44 | *0.07* | 9.88 | 16.31 | 1.00 | 1.08 | 6.51 | 555 | *2.5* |
| Corney's Farm | 2018 | Grass (grazing) | 3.51 | *0.08* | 16.22 | 19.73 | 0.00 | 0.00 | 3.51 | 498 | *1.9* |
| | 2019 | Grass (grazing) | 3.54 | *0.09* | 19.72 | 23.26 | 0.00 | 0.00 | 3.54 | 596 | *2.3* |
| | 2020 | Grass (grazing) | 3.07 | *0.09* | 19.58 | 22.65 | 0.00 | 0.00 | 3.07 | 688 | *2.1* |
| Woodwalton | 2021 | Grass (grazing/hay) | 1.43 | *0.07* | 20.49 | 21.92 | 0.00 | 1.94 | 3.37 | 521 | *1.7* |
| | 2022 | Grass (grazing/hay) | 2.72 | *0.07* | 18.63 | 21.34 | 0.00 | 0.65 | 3.36 | 450 | *1.5* |

### *Stretham, East Anglia*

The Stowbridge Farm (F. C. Palmer & Sons Ltd.) site is located close to the village of Stretham, Cambridgeshire. The site is used for rotational production of cereals and vegetables (potatoes, beets, etc.). Peat soils at the site were surveyed under the BEIS Wasted Peatland Project which ran in parallel to SP1218. This survey described the peat soils at Stretham as earthy peat of the Adventurers series, with a 0.56 m layer of peaty loam overlaying 0.12 m of humified peat and a further 0.09 m of humose silt loam, identified as buried topsoil. Underlying mineral soils are silty clay loams. Subsequent survey work (n = 16) across a wider area of the site has shown that peat soils are heterogeneous across the site (see BEIS report), with notable spatial variation in peat depth and plough layer. The flux tower is situated at a corner of a large field where it meets other large fields to the north and east, but with most of the measurements representing the large field to the southwest of the flux tower. When the three fields were under the same land management they were treated as a single unit. When the field to the north was managed differently (in 2022), flux measurements were constrained to fields that were under the same cropping pattern.

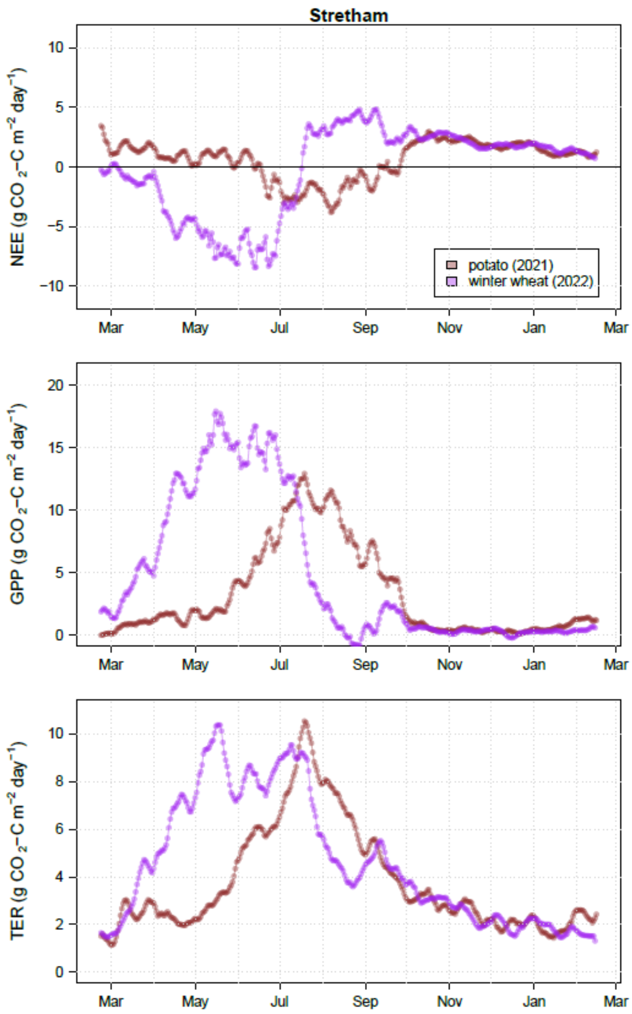



***Figure 5.15**. Measured net ecosystem exchange (NEE), gross primary production (GPP) and total ecosystem respiration (TER) for the Stretham flux tower under a potato crop (2021) and wheat crop (2022).*

The Stretham flux tower has been operational since 20th February 2021. The site was initially established for closed-path eddy covariance measurements of $CO_2$ and $N_2O$ under the BEIS Wasted Peat project and is the only arable peatland in the UK (and one of only a few globally) where emissions of $N_2O$ are being continuously monitored using eddy covariance. Additional sensors for monitoring $CO_2$ and water fluxes using standard (open-path) sensors were also installed to align data collection with other lowland peatland (and other) sites across the UKCEH-led national flux monitoring programme.

Over two years of continuous eddy covariance measurements are now available for the Stretham site (Figures 5.15, 5.16), capturing an irrigated potato crop during the 2021 production season (27th April to 14th November 2021) and a subsequent late-sown winter wheat (‘*Illustrious*’) crop (17th November 2021 to 3rd August 2022) and winter period. To construct annual $CO_2$ budgets for Stretham, the carbon year ran from 20th February to 19th February in both years rather than for full calendar years, reflecting the timing of the start of data acquisition at this location. For potato, photosynthesis (GPP), respiration (TER) and net C uptake (negative NEE) attained highest values between March and October, with maximum $CO_2$ uptake observed during the midsummer period. By contrast, the magnitude of GPP, TER and net $CO_2$ uptake was significantly higher for the winter wheat crop, with the main period of net $CO_2$ uptake and the largest gross fluxes (GPP, TER) observed between March and the senescence of the wheat in mid-July. The site functioned as a large net source for atmospheric $CO_2$ during the extended drought that followed the wheat harvest in 2022, with elevated daily $CO_2$ losses continuing throughout the year relative to the potato crop of 2021. Despite marked differences in the seasonal pattern of $CO_2$ gain and loss, and in the magnitude of gross fluxes, the Stretham site was a remarkably stable net source for $CO_2$ in both years after accounting for C export in harvested produce at 19.6 ± 0.2 and 20.9 ± 0.3 tonnes $CO_2$ $ha^{-1}$ $year^{-1}$ for potato and winter wheat for the years centred on the 2021 and 2022 cropping seasons, respectively (Figure 5.16a, Table 5.5).

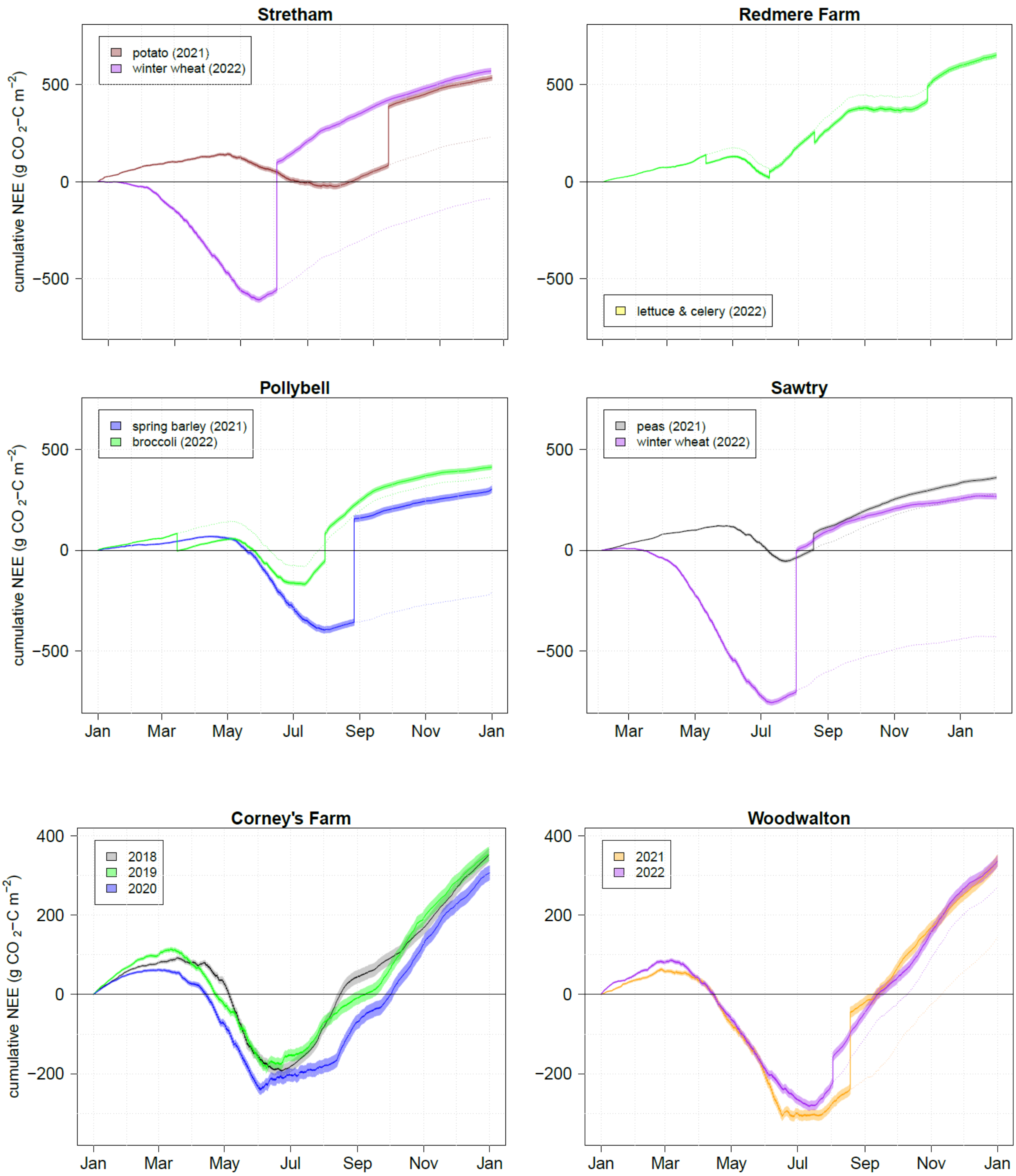



***Figure 5.16.*** *Cumulative carbon dioxide exchange carbon flux terms monitored at lowland peatland flux tower sites. Step changes illustrate lateral imports and exports of carbon. Shaded areas show the cumulative uncertainty associated with data gap-filling. Light dotted lines show the in-situ net ecosystem $CO_2$ exchange measured by the flux towers.*

*Redmere Farm, East Anglia*

New eddy covariance data are also available for Redmere Farm for 2021 (partial annual coverage) and 2022 (full coverage). Redmere represents one of the longest running eddy covariance sites in the Fenland region (active since October 2012), but measurements have now been made at three separate locations around the farm. The most recent flux tower data were obtained within a large (circa 44 ha) field close to the site of the first flux tower measurements between 2012 and 2018. Previously, this area of the farm was managed as three separate fields separated by drainage ditches. Ditches were in-filled during summer 2018 and the combined area is now under consistent cropping. Peat thickness is around 0.57 m (Figure 5.13). Flux measurements were made at the site from 29th April 2021 and are ongoing. Data collection at this site was part-funded by the Fenland SOIL group.

The cropping pattern at Redmere (Figure 5.17, 5.16) was maize (27th April to 21st October) in 2021 followed by dual cropping of lettuce (6th April to 4th June) then celery (16th July to 27th October) in 2022. In 2021, the maize crop was associated with a period of high $CO_2$ uptake and large gross $CO_2$ fluxes between June and the maize harvest in October (Figure 5.17), with net $CO_2$ emission at all other times of year. By contrast, the dual cropping pattern in 2022 resulted in two distinct periods of high GPP. TER remained high throughout the summer period of 2022, with the highest values coinciding with the peak growth phases of the two salad crops. A period of modest $CO_2$ uptake was observed for the lettuce crop during May, but the celery crop functioned as a small net $CO_2$ sink on just a few days, as GPP was outweighed by seasonally high heterotrophic respiration rates. The highest *in situ* net $CO_2$ emissions at Redmere were observed during the intercropping period, reflecting the combination of high summer temperatures, rapid decomposition of lettuce residues, and low levels of GPP at this time. This pattern is like that of previous measurements of dual cropping years at the same farm. Total annual $CO_2$ emission to the atmosphere from the Redmere site (Figure 5.16, Table 5.4) was 6.5 ± 0.7 tonnes $CO_2$-C ha$^{-1}$ year$^{-1}$ (23.9 t $CO_2$ ha$^{-1}$ yr$^{-1}$) for the dual crops of lettuce and celery in 2022. Imports of C as peat plugs (1.0 tonnes C ha$^{-1}$ year$^{-1}$) and exports in harvested biomass (1.1 tonnes C ha$^{-1}$ year$^{-1}$) were almost in balance. It was not possible to calculate a $CO_2$ budget for the partial year of data collected in 2021.

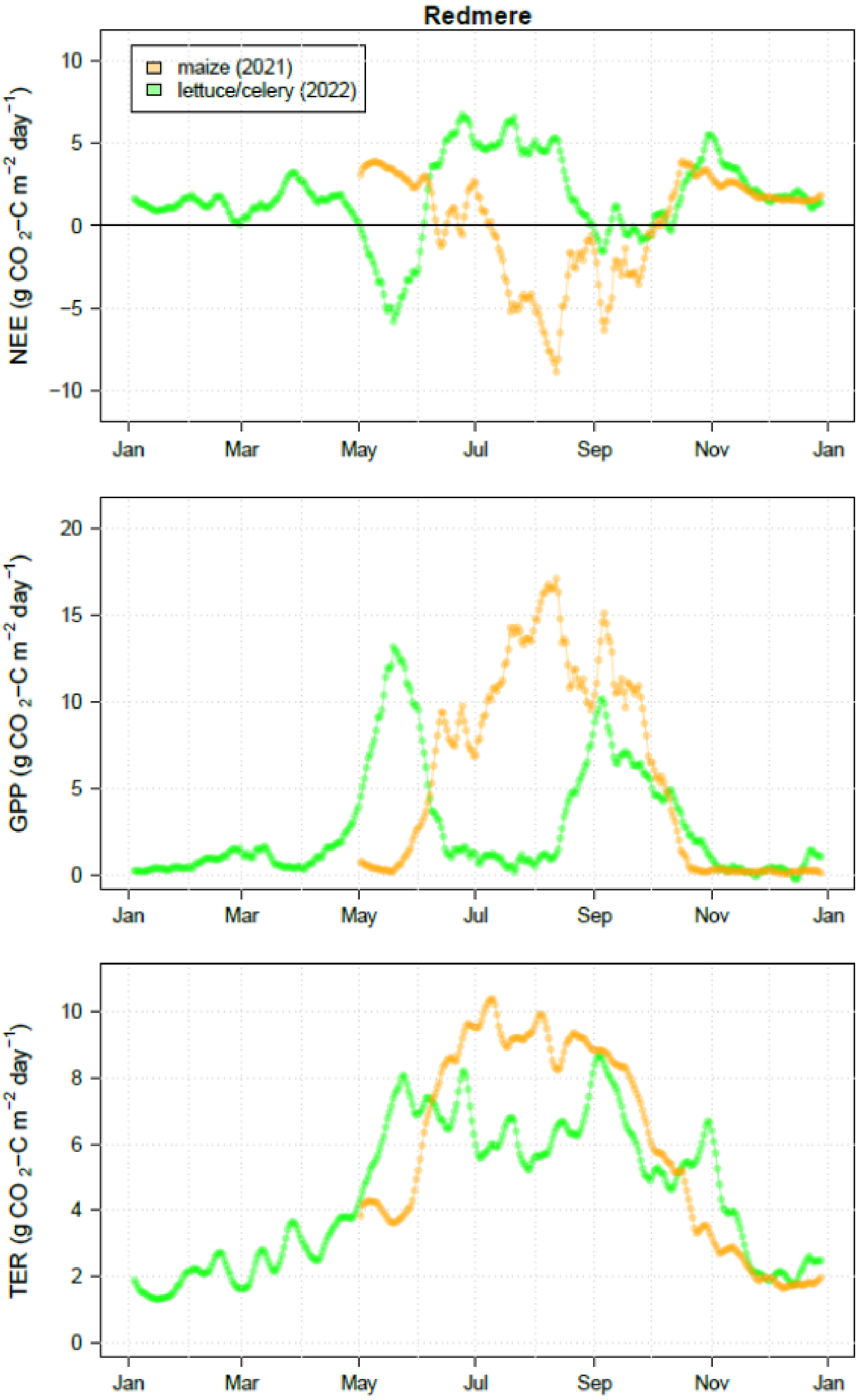



***Figure 5.17****. Measured net ecosystem exchange (NEE), gross primary production (GPP) and total ecosystem respiration (TER) for the Redmere flux tower under a maize crop (2021) and dual lettuce and celery crop (2022). Note that measurements did not start until April 29th 2021.*

*Sawtry, East Anglia*

The Sawtry flux tower site is located at T. C. Darby & Sons Ltd, close to the village of Sawtry, Cambridgeshire. It was established with direct funding from SP1218. The flux tower was installed within the northwest quarter of a large field to maximise the fetch along the predominant south-westerly wind direction within that field, but surrounding fields also have broadly similar peat soils and were managed under the same crop rotation throughout the data collection period. Peat soils are characterised by a 0.37 m layer of peaty loam over sandy and silty clay loam (Figure 5.13), classified as a humic gley soil of the Freni (Bracks) series. The Sawtry site is managed using a no-till and cover cropping regime, where crops are drilled directly into the peat after cover crops are sprayed off with Glyphosate.

The Sawtry site was used to grow peas in 2021 (30$^{th}$ March to 18$^{th}$ August) and winter wheat ('Skyscraper 2022' variety, 2$^{nd}$ September 2021 to 2$^{nd}$ August 2022). (Figure 5.18, 5.16). Like the Stretham site, the carbon balances were calculated over a period that reflects the start of high-quality data acquisition at the site (4$^{th}$ February to 3$^{rd}$ February in both years), rather than calendar years.

The Sawtry field was a net source for atmospheric $CO_2$ during spring and autumn for peas in 2021, becoming negative but at relatively low levels of daily GPP, TER and net C uptake between June and July. As expected within the same region and year, the gross and net fluxes for winter wheat showed a broadly similar pattern to Stretham in 2022, with the highest daily NEE and gross $CO_2$ exchange observed between spring and harvest in July. The peak season magnitude of total daily GPP for winter wheat was broadly similar between Sawtry and Stretham, but lower respiration fluxes were measured at Sawtry resulting in a higher overall daily $CO_2$ uptake (i.e. negative NEE). Sawtry functioned as a net $CO_2$ source after harvest in both years, but with lower levels of net $CO_2$ emission following wheat as a function of slightly elevated GPP from cover crops and volunteer species.

The inter-year cropping pattern resulted in large differences in gross $CO_2$ fluxes, with annual GPP and TER sums 42 % and 75 % smaller for peas than for winter wheat (Table 5.4), respectively. The total annual $CO_2$ emission from Sawtry was higher for the 2021 growth year (peas) compared to the same period of 2022 (winter wheat) after accounting for C inputs and offtake. Total annual emissions (NEP) were 3.6 ± 0.04 2.7 ± 0.07 t C ha$^{-1}$ yr$^{-1}$ (13.2 ± 0.1 and 9.8 ± 0.2 tonnes $CO_2$ ha$^{-1}$ year$^{-1}$) for peas and wheat, respectively (Table 5.4, Figure 5.16). These emissions are 6.4 and 11.1 tonnes $CO_2$ ha$^{-1}$ year$^{-1}$ lower than for the respective annual values for the thicker peat Stretham site (Table 5.4).

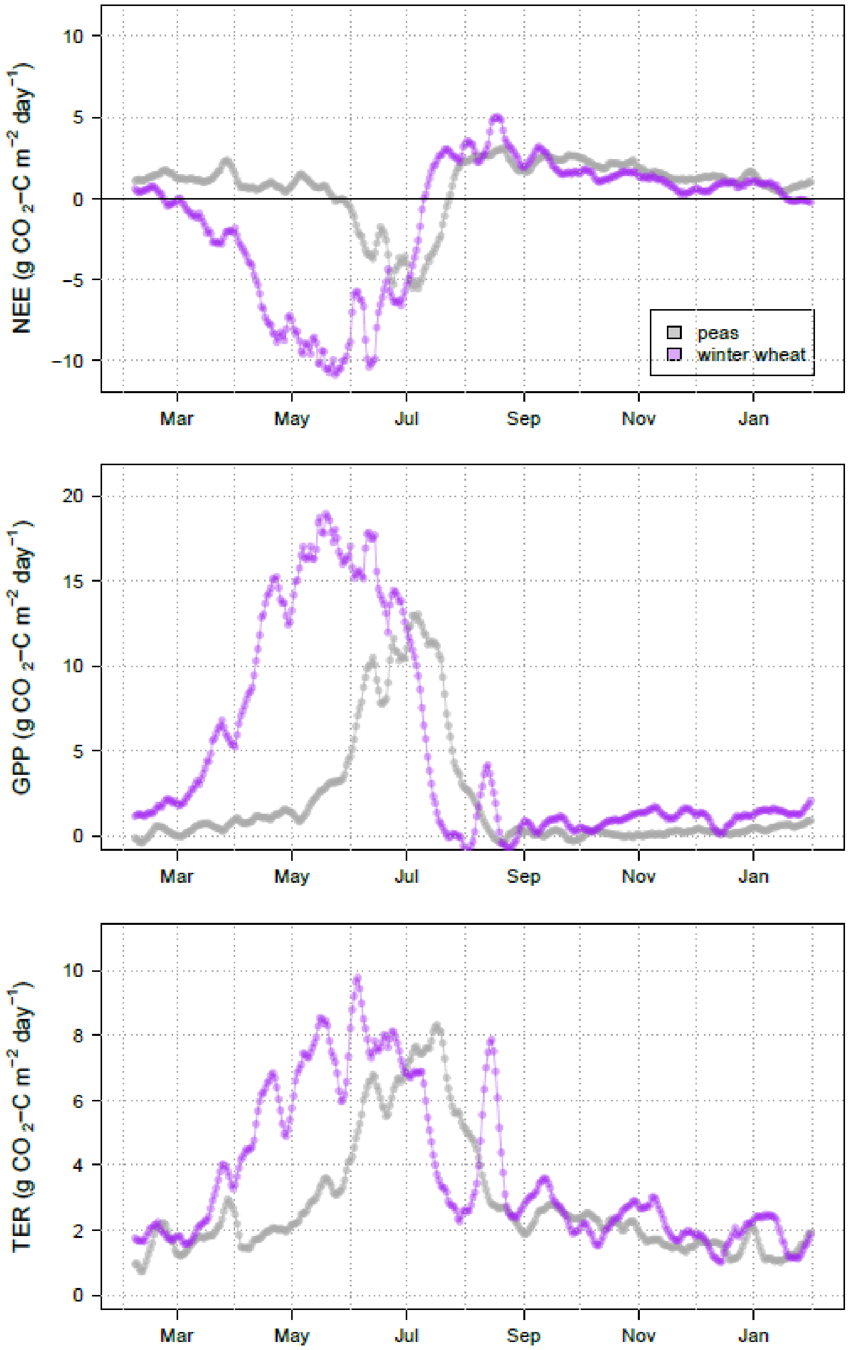



***Figure 5.18***. *Measured net ecosystem exchange (NEE), gross primary production (GPP) and total ecosystem respiration (TER) for the Sawtry flux tower under a pea crop (2021) and wheat crop (2022).*

### *Pollybell Farm, North Nottinghamshire*

The Pollybell Farm flux tower site in North Nottinghamshire forms part of the Isle of Axholme area of the Humberhead Peatlands. Pollybell is an organically managed farm that produces cereals and high-value fresh vegetables, with rotational livestock grazing elsewhere on the farm. Peat soils are characterised by a 0.24 m humified peat layer over sandy mineral substrate (Figure 5.13), classified as a humic-alluvial gley soil of the Everton series. The site has the thinnest peat layer of all the lowland peatland flux measurement sites (Table 5.4), but a clear differentiation between peat and mineral horizons (Figure 5.13) suggests that the site has not been deep ploughed (e.g. no significant mixing of peat and mineral layers). Pollybell was established as one of two new flux measurement sites under the BEIS Wasted Peatlands programme for measurements of $CO_2$ and $N_2O$. Similar to the Stretham site, UKCEH also installed a standard set of eddy covariance sensors to maintain compatibility with the wider national monitoring programme.

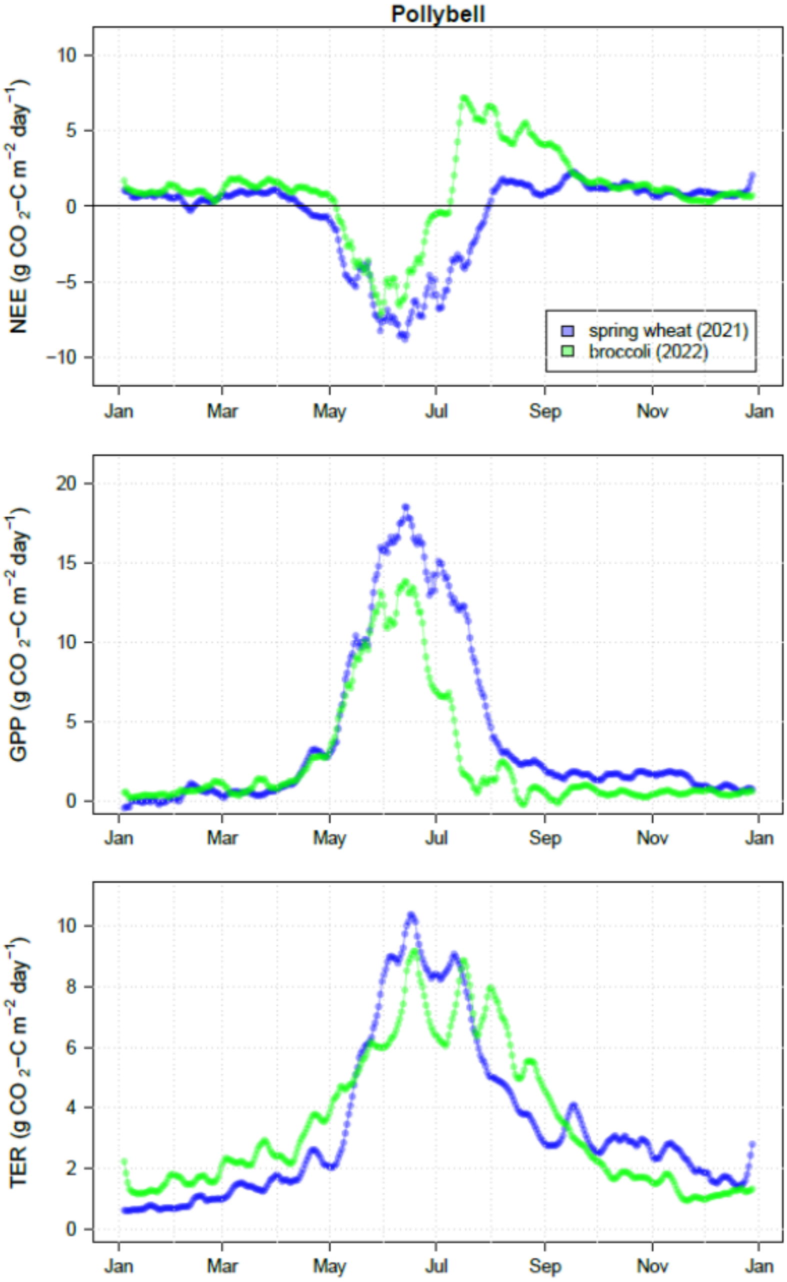



***Figure 5.18***. *Measured net ecosystem exchange (NEE), gross primary production (GPP) and total ecosystem respiration (TER) for the Pollybell cropland flux tower under a spring spring wheat (2021) and brassica crop (2022).*

The cropping pattern at Pollybell was spring wheat in 2021 (24th March to 26th August 2021) and mixed brassicas in 2022 (15th March to 30th July 2022). The brassica crop represented a mixture of broccoli and cauliflower, but cauliflower was grown within a wind sector that was excluded from the eddy covariance time series due to unfavourable measurement conditions (perturbed turbulence due to proximity of farm buildings) and was not captured by flux observations, so the data largely capture fluxes from the area under broccoli cultivation.

As an organically managed farm, no pesticides or mineral fertilisers were applied to crops at the Pollybell site. However, in the brassica year, 12.4 t $ha^{-1}$ of chicken manure was added to the field, estimated to represent a carbon input of 0.86 t C $ha^{-1}$ (Defra, 2003). No manure was added during the wheat year. Weed and pest control for the broccoli crop was achieved by growing the crop under plastic sheets which were periodically removed for mechanical weed control using a tractor. This intensive management regime resulted in more significant soil disturbance compared to single tillage for wheat and at other arable flux monitoring sites within the lowland peatland network (e.g. no-till regime at Sawtry). It is possible that the mechanically intensive organic management practices modified soil thermal and hydrological regimes, which in turn modified rates of gas production and consumption. The presence of highly reflective plastic sheeting will have altered the surface energy balance (e.g. albedo, partitioning between components of net radiation), as well as the turbulent exchange of mass and energy (partitioning between latent, sensible and soil heating) between the soil and plant system and the atmosphere. In the absence of paired measurements, it was not possible to quantify the impact of the organic management practices on the system, but the physical impacts on hydrology (e.g. infiltration, interception, runoff, etc.) and land surface-atmosphere interactions would benefit from further fundamental research.

Within-year variations in $CO_2$ fluxes at Pollybell differed considerably between years (Figure 5.18), reflecting differences between the wheat and broccoli crops. Seasonal patterns of $CO_2$ exchange for both crops were characterised by a period of net uptake during the growing season (Figure 5.18) and net losses at other times. Wheat had a longer period of net $CO_2$ uptake than broccoli, that was characterised by high gross fluxes and net $CO_2$ uptake (negative NEE). By contrast, broccoli had a shorter period of net $CO_2$ uptake and lower daily rates of GEP. $CO_2$ released via TER remained high after broccoli was harvested, resulting in a large net $CO_2$ emission to the atmosphere from harvest (late July) through to mid-September. This period of large net $CO_2$ emission could be explained by the higher rates of mechanical peat disturbance under organic vegetable production, deposition of the large amounts of labile crop residue during harvest operations, and/or elevated temperatures experienced throughout the 2022 growing season. Smaller differences in gross and net $CO_2$ fluxes outside of the growing season reflected meteorological differences, as well as the growth of 'volunteer' plants that colonised the site after the wheat harvest.

On an annual basis, TER was similar between years, but GPP was 37% lower in the broccoli year (Table 5.4). With the wheat crop present, the field was an *in situ* $CO_2$ sink in 2021 (-7.8 tonnes $CO_2$ $ha^{-1}$ $year^{-1}$) whereas it was a strong *in situ* net source in 2022 (13.4 tonnes $CO_2$ $ha^{-1}$ $year^{-1}$). However, after accounting for net C imports of manure in the broccoli year, and for the much higher harvest offtake of wheat versus broccoli (5.1 vs 1.4 t C $ha^{-1}$) the overall

NEPs for the two years converged, such that the field was an overall carbon source of 3.0 t C $ha^{-1}$ $yr^{-1}$ (11.1 t $CO_2$ $ha^{-1}$ $yr^{-1}$) in 2021, and 4.1 t C $ha^{-1}$ $yr^{-1}$ (15.2 t $CO_2$ $ha^{-1}$ $yr^{-1}$) in 2022. Higher $CO_2$ emissions in 2022 could again be the result of greater soil disturbance, soil heating under plastic sheeting, or the effects of the summer heatwave.

### Grasslands, East Anglia

Previously unreported flux tower measurements are now available for two sites in the East Anglian Fens. The Corney's (Great Fen) flux tower was initiated under the SEFLOS project on 26th April 2017, and operated until decommissioning on 14th March 2021. The site is an area of ex-arable land that was taken out of production with the goal of restoring it to wetland nature reserve. Water levels had not been raised at the time measurements commenced (a brief rewetting trial was conducted but was not successful). The site was managed as extensive grassland for the duration of flux measurements. Flux tower data from this location were not reported previously due to differences in how the site was managed to the north (hay cutting and removal) and south (low density cattle grazing, no hay removal) of the flux tower, and the challenges this presented for C budgeting. Here, eddy covariance measurements have now been restricted to the fields to the south of the flux tower, ensuring that measurements are representative of an extensively grazed grassland on thick peat. The peat is the thickest of any of the flux tower sites surveyed to date (Figure 5.13), with a total depth of 183 cm. The entire profile is humified peat, however the lower part of the profile (below 55 cm) is extremely acid, as a result of the oxidation of sulphide to sulphate (so called acid sulphate soils) which restricts rooting depths and likely also restricts microbial activity.

The Woodwalton grassland site was instrumented during summer 2020 with funding from Natural England. The site is approximately 1.3 km to the southeast of the Corney's flux tower site, separated by a high-level watercourse and an area of thick peat that is currently managed for crop production, but will soon become part of the Great Fen project area. Flux measurements commenced during heatwave conditions on 12th August 2020 and will continue until the equipment is relocated for a paludiculture project in April 2023. The site is under broadly similar management to the Corney's site, representing a deep drained and extensively grazed grassland on thick peat. Although broadly similar to Corney's, the peat only extends to 116 cm depth, and the upper 45 cm appears more modified by agricultural activity, comprising a 30 cm layer of peaty loam over a 17 cm layer of humose clay (Figure 5.13). The peat at depth is acidic, but to a lesser extent than at Corney's.

It was not possible to access grazing records for either of the East Anglian grassland sites at the time of writing, nor was it possible to estimate the net export of C associated with the live weight gain of cows. It was assumed that no hay was removed from the southern part of the Corney's site, whereas the lateral flux of C in hay was measured using destructive sampling for Woodwalton. As such, the results that follow may not reflect the full C balance of these managed grasslands. Both grassland sites showed broadly similar seasonal patterns in gross (GPP, TER) and net $CO_2$ fluxes across years (Figure 5.20). As expected from the typical pattern of grassland phenology, both grasslands had a period of high growth, commencing as temperatures rose throughout spring into early summer (late June to July), before declining as the grassland vegetation senesced throughout late summer. The evolution of TER showed a broadly similar seasonal pattern to GPP, with the increase lagging

GPP during early spring, before declining at a slower rate from late summer into autumn. The relative changes in gross $CO_2$ fluxes resulted in a period of high net $CO_2$ uptake during spring and early summer, followed by a relatively long period of net $CO_2$ emission that continued until the return of warmer conditions the following spring.

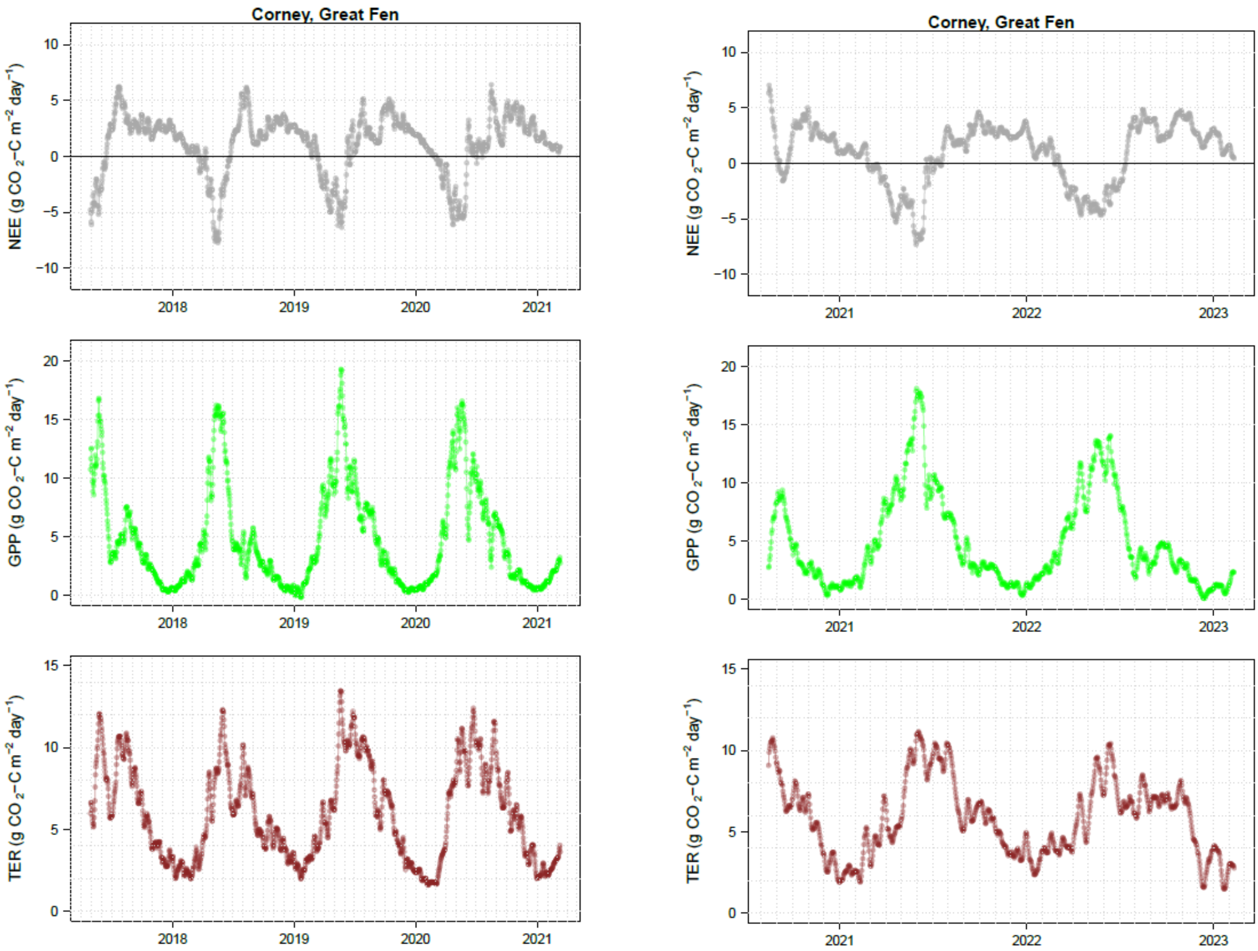



***Figure 5.20**. Daily carbon flux terms monitored at lowland peatland flux tower sites managed as extensive grassland. Data are daily sums of net ecosystem $CO_2$ exchange (NEE), gross primary production (GPP) and total ecosystem respiration (TER). To enhance the visualisation of seasonal trends, daily flux terms are presented using five-point running means. Note the different periods of record at each of the sites.*

Annual $CO_2$ emissions from the two grasslands were remarkably stable across years and sites (Figure 5.16), despite differences in gross $CO_2$ fluxes and biomass export at Woodwalton (Table 5.4). Mean GPP (± SD), TER and NEP for Corney's Farm over the 2018 to 2021 period were 18.5 ± 2.0, 21.9 ± 1.9 and 3.4 ± 0.3 t C $ha^{-1}$ $yr^{-1}$ respectively, with the NEP value equating to a $CO_2$ emission of 12.4 ± 1.0 t $CO_2$ $ha^{-1}$ $yr^{-1}$. Woodwalton had slightly higher average GPP (19.6 ± 1.4 t C $ha^{-1}$ $yr^{-1}$) and slightly lower TER (21.6 ± 0.4 t C $ha^{-1}$ $yr^{-1}$) but had a remarkably similar NEP to that at Corney's (3.4 ± 0.01 t C $ha^{-1}$ $yr^{-1}$) once C offtake as hay was accounted for. This stability in annual NEP was particularly noteworthy at Woodwalton, where GEP and hay yield were both reduced (by 9 % and 33 %, respectively) relative to 2021 during the drought conditions of summer 2022.

## Field water table manipulation experiment 1 – Stretham

### *Introduction*

A key aim of the project was to evaluate the impacts of raising water levels under agricultural land as a mitigation measure to reduce peat-derived GHG emissions, whilst permitting continued agricultural use of the land. At the outset of the project, we were not able to implement changes in water management at a whole-field scale, so constructed the Skyline 2D plot-scale experiment to test the hypothesis that raising water levels would reduce emissions, which was based on a previous analysis of flux tower data (Evans et al., 2021), which predicted that $CO_2$ emissions would be reduced by 3 t $CO_2$e year$^{-1}$ for every 10 cm reduction in the depth of drained peat based on a global dataset, with a higher estimate of 5 t $CO_2$e year$^{-1}$ per 10 cm reduction based on UK data alone (see also Section 5.6). As described above, it proved impossible to maintain controlled water levels within individual plots, so the experiment focused on surface irrigation as an alternative mitigation measure. However, over the course of the project, opportunities became available – as a result of farmer support and the availability of additional flux towers – to manipulate water levels at the field scale. We were able to examine the effects of water table change at Stretham (for a full year) and at Pollybell (for a partial year, see below).

### *$CO_2$ fluxes and hydrology*

At Stretham, a wheat field close to our existing flux tower site was managed with higher water tables throughout 2022. The 'HWT' field is approximately 1 km from the existing flux tower, at which 'business as usual' ('BAU'), water management practices were followed (Figure 5.21) and thus experiences near-identical climatic conditions. It has a broadly similar peat depth and had been planted with the same variety of winter wheat ('*Illustrious*') the previous autumn. UKCEH deployed a new Defra-funded flux tower and associated meteorological and soil sensors in the HWT field, with paired measurements of HWT versus BAU $CO_2$ fluxes taken between 14$^{th}$ May 2022 and 31$^{st}$ January 2023 (measurements are ongoing).

Results from the HWT vs BAU trial at Stretham are presented as time series in Figure 5.22, 5.23 and as summary statistics in Table 5.5. Despite extreme drought conditions during the 2022 growing season, and an incident where water levels were abruptly lowered by the activities of a neighbouring farm in early July, it was possible to maintain water levels and peat moisture content at higher levels in the HWT site relative to BAU for most of the year. Mean water levels were -1.2 m and -0.6 m below the surface for BAU and HWT, respectively. The water level was almost entirely within the underlying clay layer at the BAU site, but was above the peat-clay interface at the HWT site for much of the same period. Large differences in soil moisture were also maintained within the peat layer, with mean (±SD) values of 0.1 ± 0.09 m$^3$ m$^{-3}$ and 0.3 ± 0.09 m$^3$ m$^{-3}$ at the BAU and HWT flux towers, respectively. ET followed broadly similar seasonal trends at both sites, with a slightly higher total of 399 mm from the BAU field compared to 359 mm from the HWT field over the same 263 day period.

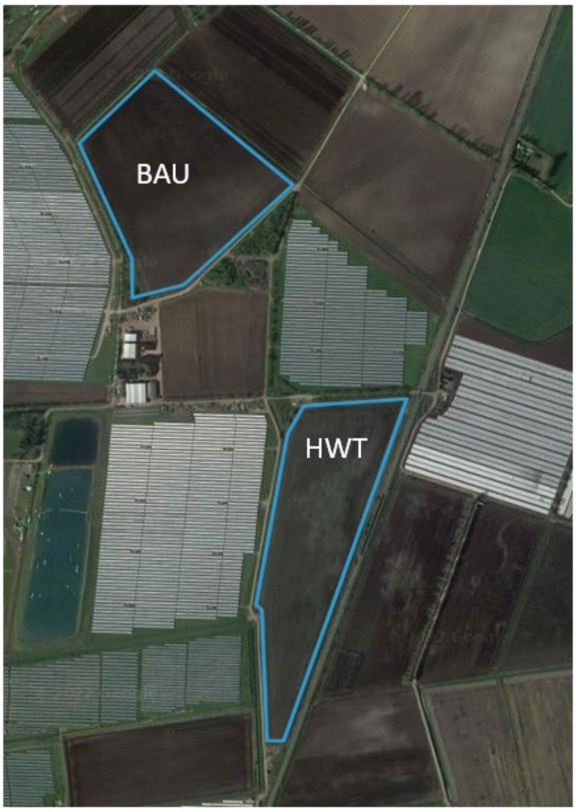



***Figure 5.21.*** *Locations of the 'business as usual' (BAU) and 'high water table' flux tower fields at Stretham.*

Daily *in situ* $CO_2$-C fluxes showed broadly similar trends, as expected for the same crop variety under near-identical climatic conditions. Higher rates of both GPP and TER, and subsequently higher rates of daily net C uptake, were measured for the HWT treatment during the growth phase of the crop, and again over the late autumn and early winter period. Overall, the HWT field had higher time integrated GPP and TER than the BAU field (Table 5.5), but functioned as a smaller *in situ* net source for atmospheric $CO_2$. Despite the higher rates of photosynthesis at the HWT site compared to BAU, measured crop yield was lower form the HWT treatment; this is discussed below. Whilst higher rates of gross photosynthesis and lower yield appear counterintuitive, this result may be explained by differences in C allocation patterns under the different treatments. Both treatments resulted in a net $CO_2$ emission to the atmosphere (positive NEP) after accounting for harvested exports. Total $CO_2$ emissions from the HWT treatment over the 263 day trial period (21 t $CO_2$ ha$^{-1}$ [263 day]$^{-1}$) were approximately 7 t $CO_2$ ha$^{-1}$ lower compared to BAU (28.2 t $CO_2$ ha$^{-1}$ [263 day]$^{-1}$). Given that average water levels were about 10-15 cm above the base of the peat layer in the HWT field over the measurement period, this emissions reduction appears quite consistent with the estimate of 5 t $CO_2$ emissions reduction per 10 cm reduction in peat drainage depth previously derived from UK flux data (Evans et al., 2021).

*Table 5.5. Summary $CO_2$ fluxes and hydrometeorological data for the Stretham high water table trial.*

| VARIABLE | UNIT | BAU | HWT |
|---|---|---|---|
| GPP | t C $ha^{-1}$ [263 days]$^{-1}$ | 11.1 | 13.1 |
| TER | t C $ha^{-1}$ [263 days]$^{-1}$ | 12.3 | 13.7 |
| NEE | t C $ha^{-1}$ [263 days]$^{-1}$ | 1.1 | 0.7 |
| YIELD | t C $ha^{-1}$ | 6.6 | 5.1 |
| NEP | t C $ha^{-1}$ [263 days]$^{-1}$ | 7.7 | 5.7 |
| Net $CO_2$ emission | t $CO_2$ $ha^{-1}$ [263 days]$^{-1}$ | 28.2 | 21.0 |
| P | mm [263 days]$^{-1}$ | 320 | 320 |
| ET | mm [263 days]$^{-1}$ | 399 | 359 |
| Mean WL | m below surface | -1.2 | -0.6 |
| Mean VWC | $m^3$ $m^{-3}$ | 0.1 | 0.3 |

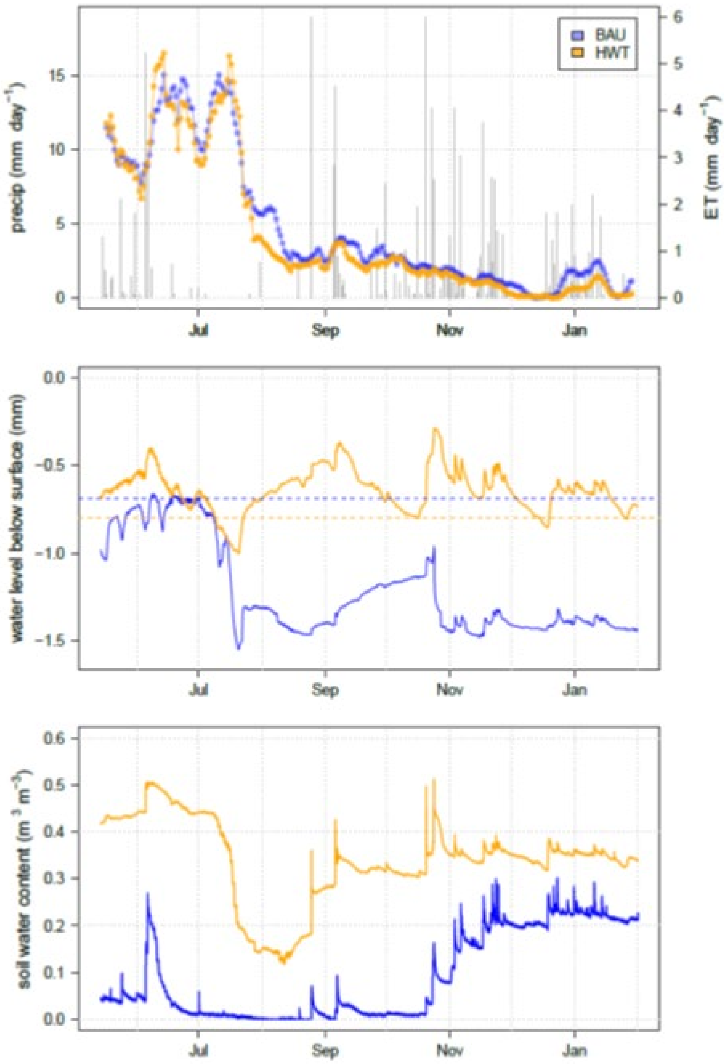



*Figure 5.22. Precipitation and evapotranspiration (top), water table (middle) and soil moisture content (bottom) for the Stretham BAU and HWT trial fields. Horizontal dashed lines in the middle plot show estimated average peat depths for the two fields.*

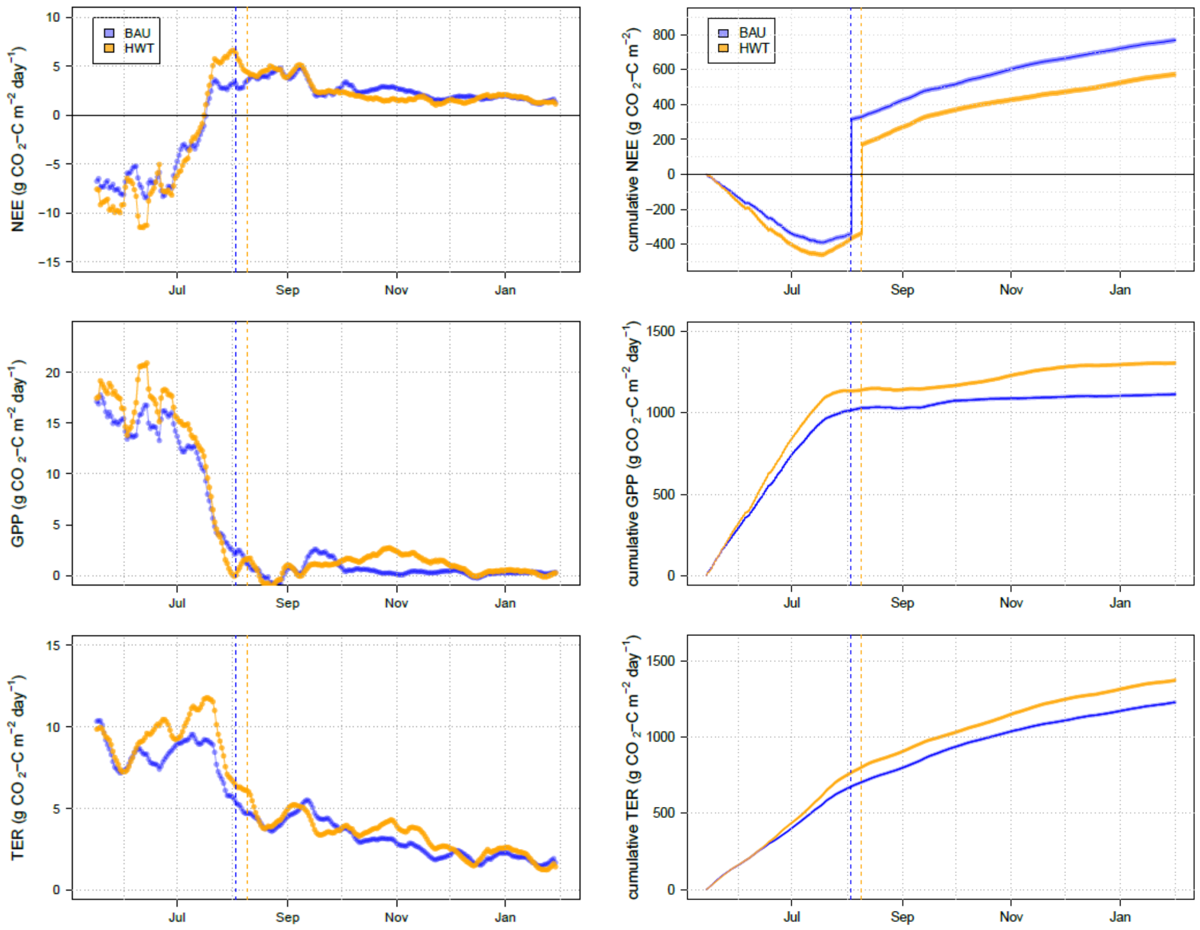


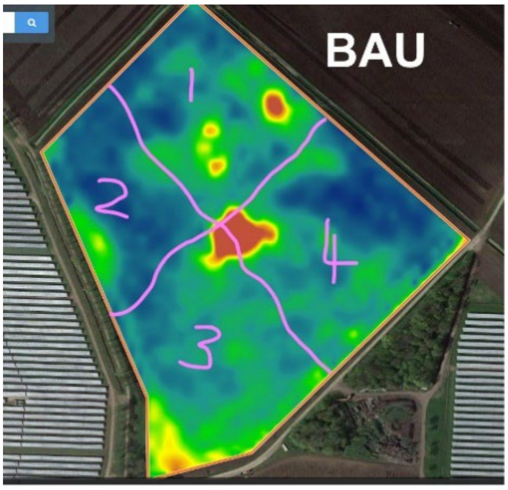

*Figure 5.23. Daily (left) and cumulative (right) $CO_2$ flux components for the BAU and HWT fields, Stretham high water trial.*

### *Wheat crop performance*

A detailed assessment of wheat crop performance was undertaken by ADAS for both the BAU and high water level (HWT) fields at Stretham. Satellite images of both field trial sites were obtained with NDVI (normalised vegetation difference index) via Data Farming https://www.datafarming.com.au/ . Based on these images (Figure 5.24), each field was divided into four quadrants for assessment to allow for potential spatial variations.

Plant health assessment were performed at growth stage 75, which is defined as GS75: Medium milk (grain content milky, grains reached final size) in the AHDB Guide to Growth stages of cereals (https://ahdb.org.uk/knowledge-library/the-growth-stages-of-cereals). The growth stage was confirmed by the grower and assessments done by the ADAS field research team on 30th June 2022. Twenty-five tillers were sampled from across a diagonal transect of each quadrant, radiating from the centre to the outside edge of the field. These tillers were assessed for all foliar, and root, stem and ear diseases, as well as Green Leaf

Area. Green leaf area was recorded as a percentage for each leaf layer until such stage that the leaf layer is completely dead. Symptoms of any other diseases were recorded if present.



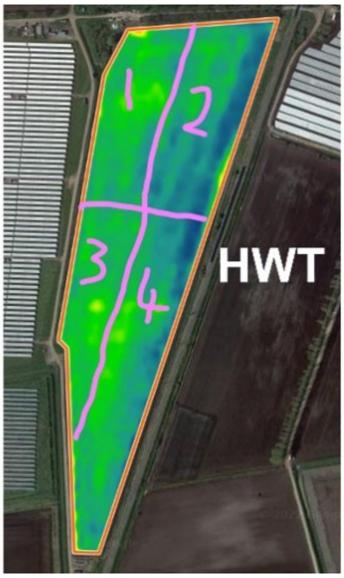




***Figure 5.24.*** *NDVI index overlays of satellite images of the BAU and HWT field sites, with assessment quadrants (1-4) marked.*

Formal weed assessments were not made but any marked visible differences in weeds (number and type) in the two fields on a quadrant basis was noted. Any pests present were noted but not formally assessed or quantified. An estimate of the percentage area of each field quadrant affected by lodging and whiteheads was made.

Immediately prior to harvest the fields were assessed using the same quadrants as in the GS75 assessment. Twenty-five tillers were taken from diagonal transects in each quadrant in each field. Tillers were assessed for the following:

**Crop height** – measured from base of stem at soil level to top of highest ear.

**Lodging** – an estimate of the percentage of stems displaced from their vertical position as a result of stem buckling or root displacement.

**Grain heads/fertile ears per m$^2$** – quadrats at 5 points in each quadrant were used to sample the area to count the number of grain heads in the area.

**Grains per head** – Each quadrant was sampled as a single plot. Samples were threshed and grains counted in a subsample using a grain counting machine. The grain number per ear was calculated as follows:

$$\frac{Number\ of\ grains\ in\ subsample\ \ \text{x}\ \ weight\ of\ whole\ sample}{Number\ of\ ears\ in\ whole\ sample\ \ \text{x}\ \ weight\ of\ subsample}$$

Harvest and grain quality was assessed on a 5kg sample of harvested grain obtained from the grower, and samples were tested to measure the following quality parameters:

**Moisture content and specific weight** using Dickey-John GAC2000 grain analysis computer. Specific weight is a standard of quality in grain trading and intervention, representing the weight of a given volume of grain, expressed in kg/hL. Specific weight measures grain plumpness which is affected by cultivar, growing conditions and husbandry. The GAC2000 is accurate to ± 0.5kg/hL of the readings produced by the UK twenty litre standard instrument. The following conversion was applied to the measured specific weight:

$$Specific\ Weight\ (final) = Specific\ Weight\ (measured) + (0.35 \times (\%MC - 15))$$

**Grain quality** – Hagberg falling number and Protein content were determine by sending samples to NRM for lab testing (methods S1001 and S1018 respectively). The Hagberg falling number is a measure of enzyme activity in the grain; a lower number indicates higher enzyme activity, which occurs as the grain starts to germinate and is associated with lower dough quality.

Post-harvesting, six root core samples were taken from each quadrant of both BAU and HWT fields. Cores were taken to a depth of 100cm using a hydrocore, with a 2.6 cm diameter borer/auger. Each core was divided into 20cm horizons (0-20, 20-40, 44-60, 60-80, 80-100cm) and the same horizons from each of the 6 replicate cores were pooled. Samples were frozen until analysis.

The two deepest horizon samples (60-80cm and 80-100cm) for each quadrant and field were thawed and washed using a Delta-T root washing system with 550 micron filters to separate soil and organic material from roots. Each horizon and quadrant was washed separately. Crop debris and non-root material was removed from samples. Clean roots were placed into containers with water for scanning using WinRHIZO root analysis package software (Regent Instruments Ltd. Quebec City, Canada) and a flatbed scanner. Root measurements were (total length, mean diameter and surface area) calculated. After scanning roots were placed into tins and weights recorded. Roots were dried in an oven at 80°C for 48 hours or until no further weight loss. Dry weights were then recorded. Paired t-tests were conducted between root measures to compare the distributions of size roots between treatments.

Full results of the ADAS crop assessment are provided in the accompanying report by Eyre et al. (2023), while key results are provided here. At GS75, Mean green leaf area at a similar level between BAU and HWT for leaves 1 to 5 but there was a trend towards reduced green leaf area in the high water table samples (Figure 5.25).

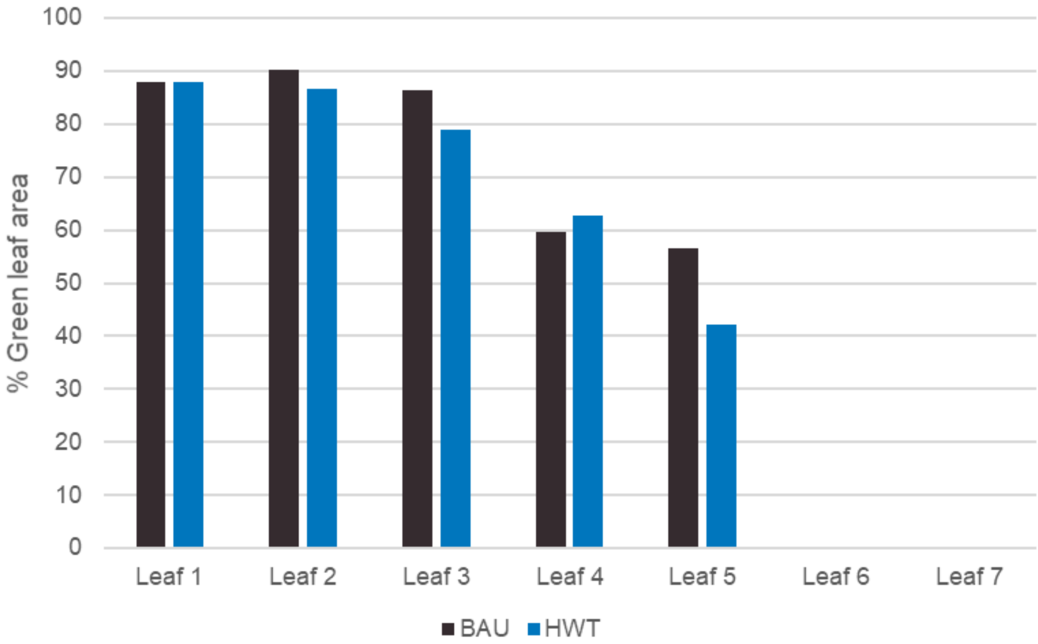



***Figure 5.25.*** *Mean green leaf area of 25 tillers sampled from 4 quadrants from winter wheat business as usual (BAU) and high water table (HWT) fields at growth stage 75.*

Between 70-90% of leaves sampled from the HWT field had some *Septoria* disease present which was consistent between quadrants. The BAU field was less consistent between quadrants and incidence ranged from under 10 to almost 70% (Figure 5.26). The severity of disease where present was generally low, however, with less than 10% leaf coverage for all samples. The severity was higher in the HWT samples compared to the BAU field.

Downy mildew incidence was much lower than that of *Septoria*. Downy mildew was only present in 2 out of 4 quadrants in BAU, but present in all four quadrants of HWT. The HWT incidence was below 10% for 3 quadrants but in quadrant 3 was 0.36. Severity was under 10% for all quadrants apart from Q1 in HWT which was 22.50% (Figure 5.27).

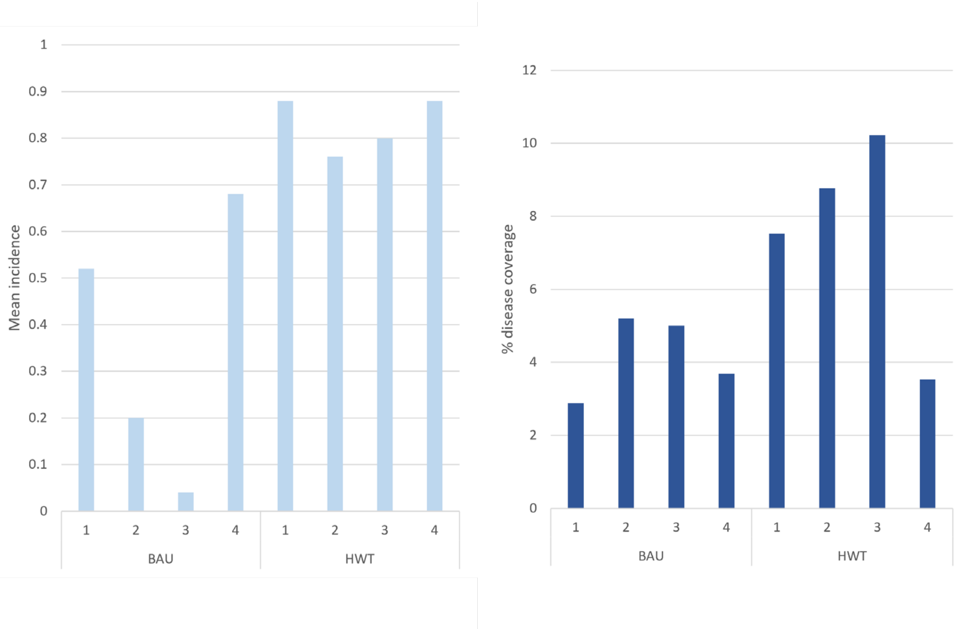



***Figure 5.26.*** *Septoria* incidence (left) and severity (right) at growth stage 75 assessment of winter wheat in business as usual (BAU) and high water table (HWT) fields.

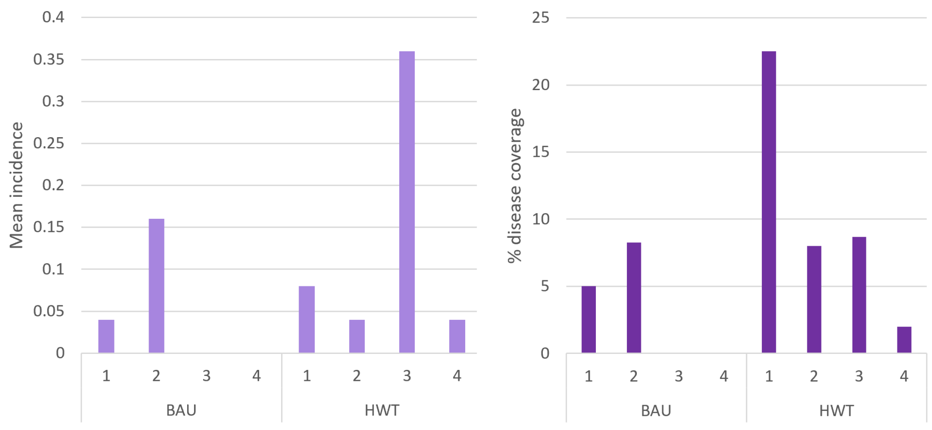



***Figure 5.27.*** *Downy mildew incidence (left) and severity (right) at growth stage 75 assessment of winter wheat in business as usual (BAU) and high water table (HWT) fields.*

The pre-harvest assessment showed very little difference between BAU and HWT fields and all metrics were with a normal range that might be expected for winter wheat crops. The major difference between the two treatments was in the whole grain weight which was greater in the BAU field (Figure 5.28).

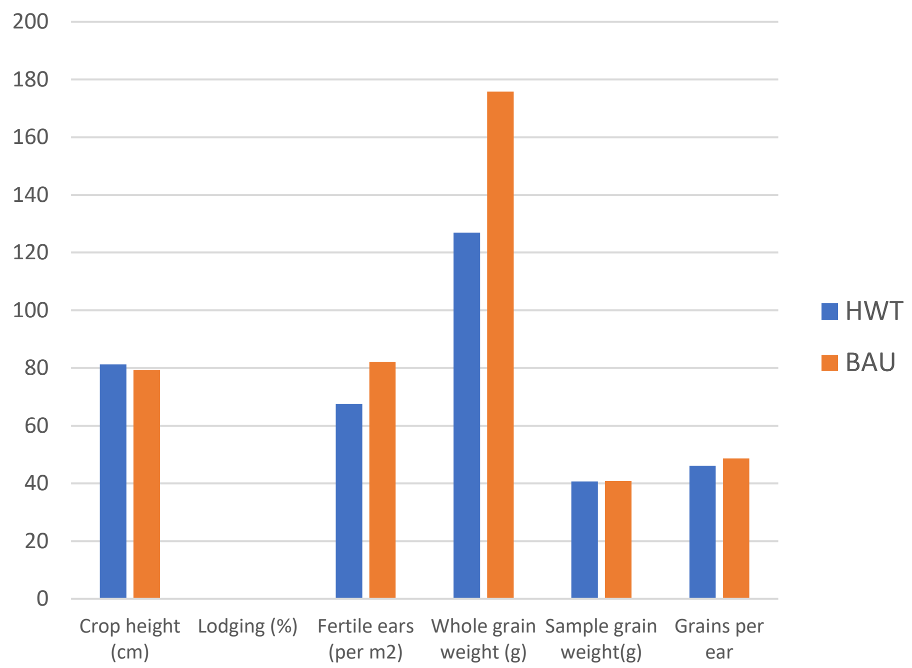



***Figure 5.28.*** *Pre-harvest assessment of winter wheat fields for business as usual (BAU) and high water table (HWT) for crop height, lodging, fertile ears, whole grain weight and sample grain weight and grains per ear.*

Yields from the BAU and HWT fields were 11.50 and 9.95 t ha$^{-1}$ respectively (Table 5.6). Grain moisture and specific weight measures were very similar between treatments and both within normal range. Protein content was greater in the HWT than BAU, but Hagberg falling number was lower, indicating slightly lower (but still acceptable) quality for baking.

***Table 5.6***. *Yield and grain quality measurements for HWT and BAU fields.*

| | Yield t/ha | Hagberg Falling Number (s) | Protein % | Grain moisture % | Specific weight kg/hl |
|---|---|---|---|---|---|
| HWT | 9.95 | 267 | 12.27 | 13.79 | 82.23 |
| BAU | 11.50 | 360 | 10.9 | 13.73 | 81.7 |

Mean root length density (RLD) was higher at 60-80 cm than at 80-100 cm in both fields, but much higher in the BAU field than the HWT field in both horizons (Figure 5.29). Average root dry weight and specific root length followed similar patterns to the RLD in both treatments and horizons, while average diameter of roots was the same across all horizons and diameters (Table 5.7). More than 97% of roots were < 0.5 cm in length.

Paired t-tests comparing the 60-80cm horizon measures between BAU and HWT treatments did not find any significant differences between root metrics, despite the apparent differences in means.

***Table 5.7.*** *Root analysis results for winter wheat post-harvest root core sampling*

| Treatment | Horizon depth (cm) | RLD (cm/cm3) | Root DW (mg/cm3) | Mean Diam. (mm) | Specific root length (m/g) | 0 < L <= 0.5 | 0.5 < L <= 1.0 | 2.0 L <= 1.5 | 1.5 < L <= 2.0 |
|---|---|---|---|---|---|---|---|---|---|
| BAU | 60-80 | 1.073 | 0.037 | 0.20 | 301 | 668 | 14.79 | 0.10 | 0.002 |
| | 80-100 | 0.389 | 0.018 | 0.20 | 213 | 245 | 3.06 | 0.02 | 0.000 |
| HWT | 60-80 | 0.607 | 0.026 | 0.20 | 240 | 378 | 8.28 | 0.11 | 0.000 |
| | 80-100 | 0.199 | 0.018 | 0.19 | 106 | 125 | 2.07 | 0.07 | 0.000 |

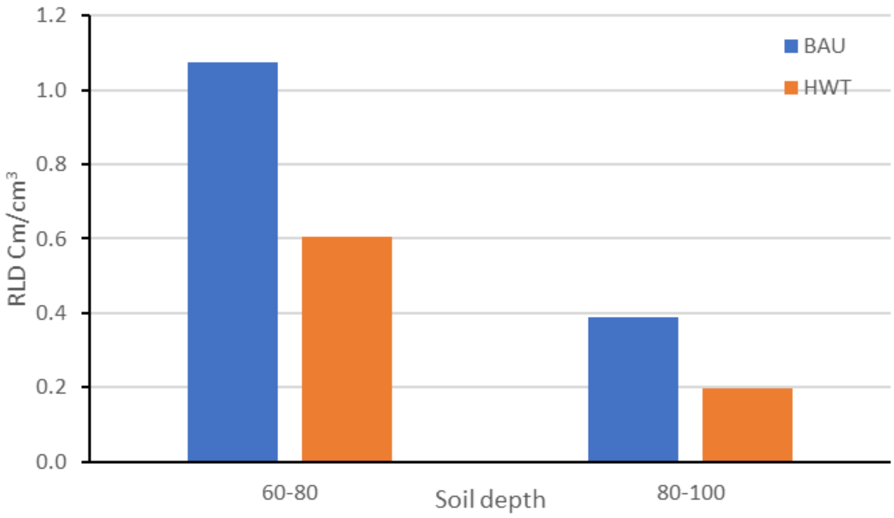



***Figure 5.29.*** *Mean root length density (cm/cm$^3$) of roots sampled from winter wheat fields for business as usual (BAU) and high water table (HWT) for 60-80 and 80-100cm horizons of soil cores.*

The development of the plants in both BAU and HWT fields was generally comparable but with a trend towards reduced green leaf area for HWT indicating that the higher water level may have had some impact on development. Disease levels were generally low overall for both *Septoria* and downy mildew, likely reflecting the hot and dry conditions in 2022 which were not conducive to high disease occurrence. However, the trend towards more *Septoria* symptoms, albeit at a relatively low level, in the HWT field could indicate that increasing soil moisture did favour disease development to some extent.

The gross yields for the two treatments were within acceptable levels, but the yield for the HWT field was 13% lower than that of the BAU field. Despite the proximity and similar management (apart from water levels) for the two fields, we cannot state with certainty that the lower yield in the HWT was due to the higher water levels; indeed given the severity of the 2022 drought it might have been expected that yields would be higher in this field. It is possible that other factors with the management of the HWT field may have confounded some of the results, notably there was an abrupt drop in water level shortly before harvest as a result of water removal from the ditch by an external party, at the same time as restrictions on water abstractions from the river network by the Environment Agency. This dried the field down immediately and likely halted crop development. This could have had a yield impact, but the crops were also at full development with ears already formed so it is unlikely that all of the yield reduction in the HWT fields was solely down to this incident.

The quality of the grain was generally good and within normal ranges. The lower Hagberg falling number in the HWT field could be caused by pre-harvest sprouting caused by an uneven crop or lodging. Although there was no lodging recorded, the issues with water management could have impacted on germination. Hagberg falling numbers can be increased via crop nutrition and it may also be that these plants were compromised in their nutrient uptake by the higher water table giving a variable soil nitrogen supply.

As would be expected, wheat in the BAU field had a higher average root length density, dry weight and specific root length than in the HWT field, at both depths sampled. This is consistent with greater exploration by the root networks, going deeper and wider to seek water. Greater allocation of energy to roots to access water might be expected to reduce yields, but this was evidently not the case. It is possible that the abrupt drop in water levels in the HWT field pre-harvest meant that the crop was no longer able to access sufficient water given the shallower and less extensive root network, and that this could have led to the lower recorded yields. In light of both the extreme weather conditions in 2022 and this potentially detrimental impact of a sudden and unanticipated water level drop, we conclude that the lower yields in the HWT trial may not have been due to the higher water level management of the field in the earlier part of the growing season, when the wheat appeared to be growing more or less as well as that in the BAU field (Figure 5.25). Repeating the high water trial at this site, and extending a similar approach to other sites, would provide more robust data on the interaction between raised water levels and crop yields, and better understanding of any trade-offs between climate change mitigation and food production.

### Field water table manipulation experiment 2 – Pollybell

A second opportunity to analyse the impact of water level variation on $CO_2$ emissions using eddy covariance was made possible by work conducted under the GGR-Peat project at another area of Pollybell Farm. Here, three new flux stations were established at an area of grassland that had previously been used to trial the impacts of high water level management on wet grassland soils and bird populations. The GGR-Peat study site is divided into two hydrologically distinct areas to the west and east of a central drainage ditch. Each hydrological unit is further subdivided (north and south) into two fields. The entire area was previously under organic arable management but converted to wet grassland with occasional mob-grazing by sheep. In November 2021, one flux tower (GGR-P1) was installed at an area of winter wet grassland to the west of the site (Figure 5.30), where water levels are controlled by the central drain to the east and 'Mother Drain' to the west.

Two other flux systems (GGR-P2, GGR-P3) were established on drier fields to the east of the central drain and are also influenced by a lower farm boundary drain to the east. All four fields are below the level of the nearby River Idle as a result of long-term peat drainage.


***Figure 5.30****. Location of grassland flux towers at Pollybell (Google Earth image taken during July 2023, shortly after field P2 was mown, and during mowing of field P3. Field P1 had visible poor vegetation growth and was not mown.*

The management of the Pollybell site resulted in two distinct hydrological periods. Between 28th November 2021 and 25th March 2022, the P1 field to the west of the site was kept wet by maintaining high levels in the central ditch, resulting in a mean (± SD) water level of -0.15 ± 0.03 m, and a soil moisture content of 0.73 ± 0.04 $m^3$ $m^{-3}$ (Figure 5.31). During this period the GGR-P2 and P3 sites had deeper water tables (0.50 ± 0.07 m and -0.41 ± 0.08 m respectively) and lower soil moisture levels 0.41 ± 0.02 $m^3$ $m^{-3}$ and 0.50 ± 0.08 $m^3$ $m^{-3}$ respectively) due to the influence of the farm boundary ditch. This pattern was reversed on 25th March 2022, when the central ditch level was lowered to facilitate access for planting of the GGR plot trials in the fourth (north-western) field. This resulted in rapid drainage of the P1 field to a new base level of around 90 cm at the end of April, and a more gradual decline in soil moisture. Although water levels also fell in the P2 and P3 fields during this period, as evapotranspiration increasingly exceeded low levels of rainfall, the decline was more gradual. As a result, water levels in P1 during April were around 20-30 cm lower than those in P2 and P3. By June the water level in P1 had stabilised, while levels in P2 and P3 continued to drop, so the pattern reversed again.

Soil moisture levels showed a slightly different pattern, showing a more gradual decrease at P1 compared to water levels, but this decrease was sustained so that by the end of June moisture levels in P1 remained lower than those in P2 and P3.

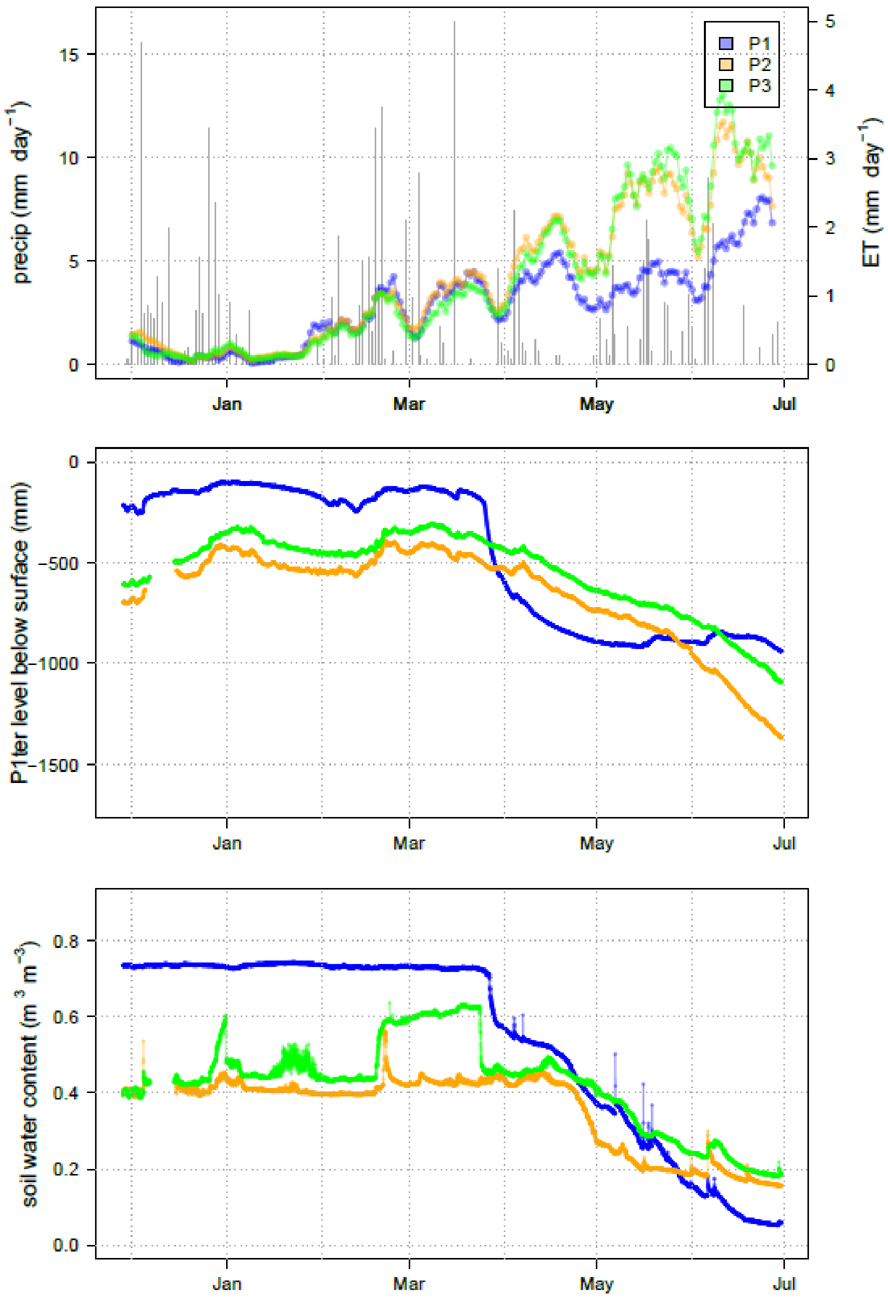



***Figure 5.31***. *Precipitation and evapotranspiration (top), water table (middle) and soil moisture content (bottom) for the Pollybell GGR flux towers, December 2021 – June 2022.*

The contrasting hydrological behaviour of the adjacent fields at Pollybell clearly affected $CO_2$ fluxes. In the first period, from December to March, GPP and TER were both lowest in the wet P1 field, and highest in the driest field (P2), resulting in marginally lower winter NEE at P1 (Figure 5.32). Over the full pre-drainage period, time-integrated NEE was 93, 91, 134 g C $m^{-2}$ for P1, P2 and P3 respectively. The onset of spring photosynthesis at each location reflected the gradients in water levels and soil water, with GPP rising first at P2, and then at P3, while it remained low at P1 until March.

Drainage of P1 at the end of March had a dramatic impact; although GPP initially rose through April, it dropped back to very low levels (< 5 g C $m^{-2}$ $day^{-1}$) in May, when the field was observed to have very limited weedy vegetation cover. Meanwhile, fields P2 and P3 had a healthier grass crop and GPP was peaking at over 15 g C $m^{-2}$ $day^{-1}$. ET was also continuously lower at P1 compared to other locations following drainage, as expected due to lower levels of plant transpiration with reduced GPP. Almost immediately after drainage, TER at P1 rose from levels well below those of P2 and P3 in early March to higher levels in April.

Although TER at P2 and P3 subsequently became higher again, this clearly coincided with high rates of photosynthesis and therefore included a large component of autotrophic (plant) photosynthesis. In contrast, with minimal photosynthesis at P1, much of the observed respiration was clearly heterotrophic, resulting from peat decomposition. As a result of the imbalance between high TER and low GPP, NEE for P1 rose immediately following drainage and remained strongly positive throughout the remainder of the measurement period. In contrast, NEE at P2 and P3 was continuously negative during the same period as a result of high rates of grass growth.

Overall, P1 functioned as a net source of 382 g C $m^{-2}$ during the post drainage period when water levels were lower than at the other two sites, while P2 and P3 functioned as net sinks of -335 and -396 g C $m^{-2}$ over the same time interval.

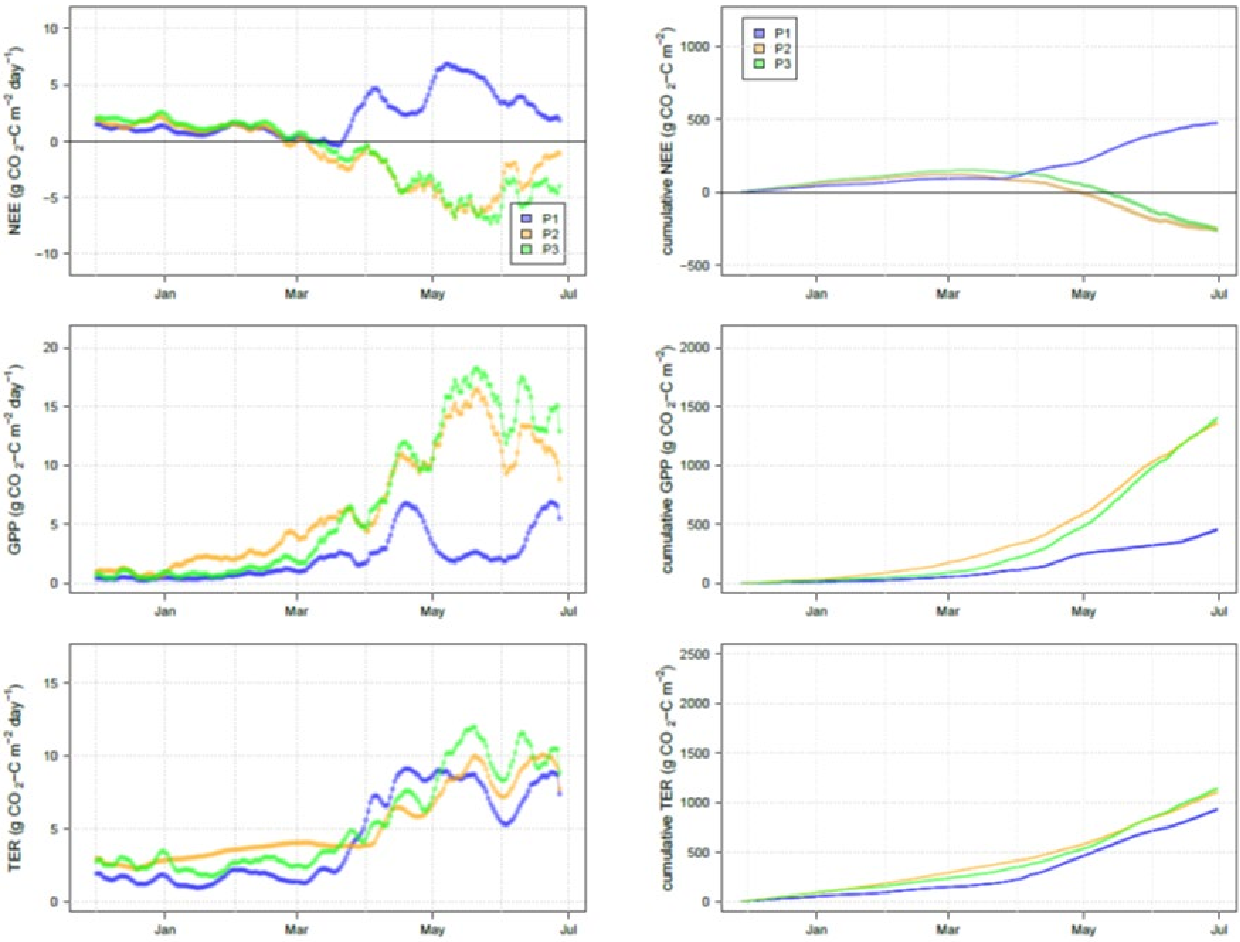



*Figure 5.32. Daily (left) and cumulative (right) $CO_2$ flux components for the three GGR flux towers at Pollybell.*

It is important to note that the marked differences in NEE between the three Pollybell GGR sites do not translate directly into differences in the long-term carbon balance; given the deep water table drawdown at P2 and P3, apparent $CO_2$ drawdown in the study period at these sites is undoubtedly the result of short-term $CO_2$ uptake into the biomass, and subsequent data (not shown) show that both sites became carbon sources later in the year. Conversely, the rapid and sustained increase in NEE at P1 after drainage appears to be partly a result of very poor plant growth in the field, probably as a direct result of the abrupt transition from very wet winter conditions to extremely dry conditions from late March onwards; as at Stretham, this likely exposed shallow-rooted plants to rapid drying. However, the accompanying switch from very low TER values at P1 in February - March (below those of P2 and P3) to very high values in April (above those of P2 and P3) is consistent with a strong water table control on heterotrophic respiration.

These results therefore also support the conclusion that higher water tables reduce $CO_2$ losses due to peat decomposition, but also highlight the importance of maintaining these higher water levels through the growing season.

## 5.4 Spatial assessment of fluxes

### Introduction

The aim of this task was to provide data on the extent of spatial variation in $CO_2$ fluxes across a range of lowland peat sites under cropland management, in order to provide a broader context for experimental and flux tower results. We aimed to measure across a gradient of drainage depth, peat thickness and land-use, including some of the flux tower sites to provide a longer-term context. All work was undertaken in the East Anglian Fens, and following the initiation of the BEIS Wasted Peat project we decided to focus effort on a subset of the 47 sites surveyed by Burton (1995), and resurveyed during the BEIS project. As well as simplifying the process of obtaining access permissions, this also enabled us to compare results to detailed soil profile descriptions and soil analyses undertaken as part of the BEIS project, as well as basal respiration measurements made on samples from all locations. This work will be reported fully in the forthcoming final report of the BEIS project; the following focuses on the results of a field survey of in situ $CO_2$ fluxes carried out for the Defra project.

### Methods

Field $CO_2$ fluxes were measured at a total of 31 sites distributed across the Fenland region, of which 25 sites formed part of the survey of Burton (1995), and six were active flux tower sites. The Burton and Hodgson survey focused primarily on sites subject to varying degrees of peat wastage, but also included some sites that would still be classified as peat. Overall, 18 of the sites surveyed met the formal definition of wasted peat (total organic horizon depth ≤ 40 cm), 12 were classed as thin peat (> 40 to ≤ 100 cm – although around half of these had organic horizons of < 50 cm) and only one site (Rosedene) met the definition of thick peat (> 100 m). All sites were under cropland management, but were sampled while the field was bare in order to exclude $CO_2$ fluxes associated with photosynthesis and autotrophic respiration, in an attempt to measure solely heterotrophic respiration, associated with peat decomposition.

$CO_2$ efflux was measured with a manual dark chamber connected to an infra-red gas analyser (Li-COR Biosciences Inc. Lincoln, Nebraska, USA). Measurements were made once at each site, but at three measurement points in order to capture local spatial variability. All sites were surveyed during 2022. Three soil collars were inserted 10 cm deep into the soil 24 hours before the measurements to avoid short-term effects of soil disturbance. At each collar, three efflux measurements were taken and the soil temperature was measured. To standardise across sites, measured $CO_2$ fluxes were normalized to 20 °C assuming a $Q_{10}$ value of 2 to describe the temperature sensitivity of the efflux (the $Q_{10}$ describes the proportional change in a biological process in response to a 10 °C temperature increase, and a value of 2 is widely applied to describe the temperature-sensitivity of respiration).

All of the sites studied during the spatial survey of $CO_2$ fluxes were surveyed by Rodney Burton. Surface soil bulk density (BD) was measured on a single intact sample taken at 20 cm below the surface, while other measurements were taken on a bulk mix of 20 soil surface samples, in line with the method of Burton (1995). Additional samples for %SOC analysis were taken from the individual measurement collars and analysed in the same way as the bulk topsoil samples. This involved treatment with 10% HCl to remove inorganic carbon, after which the samples were washed with demineralised water until neutral and subsequently freeze-dried. The dried soil was then homogenized using a ball mill and 0.5 mg of soil was weighed into tin-cups using a microbalance (accuracy of 0.001mg). SOC was determined using a SerCon ANCA GSL elemental analyser interfaced to a SerCon Hydra 20–20 continuous flow isotope ratio mass spectrometer. Typical measurement reproducibility was determined through repeat analysis of a laboratory standard and was better than 0.1%.

Total peat thickness, including the plough layer and any underlying peat, was recorded *in situ* by Rodney G.O. Burton (note that 'peat thickness' in this context includes some wasted peat sites with peaty or humose topsoils). Water table depth was estimated during the soil survey based on the depth to water within the auger hole, or (if no water was present) based on observed colour changes in the soil, which indicate the maximum depth of aeration. In many cases the water table was below the maximum depth recorded in the soil profile, but in all cases where this occurred it was also below the base of the peat layer.

Topsoil C stock was calculated from measured %SOC and BD, multiplied by the measured depth of the plough layer. Total peat C stock was crudely estimated as a multiple of topsoil %SOC and bulk density, and total peat thickness (i.e. any peat present below the topsoil was assumed to have the same %SOC and bulk density as the topsoil). The aerated peat C stock was calculated in the same way, but with the thickness of the peat layer truncated at the water table. Soil temperature was measured at the time of chamber measurements.

Testing of soil strength with a penetrometer did not deliver consistent or interpretable results, and these data were not used. Full soil profile descriptions and results of laboratory analyses will be included in the final report to BEIS.

### Results

Relationships between some key measured soil properties are shown in Figure 5.33. As expected, there was a strong inverse relationship between %SOC and bulk density, but only a weak relationship between either of these variables and peat thickness. The use of the Burton and Hodgson (1995) sites resulted in a high proportion of relatively thin, high-density and low-%SOC sites, with the smaller number of flux tower sites helping to extend the range of observations to include thicker, higher-%SOC locations.

The water table was below the base of the peat layer in all but two sites, so for the remaining 29 sites the aerated peat C stock was the same as the total peat C stock.

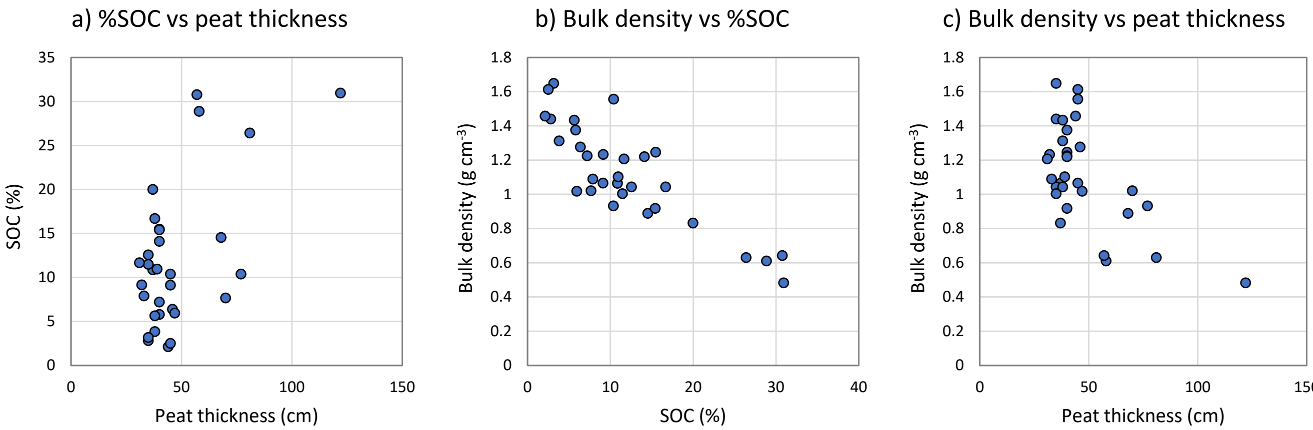



***Figure 5.33**. Relationships between key soil chemical and physical properties for the 31 extensive chamber survey sites.*

Temperature-normalized measurements of $CO_2$ efflux per site are shown in Figure 5.34. As is evident from the figure there was considerable variability between sites, and also within sites. However, where high fluxes were observed (e.g. Tick Fen, Warboys, Redmere), all three measurements were high compared to other sites, suggesting that the higher measured fluxes were real, rather than random anomalies. We therefore retained all measurements in the subsequent analysis.

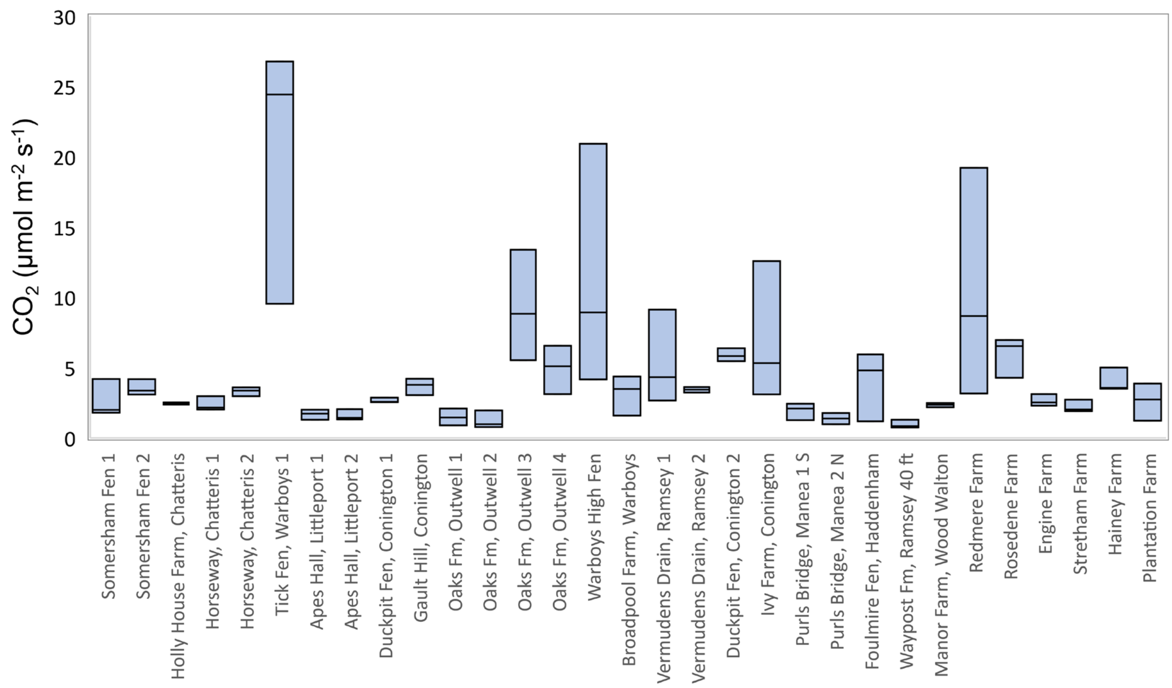



***Figure 5.34**. Boxplots of measured dark chamber $CO_2$ flux for all surveyed sites. Boxes show maximum and minimum measured value, horizontal line shows the median.*

Overall, there was a weak relationship between temperature-adjusted $CO_2$ efflux and any of the measured soil variables (Figure 5.35). Neither peat thickness nor the effective water table depth (WTD*e*) could explain observed variability in $CO_2$ efflux. There was a weak positive relationship with topsoil %SOC, and a correspondingly weak negative relationship with topsoil BD. Most of the sites with low $CO_2$ effluxes (< 5 µmol $m^{-2}$ $s^{-1}$) tended to be those with %SOC < ~15%, whereas the majority of sites with %SOC > 20% had a $CO_2$ efflux > 5 µmol $m^{-2}$ $s^{-1}$. However, there were several apparent outliers with high $CO_2$ efflux despite low %SOC. Comparing $CO_2$ efflux to the amount of C in the soil, there appears to be little relationship with the aerated peat C stock, or the total peat C stock (not shown, but very similar to aerated C stock). The relationship with topsoil C stock appears slightly stronger, but the data is still fairly noisy.

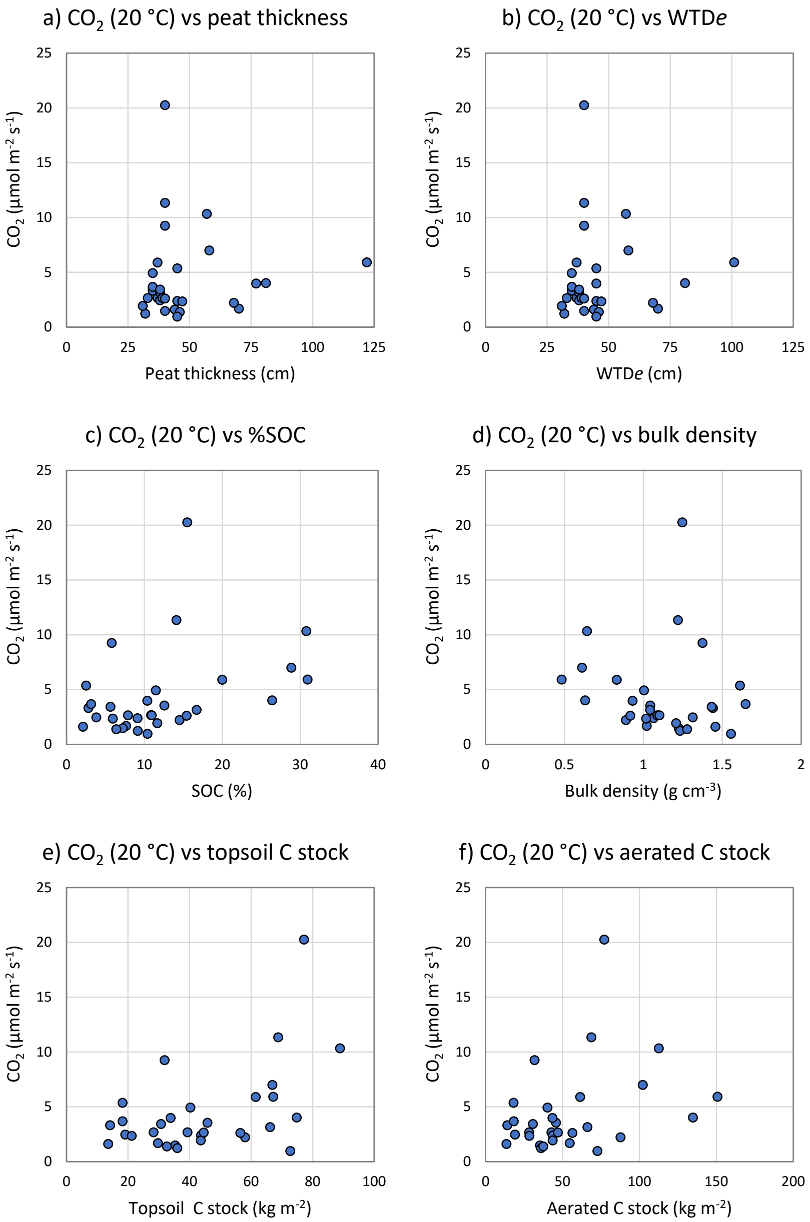



***Figure 5.35**. Temperature-adjusted $CO_2$ efflux for the extensive chamber study versus a range of soil physical and chemical variables.*

### Discussion

One-off measurements of dark-chamber respiration fluxes across a range of cropland sites in the Fens, ranging from thick to wasted peat, showed high variability but little relationship with any measured attributes of the soil. One of the original aims of the study was to compare $CO_2$ fluxes across a gradient of peat drainage depths, but due to consistently deep drainage across all sites, likely exacerbated by the severe drought of summer 2022, the entire peat layer was drained at almost all sites, making it impossible to assess this. We also found no relationship between $CO_2$ fluxes and the depth of the peat layer, or with the total aerated carbon stock. There was some suggestion of weak positive relationships with topsoil %SOC and topsoil C stock, but these were not sufficient to draw reliable conclusions. It is likely that variations in site management, and variable hydrological and meteorological conditions at the time of sampling, led to high spatial and temporal variability in $CO_2$ efflux, to the extent that one-off measurements across a relatively small number of sites could not detect the underlying effects of long-term management of peat properties. Notably, soil temperature at the time measurements were made varied from 13.2 to 35.6 °C, which is a very wide range, and likely to have been accompanied by large changes in soil moisture. The use of a simple $Q_{10}$ function to standardise temperatures across this range is questionable given the likelihood that other factors such as soil moisture also varied over this range which (as demonstrated in the Skyline 2D experiment) almost certainly influence respiration rates. It is possible that laboratory basal respiration measurements on samples collected from these sites, which are made under standardised conditions, may reveal more about the fundamental controls on peat $CO_2$ fluxes across management-related soil gradients; these data are currently being analysed for the BEIS project.

## 5.5 Synthesis

### The role of water table

In the early part of the project, we analysed data from all available UK and Irish flux towers for which at least one year of data were available (seven blanket bogs, one raised bog, three semi-natural fens and five agricultural fens), along with methane flux data collected during the first Defra Lowland Peat project and related work, as part of a synthesis of the relationship between emissions and water table depth which was subsequently published in Nature (Evans et al., 2021a). The analysis suggested that $CO_2$ emissions are linearly related to the 'effective' water table depth, defined as whichever is the smallest of the actual water table depth and the peat depth. This relationship essentially implies that $CO_2$ emissions will increase linearly with increasing drainage intensity until the water table drops below the base of the peat layer, beyond which additional drainage into the underlying mineral soil will not lead to a further increase in emissions. Emissions of $CH_4$ were near-zero for all sites at which the water table was below approximately 30 cm, but increased exponentially as the water table approached the surface. Taking into account the relative 100-year climate-warming impact of $CH_4$ vs $CO_2$, the optimal water table depth from a climate change mitigation perspective was found to be around 5-10 cm, a depth which is typical of many natural peatlands.

In March 2022 we updated the analysis of Evans et al. (2021a) as part of the analysis of emission factors for cropland on wasted peat for the BEIS project, as well as the update of emission factors for the Peatland Code and UK National Atmospheric Emissions Inventory (Evans et al, 2022a,b). The additional sites largely conformed to the existing empirical relationship between $CO_2$ emissions and effective WTD (Figure 5.36a), and indicated that thin and wasted peats conformed to the expectation that their emissions would be lower than for a thick peat where the remaining peat depth was smaller than the water table depth (i.e. WTDe < WTD).

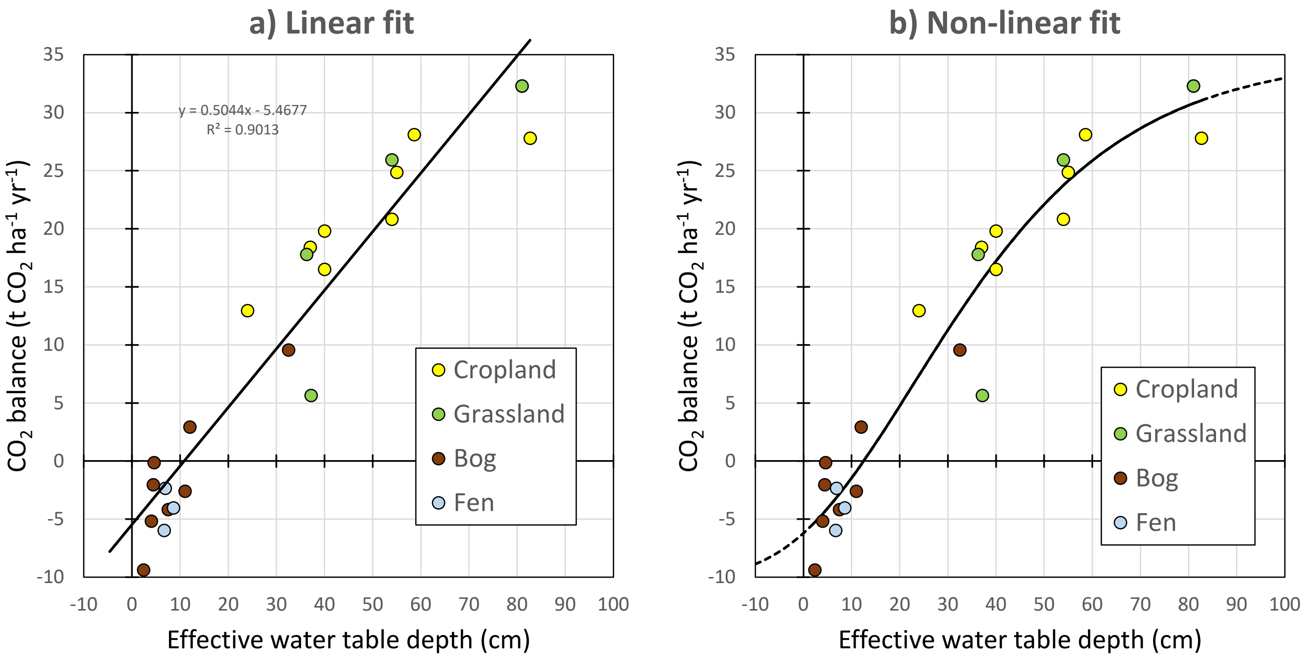



***Figure 5.36**. Linear (a) and non-linear (b) fits to measured $CO_2$ balance versus effective water table depth (WTDe) for all UK flux towers on peat and wasted peat for which annual balances could be derived. Non-linear best fit line in (b) is shown as a solid line within the range of observed WTDe values, and as a dashed line outside this range. Note that flux data were are collated for the Evans et al. (2022a,b) and will be updated following full analysis and QA of new flux data for 2022.*

The linear regression fit shown in Figure 5AAa is highly significant, and similar to that obtained for a smaller dataset by Evans et al. (2021):

$$FCO_{2\,(WTDe)} = 0.5044\,WTDe - 5.47 \qquad (R^2 = 0.90,\ p < 0.001)$$

Where the net $CO_2$ flux at a given value of WTDe ($FCO_{2\,(WTDe)}$) is expressed in t $CO_2$ $ha^{-1}$ $yr^{-1}$ and WTDe in cm. As noted previously there is no evidence that different forms of land-management lead to different emissions beyond their influence on WTD – for example the site with the highest measured emissions is a deep-drained rotational grassland of thick peat, whereas other grassland sites with shallower drainage or thinner peat layers have lower emissions. However, while this relationship appears sufficient to estimate $CO_2$ emissions as a function of drainage and peat depth over the range of observations (mean WTD 2 - 83 cm), it is clearly not realistic to extrapolate the linear function beyond this range of observations – this would result in extreme levels of $CO_2$ emission at very deeply drained sites, and in net $CO_2$ uptake increasing to implausible levels at sites where the average WTD is above the peat surface. In addition, it appears that residuals on the linear regression are non-randomly distributed, with predominantly positive residuals in the middle of the water table range, which suggests some non-linearity in the relationship. Following the approach previously applied for German chamber flux data by Tiemeyer et al. (2020) we therefore tried fitting a non-linear Gompertz function, which fits a sigmoidal relationship between two asymptotes $FCO_{2\,(min)}$ and $FCO_{2\,(max)}$, representing the minimum and maximum possible $CO_2$ flux. The Gompertz function used has the form:

$$FCO_{2\,(WTDe)} = FCO_{2\,(\mathrm{min})} + FCO_{2\,(\mathrm{diff})}\, e^{ae^{bWTDe}}$$

Where $FCO_{2\,(diff)}$ is the difference between $FCO_{2\,(max)}$ and $FCO_{2\,(min)}$, and *a* and *b* are constants. Based on the range of observations we set $FCO_{2\,(min)}$ and $FCO_{2\,(max)}$ to -10 and 45 t $CO_2$ $ha^{-1}$ $yr^{-1}$ respectively, then used the iterative Excel Solver function to derive values of *a* and *b* in order to minimise the root mean squared error (RMSE) of predicted versus observed values of $FCO_{2\,(WTDe)}$. The resulting best-fit equation is:

$$FCO_{2\,(WTDe)} = -10 + 55e^{-2.475e^{-0.0398WTDe}}$$

The Gompertz function fitted to the flux tower data is shown in Figure 5.36b. While it also would be possible to fit a different function if different values of $FCO_{2\,(min)}$ and $FCO_{2\,(max)}$ were chosen, this appears to give a plausible fit to the observations, and largely avoids the non-random residual distribution observed in Figure 5.36a.

Compared to the function derived by Tiemeyer et al. (2020) for German chamber data, our relationship is far less non-linear, and over the central range of WTD*e* values (10 to 60 cm) the choice of a linear versus a non-linear function makes rather little difference to the predicted $CO_2$ emissions. However for very wet and very dry conditions, the non-linear function appears more realistic. Once annual flux data from 2022 have been fully collated (e.g. to include the new Fenland SOIL flux towers) we will update this analysis. If the new data continue to support the use of a non-linear $CO_2$–$WTD_e$ function this could in future be included in simple predictive models such as the Peatland Code 2.0 Carbon Calculator.

Notwithstanding the above analysis, we still have limited data on the effects of *changing* drainage depths on $CO_2$ emissions, as opposed to comparisons of sites with different pre-existing drainage. At a mesocosm scale, the evidence that raising water levels can reduce peat oxidation is now fairly compelling, with multiple UK studies now demonstrating that continuous or seasonal re-wetting can reduce heterotrophic $CO_2$ emissions from cultivated peat (Musarika et al., 2017; Matysek et al., 2019, 2022; Wen et al., 2019a, 2020b; Peduru Hewa, unpublished data). However, demonstrating that similar mitigation can be achieved under field conditions remains a challenge. As described in Section 5.1, we were unable to test this hypothesis with the Skyline 2D experiment due to the difficulties encountered in maintaining water levels within the plots. However with the growing availability of flux tower data, and the growing willingness of farmers to undertake field-scale water level manipulation trials, we were able to establish two 'paired' flux tower studies over the course of the project, on grassland at Pollybell and at cropland at Stretham, which had not been considered feasible at the start of the project. As described in Section 5.2, both field-scale trials were affected by the severe drought of 2022, which made it exceptionally difficult to maintain higher water levels in individual fields within a very dry wider landscape, and with severe restrictions on water for irrigation. At Pollybell, we obtained a relatively short period of comparative data when water levels at the 'wet' P1 field were higher than those in the 'dry' P2 and P3 fields, before a sharp drop in water levels at P1 took place and between-field differences in drainage depth were lost. Heterotrophic respiration rates were clearly lower at P1 during this time that water levels were held at higher levels. This relationship effectively 'flipped' when water levels were drawn down at P1, supporting the interpretation that higher water levels were suppressing $CO_2$ emissions.

The benefits of maintaining higher water levels were more clearly demonstrated at Stretham, where despite some challenges water levels were largely held within the peat layer in the 'wet' field, but remained continuously below the peat layer in the 'dry' field. Subject to some caveats regarding the depth of peat in the two fields, and the possible effects of abrupt water table changes on the wheat crop in the 'wet' field, it appears that holding water tables higher reduced $CO_2$ emissions from around 28 to 21 t $CO_2$ $ha^{-1}$ $yr^{-1}$. Based on the linear regression shown in Figure 5.36a, this would be consistent with reducing the average effective water table depth from 67 cm to 52 cm, which appears broadly consistent with the degree of water level change achieved in the field.

The available evidence thus supports our interpretation from the analysis of data from multiple sites across the flux tower network that raising water levels in agricultural peatlands will generate emissions reductions, provided that water levels are raised *within* the remaining peat layer, and not just in the underlying mineral soil. More field-scale trials would however be beneficial in confirming that this mitigation is achievable, ideally designed in advance as full before-after, control-intervention (BACI) trials using paired flux towers on well-characterised, comparable fields. These trials should ideally be undertaken on peat with different depths (thick, thin, wasted), under different land-use (horticulture, cereals, grassland), and with different seasonal water management regimes (e.g. effects of raising water levels in summer versus winter). In addition to providing improved data on the mitigation benefits of raising water levels, these field-scale trials would also help with understanding the practical challenges and water resource implications of managing water levels higher in agricultural peat landscapes.

### The role of other emissions mitigation measures

A range of mitigation options other than raising water levels were evaluated during the project, many of them as part of linked PhD and postdoctoral research. All of these measures aimed to reduce one or more pathways of carbon loss and/or emission of one or more GHGs based on interventions that could realistically be implemented as part of farming operations. Some options, such as the use of cover crops and low/zero till agriculture, form part of a suite of measures often referred to as 'regenerative farming', a concept that continues to evolve for mineral soils, but which is increasingly now being applied to peaty soils. Other potential mitigation measures include the application of soil amendments such as different forms of organic matter, nitrification inhibitors and soil stabilisers. Here we summarise our current knowledge of the effectiveness of these measures based on work undertaken during the project, work by affiliated PhD students, and work initiated during the preceding NERC SEFLOS project which was completed and published during the current Defra project.

### *Irrigation*

Existing crop management frequently involves the use of irrigation, either to the soil surface (via sprinklers) or subsurface via seepage from the ditch network into the field. Subsurface irrigation, normally supported by the use of regularly spaced subsurface drains running in parallel across the field, is effective at raising water levels within remaining areas of thicker peat, especially where this has been levelled to ensure minimal topographic variation across fields. This approach is used at sites such as Rosedene Farm to manage water levels beneath high-value horticultural salad crops. However, subsurface irrigation is problematic in areas with thinner peat, greater topographic variation, and greater spatial variation in soil type, for example due to the presence of roddons (former river beds) within fields. Furthermore, in areas of thin or wasted peat, raising water levels to a point where they will begin to reduce emissions (i.e. to within the remaining peat layer) is challenging, and would preclude cultivation of many crops. For this reason, surface irrigation may offer a more practical option to maintain high water levels in thinner peatlands, or those that lack established networks of subsurface drains. Some previous work from Southeast Asia (Evans et al., 2021b) has shown marked reductions in peat subsidence in surface-irrigated cultivated peatlands compared to unirrigated sites with similarly deep drainage, suggesting that this measure may help to slow rates of carbon loss in areas where raising water levels is not viable.

As described above, raising near-surface soil moisture during dry conditions in the Skyline plot trial did not reduce $CO_2$ emissions, and may in fact have increased them by raising soil moisture to (but not above) field capacity after irrigation events, creating optimal conditions for microbial decomposition of organic matter, but without reducing oxygen levels sufficiently to constrain microbial activity. There was also some indication of higher $N_2O$ emissions after planting and harvesting of the lettuce crop. These results clearly do not support the use of surface irrigation as a mitigation measure, although there are several reasons why they may represent a worst-case scenario. Firstly, the soil at the start of the experiment was exceptionally dry, to the extent that biological activity may have become moisture-limited near the peat surface. Surface irrigation may have overcome this moisture-limitation to biological activity, but with ditch levels low and the remainder of the field not receiving the same level of irrigation, it would not have been sufficient to raise the water table. Surface irrigation under these conditions may therefore have increased decomposition at the peat surface, but without suppressing it at depth. Exceptionally high soil temperatures likely amplified this effect. We cannot therefore exclude the possibility that surface irrigation could deliver mitigation benefits under more favourable conditions, for example in spring when existing soil moisture is higher, ditch levels are higher (enabling the soil to retain higher moisture levels near the surface via capillary action) and temperatures are lower.

Further experiments using the established Skyline system under different seasonal and hydrological conditions are needed to investigate this possibility.

### *Cover crops*

Cover crops are increasingly widely used to retain nutrients in the soil over winter, and to reduce the exposure of bare soil to oxidation or wind erosion. Cover crops are usually ploughed back into the soil at harvest, where they may add to soil organic matter or may simply decompose back to $CO_2$. Water demand by growing cover crops could also have a drying effect on the soil, potentially increasing heterotrophic $CO_2$ emissions if soil moisture levels are brought into the optimal range for microbial activity.

New evidence produced during the project includes the mesocosm study of Wen et al. (2020b), initiated during SEFLOS, which compared GHG fluxes from vetch and rye winter cover crops with those from bare peat under relatively high (-15 cm) and low (-40 cm) water levels. The results showed clear suppression of $CO_2$ losses via respiration with higher water levels (consistent with the field data from SP1218) but no clear direct effect of the cover crops on $CO_2$ emissions. On the other hand, N uptake by the cover crops reduced porewater nitrate levels, which would be expected to lead to a subsequent reduction in $N_2O$ emissions.

With regard to wind erosion, Newman (2021) measured wind-borne transport of particles at two of the flux tower sites, Rosedene and Engine, as part of his PhD project, while Freeman (in prep) measured soil susceptibility to mobilisation for different sites, stages of the cropping cycle and wind speeds. The results of both studies showed that erosion risk is highest for soils with a higher organic content, and during periods when soils are bare and dry. Erosional losses were reduced when crops (including cover crops) were present on the fields, and cover crops along field margins may even help to capture mobilised organic matter and retain it in the field. These results support expectations that cover crops could help to mitigate carbon loss via wind erosion. However, in general wind erosion also tends to be highest during dry periods, whereas cover crops are often grown in winter when soils are wet and therefore less susceptible.

Overall, there is limited evidence to suggest that cover crops can reduce $CO_2$ emissions via heterotrophic respiration, and some risk that they could even increase emissions if they exacerbate soil drying. On the other hand, the use of cover crops to maintain near-continuous vegetation cover on the field almost certainly reduces the risk of soil carbon loss via wind erosion.

### *Soil stabilisers*

Chemical soil stabilisers are high molecular weight organic polymers, used widely for erosion control on unpaved roads, storage piles and other open surfaces in industrial/construction environments (USEPA, 1992). A wide variety of chemicals have been used for these purposes including polyacrylamide, polyvinyl acetate, petroleum resins, tree resins and asphalt emulsion, among others. Their mechanisms of action generally involve increasing adhesive forces between particles and/or creating a resistant surface crust. The capacity to reduce wind erosion losses on mineral soils has been demonstrated for polyacrylamide (Genis et al., 2013), polyvinyl acetate (Feizi et al., 2019), petroleum resin and asphalt emulsions (Lyles et al. 1974) and tree resins (Kavouras et al., 2009; Robichaud et al., 2017). However, in practice the effects of chemical soil stabilisation in agricultural contexts have been mixed; sometimes offering little advantage over natural rainfall (e.g. Armbrust, 1999; Lyles et al., 1974; Van Pelt and Zobeck, 2004). Effectiveness and duration of effect appear to depend on soil physicochemical properties, environmental conditions and application rates/mixtures. Freeman (in prep) tested several commercially available products, on constructed plots of high organic matter content peat soil, at manufacturer recommended application rates, under laboratory conditions. Results were mixed and it is not currently clear whether effective application rates are economically or agronomically practical.

If effective, chemical soil stabilisation would offer a possible solution for erosion control where/when increasing vegetation cover through cover crops or companion crops is not possible. This may apply to localised areas of vulnerability (e.g. farm tracks) or periods of vulnerability (e.g. after harrowing seed beds for planting of vegetable crops). However, whilst chemical soil stabilisers are appealing in an agricultural context because of the targeted nature of their application, their use faces several challenges. They must be cost effective and easy to apply in large quantities (Lyles et al., 1974). They must also produce persistent adhesion of surface particles without impeding infiltration of water, impeding seedling emergence, or producing negative longer-term impacts on soil structure (Armbrust and Lyles, 1975). They must also be environmentally safe, noting that even where the main chemical components are considered safe, toxicity can result from surfactants or emulsifiers added to product mixtures (Weston et al., 2009). Therefore, the environmental and food safety of individual products considered for agricultural use would need to be assessed based on their specific composition. It is not currently clear whether these conditions can be met for erosion control use of chemical soil stabilisers on peaty soils.

### *Crop residue incorporation*

Crop residues are frequently left on the field after harvest, and either left to decompose on the surface or ploughed into the soil. Given that these residues contain new organic matter, there is potential for this to add carbon to the peat and offset some of the oxidative losses that result from drainage. However, much of this organic material is highly reactive and unlikely to persist in the soil, and in addition there is a risk that this addition of reactive material will increase biological activity in the soil and lead to accelerated decomposition of the peat. Concerns about this 'priming' mechanism have led to the exclusion of organic soils from ELMS measures to enhance soil carbon stocks via residue incorporation, despite such practices already being commonplace.

Specific data on organic matter additions to agricultural peat soils are limited, and inconclusive. Wen et al. (2019a) tested the impacts of incorporating rye and vetch cover crops into peat mesocosms, and measured peaks in $CO_2$ emissions, but the amount of C lost over the 80 day experiment was less than the amount of C added in the biomass. Incorporation of the vetch also apparently caused a pulse of $N_2O$ emissions, whereas $N_2O$ emissions following rye incorporation were less than in the controls. The PhD study of Sam Musarika (Musarika, 2022) tested the impact of adding barley straw to peat mesocosms, in a series of experiments that were affected by the Covid lockdown. Again, the results showed increased $CO_2$ emissions from cores to which straw had been added, but the amount of carbon lost as $CO_2$ was less than the amount of carbon added in the straw. Given the limited duration of both experiments, it was not possible to establish whether any of the added carbon was retained in the longer term, or whether any priming of peat decomposition took place. Based on the long-term flux tower data, however, there is little evidence to suggest that annual rates of $CO_2$ loss varied systematically as a function of crop type, or of associated variations in the amount of quality of crop residues left in the field. It is possible that residue incorporation offers greater benefits in more organic matter-depleted wasted peat soils, but further work is needed to confirm this.

*Soil amendments*

A range of different soil amendments were trialled during the Defra projects, SEFLOS, linked PhDs and the ongoing Peat GGR project. This section briefly summarises the main findings. **Nitrification inhibitors** such as DCD (dicyandiamide) are designed to reduce the concentration of free nitrate in the soil, and therefore the rate of $N_2O$ emission. They were tested during the first Defra lowland peat project and found to have variable effects, from an apparent (but statistically non-significant) 50% reduction in $N_2O$ emissions from a potato crop on thick fen peat, to little or no effect on a wheat crop on thick bog peat (where $N_2O$ emissions were already low) or on an intensive grassland (Evans et al., 2016). Overall, the evidence for effectiveness of nitrification inhibitors on organic soils does not appear sufficient to support their widespread use as a mitigation measure, but further work to establish whether they may be effective under certain conditions (for example during periods of high soil moisture and low plant demand, which the Skyline 2D experiment suggests lead to high $N_2O$ emissions) would be beneficial.

Application of **iron** (Fe) compounds to organic soils could reduce $CO_2$ and $CH_4$ emissions by reducing organic matter solubility and therefore the amount of substrate available to decomposers and methanogens. Iron sulphate could additionally help to suppress $CH_4$ emissions from re-wetted peat, because sulphate-reducing bacteria outcompete methanogens for available substrate. A mesocosm experiment by Wen et al. (2019b) produced mixed results, but suggested that iron(II) sulphate addition reduced soil organic matter decomposition rates. Ongoing mesocosm experiments for the GGR Peat project (Peduru Hewa, unpublished data) suggest that iron sulphate application may suppress $CO_2$ emissions from fresh organic matter, along with emissions of $CH_4$ and $N_2O$. These results are not fully transferrable to drained agricultural soils, but the suppressive effect of Fe on organic matter degradability is likely to apply to all soils. However, in practice it is unlikely that farmers would choose to apply iron sulphate to their soils, because of the risks of soil acidification and the leaching of insoluble iron hydroxide (ochre) from the soil, which can clog up subsurface drains and ditches, and damage aquatic ecosystems. Calcium sulphate (**gypsum**) provides an alternative means of suppressing $CH_4$ emissions, which is now being trialled for the Peat GGR project, but this is only relevant for re-wetted soils.

Finally, **biochar** application to peat soils could provide a mechanism by which to add carbon to peat soils in a more persistent form than standard crop residues, and with a lower risk of priming peat oxidation. Biochar is widely assumed to be unreactive in all soils (mineral or peat, drained or undrained), but this partly depends on the nature of the biochar applied. Biochar produced via high temperature pyrolysis is highly unreactive, but energy intensive to produce, and much of the carbon contained in the feedstock is lost as $CO_2$ during the production process. Biochar produced at lower temperatures retains more of the original feedstock carbon but is less stable under aerobic conditions, precluding its use on mineral soils. In organic soils, however, and particularly where these soils have been re-wetted, this material may be more stable, making peat a potentially efficient environment in which to store biochar. There is also some evidence that biochar application can suppress $N_2O$ and $CH_4$ emissions. Mesocosm experiments comparing biochar stability to a range of other organic amendments for the GGR Peat project show considerable promise, with very little of the biochar carbon being lost as $CO_2$ over a period of 250 days (to date; the experiment is ongoing), compared to higher losses of all other amendments (Peduru Hewa, unpublished data). These experiments also show effective suppression of both $CH_4$ and $N_2O$ emissions. The experiments have been undertaken on re-wetted agricultural peat, however, and the extent to which biochar application could be effective under drained conditions remains to be fully tested. However, results to date suggest that biochar application could offer an effective climate change mitigation measure, ideally in combination with the raising of peat water levels, where co-located feedstocks and pyrolysis facilities are available.

### *Conversion to grassland*

On average, grasslands emit lower amounts of $CO_2$ than croplands on peat of the same depth, resulting in lower Tier 1 and Tier 2 emission factors for both intensive and extensive grassland (IPCC, 2014; Evans et al., 2017). This might suggest that conversion to grassland could offer an effective mitigation option. However, emissions from the peat will only be reduced if water levels are raised to those typical of grasslands on peat in general, i.e. simply converting cropland to grassland without changing water management would not be expected to lead to any reductions in $CO_2$ emissions (see flux tower analysis above). In addition, our analysis does not incorporate livestock-related emissions of $CH_4$ and $N_2O$ resulting from enteric fermentation and animal wastes, which can occur either on the field or remotely where animals are fed from hay and silage grown on the field. These emissions could cancel out much of the mitigation benefit of cropland to grassland conversion (e.g. Wen et al., 2021). Finally, most land-use scenarios for achieving net zero emissions (e.g. CCC, 2021) involve a reduction in meat and dairy consumption, and an increase in plant-based foods, implying a reduction in the overall area of grassland required in the UK. Shifting land-use on peat from cropland to grassland may therefore be inconsistent with the UK's wider net zero land-use strategy, and also risks displacing emissions from crop production to other areas (e.g. Rhymes et al., 2022).

## Updated lowland peat emissions estimates

A major update and alignment of all Tier 2 emission factors (EFs) used in the UK National Atmospheric Emissions Inventory and Peatland Code was undertaken during 2022, drawing on a comprehensive analysis of all flux tower data collected up until the end of 2021, as well as a collation of other recently published literature led by the James Hutton Institute. The report containing the full EF update was recently published in a report to Defra to support version 2.0 of the Peatland Code (Evans et al., 2022a). An accompanying report to BEIS (Evans et al., 2022b) developed a new EF for $CO_2$ emissions from cropland on wasted peat, based on flux tower data collected in both projects. Given that this update happened relatively recently we have not attempted a further update at this point, although the new flux tower data collected in 2022 could support another update in due course, particularly for the lowland agricultural peatlands.

New Tier 2 EFs for the main categories relevant to lowland agricultural peatland regions are shown (in t $CO_2$e $ha^{-1}$ $yr^{-1}$) in Table 5.8. Compared to the previous assessment of Evans et al. (2017), emissions for near-natural fen are largely unchanged. For re-wetted fen, stricter inclusion criteria for published studies (excluding sites that were either permanently flooded or where water tables remained far below the surface) resulted in this category becoming a small net $CO_2$ sink rather than a small net source, and having slightly lower $CH_4$ emissions. For both grasslands and croplands on peat > 40 cm, a combination of new data from the UK flux towers and the exclusion of some published static chamber studies following identification of apparent methodological issues led to downward revisions of the previous Tier 2 EFs for $CO_2$, particularly for the two grassland categories. The new $CO_2$ EF for wasted peat is approximately 60% of the updated EF for cropland on peat > 40 cm, indicating that wasted peats are smaller but still substantial sources of emissions. A separate $CO_2$ EF for grassland on wasted peat could not be derived due to insufficient data, but the EF for intensive grassland on peat > 40 cm is already slightly lower than the EF for cropland on wasted peat.

***Table 5.8***. *Combined Tier 2 emission factors for gaseous GHG source/sink pathways for the main lowland peat categories in the UK National Atmospheric Emissions Inventory, expressed in t $CO_2$ $ha^{-1}$ $yr^{-1}$ based on IPCC AR5 100-year Global Warming Potentials (28 for $CH_4$ and 265 for $N_2O$). Emission factors based on IPCC Tier 1 defaults are shown in italics. All data are from Evans et al. (2022a).*

| **Peat Condition category** | **Direct $CO_2$** | **Direct $CH_4$*** | **$CH_4$ from Ditches** | **Direct $N_2O$** | **Total** |
|---|---|---|---|---|---|
| Near-Natural Fen | -5.06 | 4.01 | *0* | *0* | **-0.36** |
| Rewetted Fen | -0.69 | 3.12 | *0* | *0* | **3.31** |
| Grassland – Extensive | 11.78 | 0.96 | *0.74* | 0.76 | **15.88** |
| Grassland – Intensive | 14.87 | 0.77 | *1.63* | 3.08 | **22.00** |
| Cropland (peat > 40 cm) | 27.06 | 0.05 | *1.63* | 6.78 | **37.17** |

Table 5.9 shows the total estimated $CO_2$ emissions from lowland cropland and intensive grassland on peat in England based on the IPCC Tier 1 EFs, the Tier 2 EFs previously used in the UK inventory, and the new Tier 2 EFs following the updates above. These two categories make up the vast majority (> 99%) of the English agricultural peatland area. The previous Tier 2 EFs were not greatly different from the IPCC Tier 1 EFs, due to the similar dataset used to derive the two sets, and wasted peat was assigned the same EF is remaining areas of peat > 40 cm, so total emissions estimates were quite similar. However, the introduction of a separate EF for on wasted peat reduces total cropland emissions by 1.62 Mt $CO_2$ $yr^{-1}$, and revisions to the Tier 2 EF for intensive grassland reduce total emissions from this category by a further 1.13 Mt $CO_2$ $yr^{-1}$. The revised total $CO_2$ emission for cropland and intensive grassland on English peat of 6.26 Mt $CO_2$ $yr^{-1}$ is almost one third lower than the previous estimate, although this lower value still represents almost 2% of total UK $CO_2$ emissions for

2020 (ONS, 2021). The downward revision of the EFs based primarily on new flux tower measurements and understanding highlights the value of collecting new, robust data from representative locations.

***Table 5.9**. Total $CO_2$ emissions (in Mt $CO_2$ $yr^{-1}$) from cropland and intensive grassland on lowland peat for the UK as a whole, based on previous and revised emission factors.*

| | **Cropland** | | | **Intensive Grassland** | | | **Combined** | | |
|---|---|---|---|---|---|---|---|---|---|
| **Peat type** | > 40 cm | Wasted | Total | > 40 cm | Wasted | Total | > 40 cm | Wasted | Total |
| **Tier 1** | 1.75 | 3.83 | 5.58 | 3.16 | 0.79 | 3.95 | 4.91 | 4.62 | 9.53 |
| **Tier 2 (2019)** | 1.73 | 3.78 | 5.51 | 3.01 | 0.75 | 3.76 | 4.74 | 4.53 | 9.27 |
| **Tier 2 (2022)** | 1.51 | 2.11 | 3.63 | 2.10 | 0.53 | 2.63 | 3.62 | 2.64 | 6.26 |

# 6. Assessment of the economic, environmental and social impact and practicality of mitigation measures

## 6.1 Efficiency and practicality of mitigation measures

The following section synthesises the methods and main findings of work that was described in greater detail in previous reports to Defra. These will also be published on Defra R&D pages as an annex to the project: Managing agricultural systems on lowland peat for reduced GHG emissions - SP1218 (Arnott, et al., unpublished). We also plan to submit this work for peer-reviewed journal publication.

### Introduction

The research described in the preceding section has helped to identify a range of potential greenhouse gas reduction measures, although (as discussed) some appear more effective than others. The successful on-farm adoption of these measures is contingent upon farmer perceptions of the relative practicality of implementing the measures, and the economic impact that adoption will have on the farm business. It is also important to understand the critical barriers to their adoption, as well as any opportunities they may offer, in order to develop an appropriate policy and regulatory framework, and to provide appropriate infrastructure and financial support mechanisms. As identified during the Defra Lowland Agricultural Peat Task Force, overcoming these barriers will be essential if we are to reduce greenhouse gas emissions and the other negative consequences of drainage-based agriculture on peat, and deliver the UK's Net Zero and biodiversity targets, whilst maintaining a healthy food and farming sector.

### Methods

We undertook a discrete choice survey method, Best Worst Scaling (BWS) to elicit expert (climate change, policy and biodiversity) and farmer opinion on the relative effectiveness, practicality and level of economic impact of mitigation measures (MMs) to reduce greenhouse gas emissions at the farm level. The method enabled the individual MMs that were explored to be ranked by effectiveness (based on the opinion of a range of peatland experts), practicality and economic impact (via farmer opinions). This was followed by four focus groups and three interviews (n = 26 in total) that were undertaken across three major lowland peat regions in England (Northwest, East Anglian Fens, Somerset Levels and Moors). These focus groups sought to identify the main barriers to the implementation of these strategies, how much their implementation would impact on profit per hectare, and the levels of incentive/carbon credit payments needed to switch farming practice to incorporate strategies that raise water levels across the whole farm.

### *Best-Worst Scaling*

An initial list of 73 candidate MMs were identified from relevant peer-reviewed papers and grey literature. These MMs were grouped by the following themes: nutrient management (x 15); soil moisture management (x 7); crop management (x 10); fallow and residue management (x 5); tillage and machinery operations (x 3); general soil management (x 7); fossil fuel consumption (x 5); carbon sequestration (x 7); paludiculture (x 7) and 'other' mitigation measures (x 7). The 73 MMs were reduced to 50 by removing any identified in previous studies as being slightly or not effective, and the list of MMs was further shortened to a manageable 30 by panel of 13 peatland experts. The top 30 scoring MMs (soil moisture management x 7, paludiculture x 6, C sequestration x 5, nutrient management x 4, tillage and machinery operations x 2, fossil fuel reduction x 2, fallow and residue management x 1 and miscellaneous items x 1) were subsequently used to populate the BWS surveys.

For the 'effectiveness' BWS, experts with knowledge of GHG and peat loss mitigation measures were recruited from academia, government and environmental NGOs. For the 'practicality' and 'economic impact' BWS, farmers and other landowners were recruited through engagement with the National Farmers Union (NFU) and other stakeholder contact lists. In total, data from 27 experts, and from 141 (practicality BWS) and 121 (economics BWS) farmers, were used in the subsequent analysis.

### *Focus groups and Interviews:*

Four focus groups, each lasting 1-2 hours, and three one-to-one interviews lasting ~1 hour were conducted online via a Microsoft Teams conference call. The focus groups were semi-structured and consisted of open-ended questions which were approved by Bangor University Ethics Committee and Defra Survey Control. These questions were designed to focus the discussion on issues surrounding lowland peats and water management MMs to clarify and expand upon the findings of the BWS survey.

The findings of BWS farmer surveys guided focus group and one-to-one discussions, and this also guided the operationalising of categories and themes. Participants were given the opportunity to comment on their feelings on GHG reduction, the practicality and economic impact of implementing GHG MMs on the farm and the level of compensation required to adopt MM strategies. Comments were collated and analysed using a deductive content processing and arranged under five primary category headings (feelings on GHG reduction; mitigation measures; incentives to encourage the implementation of GHG and soil erosion mitigation; information; and control). Data from the five primary categories were used to reduce the categories to three focus areas (Farmer types, mitigation measures including barriers and positive actions, and compensation levels).

## Results

### *Best-Worst Scaling*

In the farmer BWS, no MMs were ranked as being effective and practical or effective with low economic impact. This makes it unlikely that (based on current financial incentives) farmers will be prepared to adopt the MMs deemed most effective by peatland experts, which generally involved major re-wetting and land-use change to peatland restoration or paludiculture, on a large scale. On the other hand, farmers ranked a number of MMs as being practical with low economic impact, for example those addressing more effective nutrient management and an increased reliance on legumes, a move towards reduced or zero no tillage, the installation of buffer zones, increased fossil fuel efficiency, and the optimisation of irrigation systems which keep the soil moist but not saturated. Individually these MMs did not rank high for effectiveness at the time that the assessment was undertaken, and our subsequent results (described in Section 5) support the conclusion that some of the MMs favoured by farmers would deliver limited (or in some cases possibly no) mitigation of $CO_2$ emissions. On the other hand, there is reasonable evidence that some of these measures would likely reduce the risk of wind erosion, and any measures that reduce nitrate concentrations in the soil can be expected to reduce the risk of $N_2O$ emissions, and improved nutrient management could also help to reduce $CH_4$ emissions from ditches. A combination of these MMs implemented at the farm scale could therefore contribute to reductions in overall GHG emissions and rates of peat carbon loss. As these MMs are practical to implement policy makers may, in the short term, look to put together a package containing these MMs for inclusion in ELMS. Beyond this, and as identified by the Defra Lowland Agricultural Peat Task Force, additional incentives are likely to be needed to facilitate implementation of the MMs identified in this study as being the most effective.

### *Focus groups and interviews:*

Based on the results of the BWS analysis, and wider results from the project, it is clear that the most effective options to mitigate GHG emissions from agricultural peatlands will require changes in water management (see Section 5, and Freeman et al., 2022). However, those farmers surveyed considered all water management or conversion strategies to be impractical to implement, with a high economic impact on the farm business. In order to seek solutions to this challenge, the focus groups therefore primarily considered issues related to soil moisture and water management, with a focus on those MMs identified as being most effective at reducing GHG emissions and carbon loss.

The focus groups confirmed that implementing changes in water management would be extremely challenging for most farms, especially if they were to try to implement them at an individual farm level without supporting infrastructure. However, the results of the focus groups study suggested that more farmers would be willing to change farming practice, and adopt mitigation measures and strategies involving multiple mitigation measures, than the BWS farmer survey would suggest. Nine percent of participants are proactively implementing strategies to better understand GHG fluxes from different soil types, and 34% expressed a willingness to implement production techniques such as paludiculture which would see them move away from food production. A significant proportion of farmers (56%)

in what we termed the ‘food producer’ category also expressed a willingness to convert some of the farm to wetter practices if it was financially viable to do so and the infrastructure was in place. While we recognise that membership of the focus groups was limited and potentially not representative of the entire farming community, these findings offer some encouragement that large-scale change in lowland peat landscapes could be achieved, if appropriate support and incentives were put in place.

It is clear that policies to incentivise farmers to rewet or raise water levels over the large areas of lowland peats soil required to deliver the Net Zero Strategy need to take full account of the complexities of doing so. There is a need to initially invest in water management (storage and distribution) infrastructure at a catchment scale and Internal Drainage Board level, to allow individual farmers to raise water levels across the farm without impacting on others within the local community. The focus groups confirmed that budgeting to fund peat restoration to this level will be challenging, as in addition to (removal and replacement) infrastructure costs there are (at present) capital costs associated with decreases in land value that would need to be accounted for. Participants also highlighted the potential social costs of full rewetting, as intensive arable and horticulture businesses employ large numbers of farm workers. While higher water level management of farmland would not necessarily reduce employment, it is probable that a large-scale shift to restoration or paludiculture management would. Farmers are unlikely to adopt conversion or wetter farming practices unless they are guaranteed to be compensated for losses and supported in transitioning to new farming practices. Support schemes must be long term if they are to provide a level of security which outweighs the risk, for example to enable farmers to invest in the new machinery required to operate on wetter soils.

A clear vision for lowland peat restoration at a regional level, supported with clear and usable information and data showing how lowland peats fit into the bigger picture will go some way to overcoming a general lack of trust in government policy and agencies. This study shows that many farmers are prepared to adopt wetter farming practices and that there may be compromise and cooperation between more sustainable food producers and producers willing to adapt to non-food production. This will however need significant physical and financial input from the government. The broader issue of maintaining a secure and affordable food supply at a UK scale was not specifically addressed in the focus groups but clearly needs to be taken into account as part of any prospective move away from food production on lowland peatlands.

## 6.2 Farm-scale economic analysis of mitigation options and regional upscaling

The following analysis forms the main deliverable for WP4.4 of the project and is reported here in full.

### Introduction

We compared the economic and environmental performance of a range of alternative management scenarios built around the most effective and practical greenhouse gas (GHG) mitigation strategies identified during the field research programme (Section 5) and the farmer consultations (Section 6.1). Based on the results of the field programme in particular, we focused on management options that involved changes in water table depth (WTD) relative to a business-as-usual (BAU) baseline. This assessment was repeated for areas of remaining thick peat, and for a typical wasted peat. In each case, we evaluated how management changes would impact on costs, financial returns, GHG emissions and nutrient burdens. This information was used to identify the most cost-effective means of reducing GHG emissions through changes in the agricultural management of lowland peat.

### Methods

We constructed a Microsoft Excel based tool (Figure 1) to compare potential 'new' management systems to a business-as-usual scenario based on:

- Financial performance - gross and net margins.
- Environmental performance - carbon footprint/GHG emissions and the acidification and eutrophication burden.
- Cost-effectiveness - GHG saving per £1 of annual operating cost.

Within our modelling tool, we used data derived from WP3, WP4.1 and 4.2; actual farm data; other industry sources (e.g., John Nix Pocketbook for Farm Management); and data from the peer reviewed literature. The model was built on lookup tables of GHG emissions factors for different water table depths, allowing users to alter WTD depths (within defined ranges) to see how this impacts GHG emissions. The model can be parameterised with farm specific data relating to cultivations, inputs, and costs.

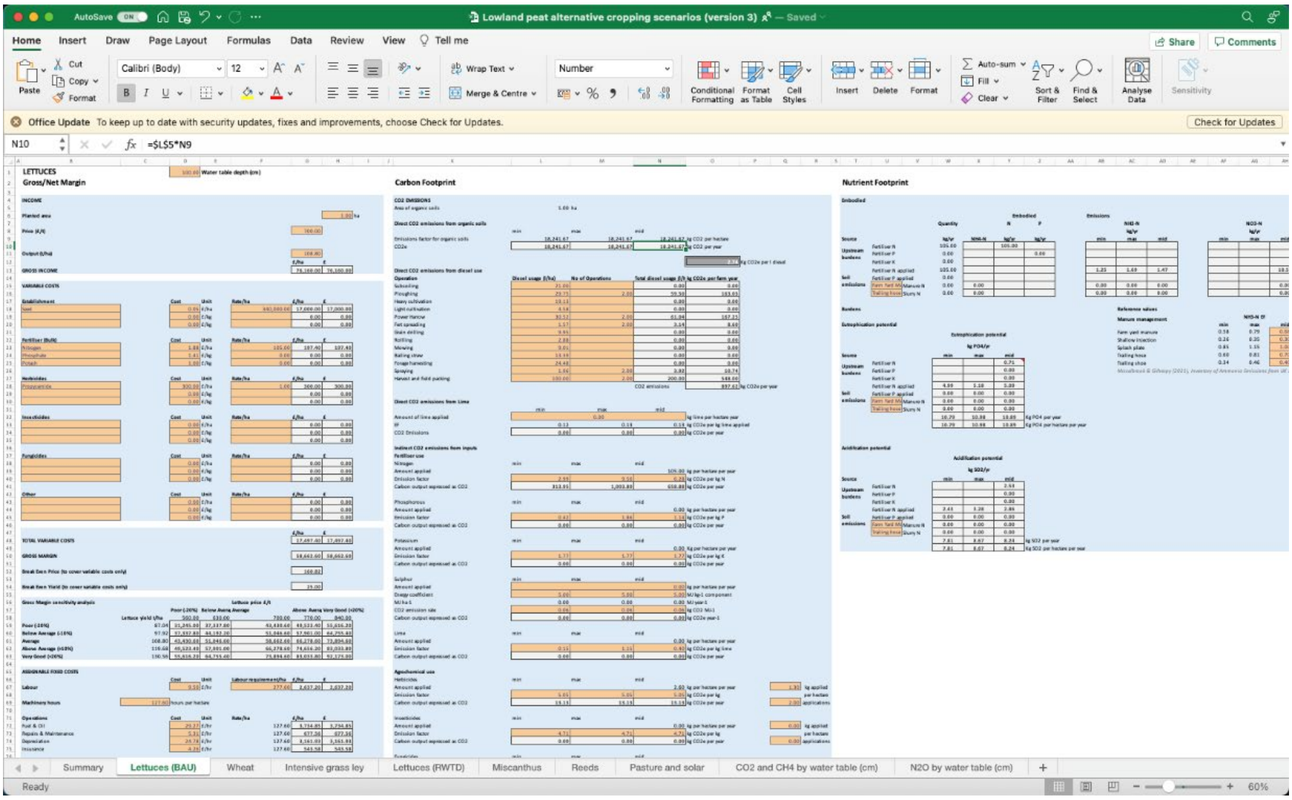



***Figure 6.1.*** *Screenshot of one of the scenario modules within the modelling tool showing financial margins on the left, carbon footprint in the centre and nutrient burdens on the right. Orange cells allow the model to be tailored to different peat depth and water table depth. The orange cells also allow the tool to be parameterised with farm-specific information relating to inputs, cultivations, and outputs.*

### Scenarios

We bundled the most practical and efficient mitigation measures identified in the preceding work into seven illustrative land management scenarios, based around changes in water table depth and agricultural 'enterprise'. We used our Excel-based tool to model the financial and environmental outcomes of these seven scenarios on thick peat soils (assuming a depth of > 100 cm) and wasted peat soils (assuming a depth of 40 cm). This allowed us to model the impacts of a situation where a) the water table remains within the remaining peat layer and b) the water table lies below a residual peaty plough layer. We assumed a different BAU agricultural enterprise for these two situations, reflecting current agricultural management practices. The seven land management scenarios that were modelled using our tool were as follows:

1. **Scenario one:** 'Business as usual' management with a 100 cm WTD. For the thick peat assessment we assumed high value horticulture with lettuces as the default agricultural enterprise, and for wasted peat we assumed arable cropping with wheat as the default agricultural enterprise.

2 **Scenario two:** WTD was reduced to 50 cm, and arable cropping with wheat was the agricultural enterprise (for both soil types).

3 **Scenario three:** WTD was reduced to 40 cm, and a five-year non-grazed intensive grass ley was established as the agricultural enterprise, with two cuts of silage per year.

4 **Scenario four:** WTD was reduced to 30 cm and high value horticulture growing two crops of lettuce was the agricultural enterprise.

5 **Scenario five**: WTD was reduced to 20 cm, and paludiculture with Miscanthus (*Miscanthus x giganteus*) over 20-year crop lifetime was the agricultural enterprise.

6 **Scenario six**: WTD was reduced to 10 cm and paludiculture with common reeds (*Phragmites australis*) over a six-year crop lifetime was the agricultural enterprise.

7 **Scenario seven**: WTD was reduced to 5 cm, and establishment of no input (non/minimally-grazed) permanent pasture with a lease for solar-pv installation was the agricultural enterprise.

### *Financial performance*

Within our Excel-based tool we used gross and net margins to compare the financial performance of each of the seven alternative management scenarios on both deep and shallow peat soils. We calculated the gross margin of each management scenario as:

*Gross margin (£/hectare) = income – variable costs*

Variable costs included establishment (seed, plugs or rhizomes), fertiliser and sprays. We calculated the net margin of each management scenario as:

*Net margin (£/hectare) = income – (variable costs + fixed costs)*

Fixed costs included labour, machinery costs (fuel, repairs, depreciation, and insurance), cultivations and assignable farm overheads (maintenance, utilities, and general overheads).

We took general agricultural management (including fertiliser requirements and cultivations) and budgeting data (including income, variable costs and fixed costs) from the John Nix Pocketbook for Farm Management (Redman, 2022). We took operational data for high value horticulture with lettuces from a range of sources in the literature (Bartzas et al., 2015; Canals et al., 2008; NSW DPI, 2013). We took costs for specialist paludiculture establishment from specialist online information (e.g. https://www.crops4energy.co.uk/).

### *Environmental performance*

Within our Excel-based tool we used calculated GHG emissions and nutrient burdens to compare the environmental performance of each of the seven alternative management scenarios on both thick and wasted peat soils.

*GHG emissions*

GHG emissions were calculated using our Excel-based tool were based on $CO_2$, $N_2O$ emissions (both direct and indirect) and $CH_4$ emissions (all expressed as $CO_2e$) up to the 'farm gate'. $CO_2$ emissions included direct emissions from organic soils, diesel use and lime and Indirect emissions from fertiliser use and agrochemical use. $N_2O$ emissions from included direct emissions from organic soils, fertilisers, crop residues returned to soils and indirect emissions from ammonia volatilisation (and subsequent nitrogen deposition) and nitrate leaching and runoff. $CH_4$ emissions included only those from organic soils due to our management scenarios not involving any livestock. All results were converted to $CO_2$ equivalent emissions based on IPCC AR5 100-year Global Warming Potentials for $CH_4$ and $N_2O$.

We followed the *IPCC Guidelines for National Greenhouse Gas Inventories Volume 4 AFOLU* (IPCC, 2019) for calculating indirect $CO_2$ emissions from fertiliser, direct $N_2O$ emissions from fertilisers and crop residues returned to soils, and indirect $N_2O$ emissions from volatilisation and leaching and runoff. Emissions of $CO_2$ were calculated based on the linear empirical relationship between emissions and effective WTD derived from the UK flux tower network as published in Evans et al. (2021a) and described in Section 5 of this report (note that we did not implement the non-linear function between $CO_2$ emissions and effective WTD at this stage, although this may be implemented in future versions). Emissions of $CH_4$ were also calculated from the empirical function described in Evans et al. (2021a), while direct soil-derived $N_2O$ emissions were calculated from the updated emission factors used in the UK National Atmospheric Emissions Inventory (shown in Table 5.8 above). We took all other emissions factors from the IPCC emissions factor database (IPCC, 2021). Additional operational data used to calculate carbon footprints relating to machinery diesel use were taken from the SAC Farm management handbook (SAC Consulting, 2022), fertiliser application rates from the John Nix Pocketbook for Farm Management (Redman, 2022), and pesticide application rates from Pesticide Usage survey reports (Ridley et al., 2021a, 2021b, 2020).

*Nutrient burdens*

The nutrient burdens calculated using our Excel-based tool were based on the eutrophication potential (from nitrate and phosphate leaching) and acidification potential (from sulphate leaching) of applied nutrients. We followed the *FAO guidelines for environmental quantification of nutrient flows and impact assessment* (FAO, 2017) for the calculating the eutrophication potential of nitrate and phosphate leaching and the acidification potential of ammonia emissions. We used emissions factors for ammonia ($NH_3$) emissions and nitrate ($NO_3^-$) leaching from the IPCC emissions factor database (IPCC, 2021). We used characterisation factors for calculating the eutrophication and acidification potential of the ammonia emissions and nitrate leaching from the Ecoinvent© database. The estimates of eutrophication and acidification potential also included estimates of embodied burdens from fertiliser production from the Ecoinvent© database.

### *Cost-effectiveness*

Using the financial costings and carbon footprints modelled with our Excel-based tool we calculated the cost effectiveness of each of the alternative management scenarios as a GHG mitigation option on both deep and shallow peat soils. We calculated the cost-effectiveness of each of the management scenarios as the reduction in GHG emissions per £1 of annual operational costs using the following formula:

*Cost-effectiveness (tonnes $CO_2$e/£) = $CO_2$e emissions reduction/(variable + fixed costs)*

The $CO_2$e emissions reduction was calculated as:

*$CO_2$e reduction = $CO_2$e emissions of the BAU scenario - $CO_2$e emissions of the alternative scenario.*

Note that these calculations effectively treat GHG mitigation as the only return on the operating costs incurred, and therefore need to be interpreted with caution in situations where these operating costs also generate other returns, such as income from crops or energy produced. A full cost-benefit analysis was beyond the scope of the current study but could be undertaken in future.

## Results

### *Financial Margins*

Our results suggest that on thick peat (> 100 cm), where BAU is based on high value horticulture (lettuces), reducing the water table depth (WTD) and shifting over to all of the alternative management scenarios would reduce gross and net margins (Table 6.1). Paludiculture options with *Miscanthus* (20 cm WTD) and common reeds (10 cm WTD) offer the lowest potential financial returns. While still generating lower financial returns than horticulture at a WTD of 100 cm, reducing the WTD to 30 cm and continuing with lettuce production could still (based on our assumptions regarding crop yields) generate a gross margin of around £40 k per ha and a net margin of around £28 k per ha.

***Table 6.1**. Gross and net margins (£ per hectare) of the alternative management scenarios assuming a peat depth of > 100 cm versus high value horticulture (lettuces) as the business-as-usual scenario.*

| Scenario | Gross Margin (£/ha) | Net Margin (£/ha) |
|---|---:|---:|
| 100 cm WTD + lettuces (BAU) | 58,663 | 47,268 |
| 50 cm WTD + wheat | 1,642 | 959 |
| 40 cm WTD + grass ley | 1,529 | 674 |
| 30 cm WTD + lettuces | 39,623 | 28,228 |
| 20 cm WTD + Miscanthus | 604 | 367 |
| 10 cm WTD + reeds | 294 | 104 |
| 5 cm WTD + pasture and solar | 2,196 | 1,983 |

Our results suggest that on areas of wasted peat (40 cm), where BAU is based on arable cropping (wheat), reducing the WTD and shifting to high value horticulture with lettuces (30 cm WTD) and permanent pasture with a solar lease (5 cm WTD) could increase gross and net margins (Table 2). Continuing with arable cropping or establishing an intensive grass ley at a reduced WTD are still profitable but would reduce financial returns compared to BAU on thin peat soils. Paludiculture options with *Miscanthus* (20 cm WTD) and common reeds (10 cm WTD) also offer the lowest potential financial returns for farmers on shallow peats.

***Table 6.2***. *Gross and net margins (£ per hectare) of the alternative management scenarios for a wasted peat of 40 cm thickness and arable cropping (wheat) as the business-as-usual scenario.*

| Scenario | Gross Margin (£/ha) | Net Margin (£/ha) |
|---|---:|---:|
| 100 cm WTD + wheat (BAU) | 1,936 | 1,443 |
| 50 cm WTD + wheat | 1,642 | 959 |
| 40 cm WTD + grass ley | 1,529 | 674 |
| 30 cm WTD + lettuces | 39,623 | 28,228 |
| 20 cm WTD + Miscanthus | 604 | 367 |
| 10 cm WTD + reeds | 294 | 104 |
| 5 cm WTD + pasture and solar | 2,196 | 1,983 |

## *Greenhouse Gas Balance*

All of the alternative management scenarios would reduce GHG emissions (on an area basis) compared to the BAU option (100 cm WTD), both for thick peats where high value horticulture is the assumed default enterprise option (Panel a in Figure 6.2) and for wasted peats where cereal cropping is the assumed default enterprise option (Panel b in Figure 6.2). The potential reductions in GHG emissions are greater for thick peats, at around 20 – 40 tonnes of $CO_2$e per hectare, compared to wasted peats where raising water levels only starts to reduce $CO_2$ emissions when WTD is less than the 40 cm of remaining peat.

The maximum potential GHG reductions for wasted peats were 20 tonnes of $CO_2$e per hectare under the best performing option, the 5 cm WTD + pasture and solar scenario (Panel b in Figure 6.2).

Breaking down the GHG balance into individual gases shows the relative contribution of each gas to the total emissions (Figure 6.3). Reducing the water table depth has a positive effect on both $CO_2$ and $N_2O$ emissions on both deep and shallow peats however the reduction in $CO_2$ emissions is higher on deep peat soils where the peat depth exceeds the effective water table depth (Panel a in Figure 6.3). On shallow peat soils, $CO_2$ emissions declined only when the effective water table depth was reduced to a point higher than the peat depth, i.e., the scenarios with a WTD < 40cm (Panel a in Figure 3). $N_2O$ emissions decline with an increase in WTD on both deep and shallow peat soils, however much less severely than $CO_2$ emissions (Panel a and b in Figure 6.3).

As expected $CH_4$ emissions peak in the scenarios (10cm WTD + reeds and 5 cm WTD + pasture and solar) where the WTD is raised closest to the surface (Panel a and b in Figure 6.3).

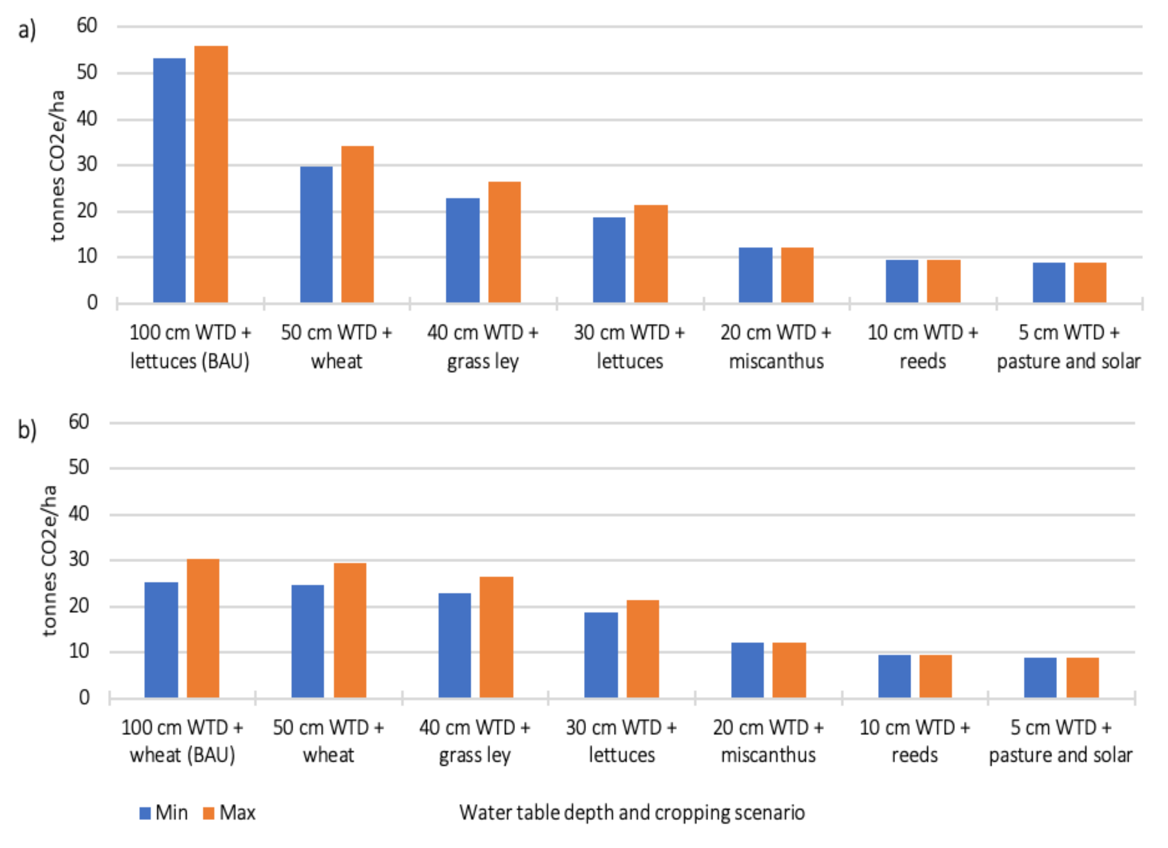



***Figure 6.2**. GHG emissions (t $CO_2$e per ha) of the alternative management scenarios assuming a) thick peat of > 100 cm with high value horticulture as the BAU scenario and b) 'wasted peat of 40 cm with arable cropping (wheat) as the BAU scenario. The GHG balance is the sum of $CO_2$, $N_2O$ and $CH_4$ from all sources, expressed in $CO_2$ equivalents.*

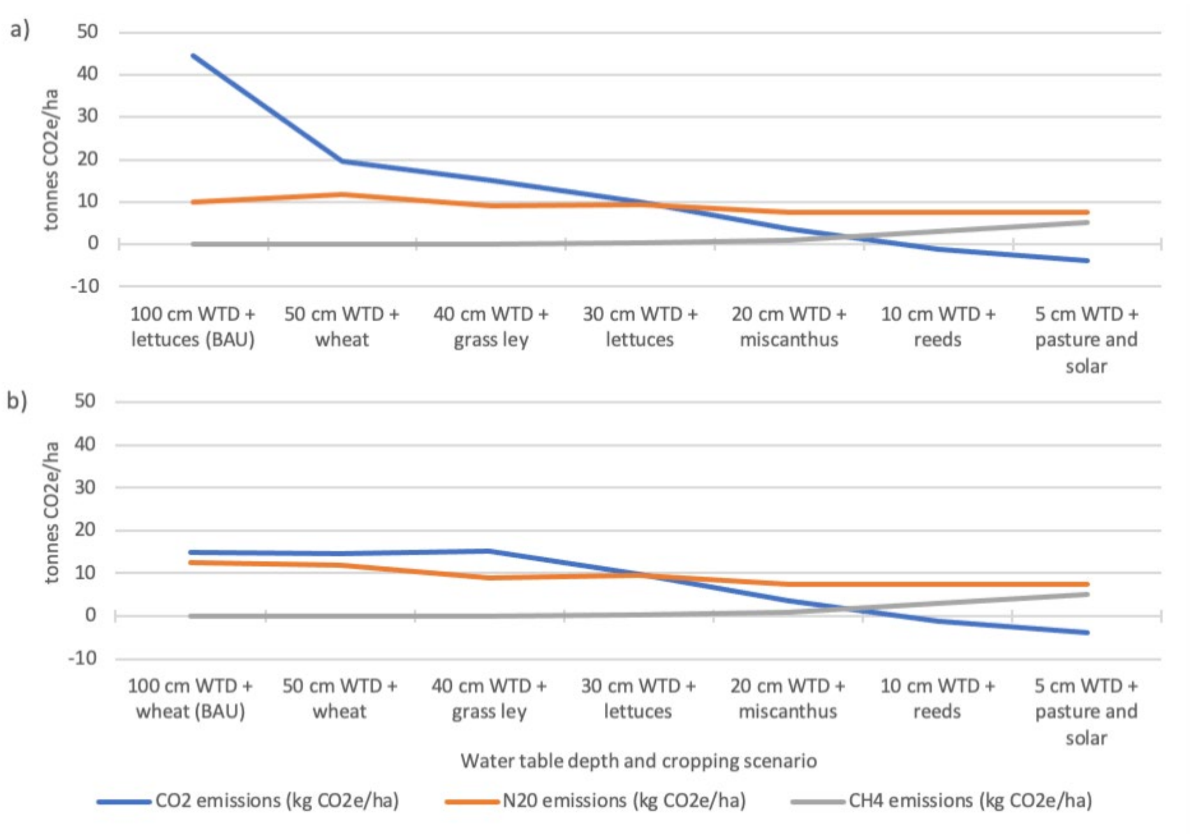



***Figure 6.3.*** *Individual Greenhouse gas emissions (expressed as tonnes $CO_2$e per ha) of the alternative management scenarios assuming a) thick peat of > 100 cm with high value horticulture as the BAU scenario and b) 'wasted peat of 40 cm with arable cropping (wheat) as the BAU scenario.*

### *Cost effectiveness*

Our results suggest that on thicker peat soils, raising the water table to 30 cm and continuing with high value horticulture is the least cost-effective way of reducing GHG emissions per hectare (6.3). Our modelling suggests the most cost-effective way of reducing GHG emission on deep peat would be to reduce the WTD to 5 cm and establish permanent non-grazed pasture with a solar-photovoltaic facility lease; for every £1 costs to run this option, emissions from thick peat soils could be reduced by 0.2 tonnes of $CO_2$e per hectare per year. Paludiculture options with *Miscanthus* or common reeds could also be cost-effective ways of reducing GHG emissions on deep peat soils. As noted above, however, this analysis treats the GHG mitigation achieved as (in effect) the sole return on operating costs, and so does not take account of any income generated from food or biomass crops, energy generated etc.

***Table 6.3.*** *Greenhouse gas reductions per £ of operating costs (fixed and variable) of alternative management scenarios on thick peat compared to lettuce production with a WTD of 100 cm as the BAU scenario.*

| Scenario | Total annual cost (£/ha/yr) | Reduction in GHG emissions compared to BAU (tonnes $CO_2e$/ha/yr) | | GHG reduction per £ of operating cost (tonnes $CO_2e$/£) | |
|---|---|---|---|---|---|
| | | Min | Max | Min | Max |
| 50 cm WTD + wheat | 1,453 | -23.47 | -18.81 | 0.016 | 0.013 |
| 40 cm WTD + grass ley | 1,441 | -30.23 | -26.59 | 0.021 | 0.018 |
| 30 cm WTD + lettuces | 28,892 | -34.34 | -31.82 | 0.001 | 0.001 |
| 20 cm WTD + Miscanthus | 543 | -40.96 | -40.95 | 0.075 | 0.075 |
| 10 cm WTD + reeds | 456 | -43.78 | -43.78 | 0.096 | 0.096 |
| 5 cm WTD + pasture and solar | 217 | -44.35 | -44.35 | 0.204 | 0.204 |

*-ve indicates reduction in emissions vs BAU and +ve indicates increase in emissions.*

On wasted peat soils (40 cm), raising the water table to 30 cm and converting to high value horticulture is the least cost-effective way of reducing GHG emissions per hectare, only saving 0.0002 tonnes of $CO_2e$ per hectare per £1 of annual operating costs (Table 6.4). Our modelling suggests the most cost-effective way of reducing GHG emission on shallow peat would be to reduce the WTD to 5cm and establish non-grazed permanent pasture with a solar-photovoltaic facility lease. Paludiculture options with *Miscanthus* or common reeds could also be very cost-effective ways of reducing GHG emissions on shallow peat soils. Again, this analysis does not take account of other financial returns on operating costs incurred to implement different management scenarios, and a full cost-benefit analysis would lead to different conclusions.

***Table 6.4.*** *Greenhouse gas reductions per £ of operating costs (fixed and variable) of alternative management scenarios on wasted (40 cm) peat compared to wheat production with a WTD of 100 cm as the BAU scenario.*

| Scenario | Total annual cost (£/ha/yr) | Reduction in GHG emissions compared to BAU (tonnes $CO_2e$/ha/yr) | | GHG reduction per £ of operating cost (tonnes $Co_2e$/£) | |
|---|---|---|---|---|---|
| | | Min | Max | Min | Max |
| 50 cm WTD + wheat | 1,453 | -0.38 | -0.89 | 0.0003 | 0.0006 |
| 40 cm WTD + grass ley | 1,441 | -2.23 | -3.76 | 0.0015 | 0.0026 |
| 30 cm WTD + lettuces | 28,892 | -6.34 | -8.99 | 0.0002 | 0.0003 |
| 20 cm WTD + miscanthus | 543 | -12.96 | -18.12 | 0.0239 | 0.0334 |
| 10 cm WTD + reeds | 456 | -15.78 | -20.95 | 0.0346 | 0.0460 |
| 5 cm WTD + pasture and solar | 217 | -16.35 | -21.52 | 0.0752 | 0.0990 |

*-ve indicates reduction in emissions vs BAU and +ve indicates increase in emissions*

*Nutrient burdens*

On thick peat soils where high-value horticulture was the assumed default BAU enterprise, our modelling suggests that raising the water table and implementing nutrient intensive cropping such as wheat (at 50 cm WTD) or intensive grass leys (at 40 cm WTD) could lead increases in nutrient leaching and acidification burden, expressed in terms of phosphate and sulphate leaching (Table 6.5).

The adoption of paludiculture crops with low additional fertiliser requirements such as *Miscanthus* (at 20 cm WTD) or common reeds (at 10 cm WTD) could eliminate nutrient and sulphate leaching. Raising the water table to 30 cm and continuing with lettuce production is likely to have no impact on either measure, assuming that fertiliser input requirements would be unchanged. There is some evidence to suggest that fertiliser requirements could increase under high water table management, because less nitrogen and phosphorus would be released from the peat as a result of organic matter oxidation, in which case it is possible that eutrophication risk could increase.

At this stage we lacked sufficient data to incorporate this in the model, but this could be done in future.

***Table 6.5.*** *Eutrophication and acidification potential of the alternative management scenarios on thick peat compared to lettuce production with a WTD of 100 cm as the BAU scenario.*

| Scenario | Eutrophication potential (kg $PO_4$/ha) | | Acidification potential (kg $SO_4$/ha) | |
|---|---|---|---|---|
| | Min | Max | Min | Max |
| 100 cm WTD + lettuces (BAU) | 10.79 | 10.98 | 7.81 | 8.67 |
| 50 cm WTD + wheat | 16.77 | 17.04 | 12.31 | 13.53 |
| 40 cm WTD + grass ley | 21.28 | 21.64 | 15.57 | 17.20 |
| 30 cm WTD + lettuces | 10.79 | 10.98 | 7.81 | 8.67 |
| 20 cm WTD + miscanthus | 0.00 | 0.00 | 0.00 | 0.00 |
| 10 cm WTD + reeds | 0.00 | 0.00 | 0.00 | 0.00 |
| 5 cm WTD + pasture and solar | 0.00 | 0.00 | 0.00 | 0.00 |

On shallow peat soils where arable cropping was the assumed default BAU enterprise, our modelling suggests that reducing the water table depth and shifting enterprise could lead to a reduced nutrient leaching and acidification burden (Table 6.6).

The exception would be the establishment of (non-grazed) intensive grass leys (at 40 cm WTD) which could lead to increases in leaching. As for thick peat soils, the adoption of paludiculture crops with low additional fertiliser requirements such as *Miscanthus* (at 20 cm WTD) or common reeds (at 10 cm WTD) would minimise nutrient and sulphate leaching.

***Table 6.6***. *Eutrophication and acidification potential of the alternative management scenarios on wasted (40 cm) peat compared to wheat production with a WTD of 100 cm as the BAU scenario.*

| Scenario | Eutrophication potential ($kg\ PO_4/ha$) | | Acidification potential ($kg\ SO_4/ha$) | |
|---|---|---|---|---|
| | **Min** | **Max** | **Min** | **Max** |
| 100 cm WTD + wheat (BAU) | 17.80 | 18.09 | 13.05 | 14.36 |
| 50 cm WTD + wheat | 16.77 | 17.04 | 12.31 | 13.53 |
| 40 cm WTD + grass ley | 21.28 | 21.64 | 15.57 | 17.20 |
| 30 cm WTD + lettuces | 10.79 | 10.98 | 7.81 | 8.67 |
| 20 cm WTD + miscanthus | 0.00 | 0.00 | 0.00 | 0.00 |
| 10 cm WTD + reeds | 0.00 | 0.00 | 0.00 | 0.00 |
| 5 cm WTD + pasture and solar | 0.00 | 0.00 | 0.00 | 0.00 |

## Summary

The modelling we carried out using the economic analysis and environmental footprinting tool developed for the project highlights some of the costs and benefits of different emissions mitigation strategies. The assessment suggests that while raising the water table to 30 cm and continuing (on thick peat) or adopting (on wasted peat) high value horticultural production may be the best option for farmers in lowland peat areas financially, it is not the best environmental performer in terms of emissions reductions and nutrient burdens on an areal basis. Our modelling suggests that reducing the WTD to 5 cm and establishing (non/minimally-grazed) permanent pasture with a solar-photovoltaic facility lease could be a viable option for balancing financial returns to farmers while also delivering significant cost-effective reductions in GHG emissions, although (as noted in the following section) there are to the best of our knowledge currently *no* solar farms on peat in which water levels have been raised, so this mitigation option remains theoretical. On wasted peat soils this management scenario would leave farmers financially better off than BAU (wheat), however on thicker peat with higher-value horticulture as the BAU counterfactual, it would lead to a reduction in income based on current prices and available financial incentives.

The model developed for this task should provide a useful basis for further development and assessment of different land-management and mitigation strategies. We acknowledge that it does not yet incorporate all of the potential costs of implementing the scenarios considered, including the impacts (positive or negative) of higher water tables on yields, fertiliser requirements or other input costs, or the costs of acquiring, storing and distributing water to facilitate different hydrological management regimes. To a large extent these costs are not fixed at a field level, as they depend on the wider configuration, elevation and internal topographic variation of fields within the wider farm landscape, drainage and water distribution network. Costs associated with risk of crop failure (e.g. due to spring flooding after planting) or disease were not accounted for. Our scenarios are also simplistic in that they consider a single crop rather than crop rotations, and do not include emissions from livestock, either present on the field of reliant on fodder harvested from the field, or ditch emissions. Conversely, we did not attempt to capture all of the direct or indirect financial benefits of different management option, for example (in addition to marketable products) the potential value of saleable carbon credits or other ecosystem services such as biodiversity gains, reductions in nutrient loadings to watercourses, or flood water storage potential. To some extent, these limitations reflect a lack of data with which to quantify the costs, benefits and wider impacts of different land-management and mitigation scenarios. On this basis, the following section provides a more comprehensive, but largely qualitative, assessment of the wider implications of management change on lowland peat.

## 6.3 Environmental impacts of mitigation measures

### Introduction

Based on the farmer consultation and largely following the economic assessment above, we identified a set of mitigation measures that were considered relatively feasible, and which would be expected to deliver some degree of emissions mitigation, relative to a 'business as usual' (BAU) counterfactual of conventional drainage-based agriculture. The assessment was repeated for thick peat and thin/wasted peat, with (as in Section 6.2) a BAU counterfactual of horticulture on thick peat, and cereal production on the thin/wasted peat. 'Thick' peat is generally defined as having a thickness of > 1 m, 'thin' peat as having a thickness of > 40 cm to 1 m, and 'wasted' peat a thickness of 40 cm or less. In practice, our classifications for 'thick' peat are applicable to any agriculturally drained peatland in which the water table remains largely or entirely within the peat layer throughout the year, whereas in 'thin' and 'wasted' peat, the water table is assumed to fall into the underlying mineral soil for some or all of the year. These soils are expected to show qualitatively similar responses to drainage and re-wetting (see Section 5) and were therefore combined in the analysis.

Mitigation measures considered ranged from modifications to existing agricultural practices (raised water levels), which would deliver relatively modest climate mitigation, through to full re-wetting and restoration. Solar power generation was considered as an alternative land-use based on both BAU drainage (typical of current solar farms) and full re-wetting (on the basis that solar farms do not necessarily require drainage, and indeed can be installed over water). Biomass production was considered as a plausible form of paludiculture, although as noted earlier this encompasses a broad range of options from reed-cutting for thatch through to intensive production of *Miscanthus* or willow for bioenergy or carbon capture. The same mitigation measures were applied to both peat categories. Key attributes and assumptions for each mitigation measure are summarised in Table 6.7. Note that our assessment is limited to areas of lowland peat that are currently under cropland management, in line with the overall focus of the project, and the resulting evidence available to inform this work. In future, as additional data become available, it would be beneficial to repeat the assessment for areas that are currently under grassland management.

***Table 6.7***. *Summary of management and mitigation options considered in this assessment*

| **Management option** | **Thick peat** | **Thin peat** | **Comments** |
|---|---|---|---|
| Horticulture - BAU | Y | N | Standard horticultural management including fertilisation and irrigation. Mean annual WTD > 50 cm |
| Cereals - BAU | N | Y | Standard arable management including fertilisation but limited irrigation. Mean annual WTD > 50 cm |
| Solar - BAU | Y | Y | Solar farm on conventionally drained peat (mean annual WTD > 50 cm) with low-intensity grazed grassland below solar panels |
| Horticulture - HWT | Y | N | Standard horticultural management including fertilisation and irrigation. Mean annual WTD 30-50 cm |
| Cereals - HWT | N | Y | Standard arable management including fertilisation but limited irrigation. Mean annual WTD 30-50 cm |
| Grass - HWT | Y | Y | Permanent or ley grassland managed with high water levels for hay production or pasture. Mean annual WTD 20-40 cm. |
| Biomass crops | Y | Y | Paludiculture-based production of native wetland species (e.g. *Phragmites, Typha*), water tolerant trees (e.g. short-rotation willow coppice) or non-native biomass crops (e.g. *Miscanthus*) to produce biomass for bioenergy or carbon capture (e.g. via biochar production). Mean annual WTD 5-25 cm |
| Solar - HWT | Y | Y | Solar farm on re-wetted peat (mean annual WTD 0-10 cm) with low-intensity grazed grassland or wetland species below solar panels |
| Restoration | Y | Y | Conservation-driven re-wetting and restoration of peatland to near-natural wetland cover (e.g. reedbed, carr woodland). Zero or low-intensity biomass removal, e.g. reed for thatch, |

Our approach to assessing environmental impacts of mitigation broadly followed an ecosystem services/natural capital approach, but focused on the key properties, functions and outputs of lowland peatlands, rather than imposing a rigid framework. We considered three broad categories. The first, termed detrimental impacts, comprises the main negative consequences (i.e. ecosystem disservices) of drainage-based peatland management such as GHG emissions, erosion, nutrient leaching to watercourses, subsidence, as well as the implications of both low and high water table management options for summer water demand. The second category includes a range of beneficial regulating services such as flow and water quality regulation, cultural services such as recreation and landscape aesthetics, the peatland carbon stock as a key natural capital metric, and biodiversity. Finally, we considered key provisioning services (food, fibre and energy production) and resulting consequences for farm incomes (see also Section 6.2). Impacts on farm incomes were limited to those derived from marketable products, rather than potential future income from carbon or natural capital markets, biodiversity net gain or other environmental benefits of mitigation measures.

Based on the available data we considered that it was not feasible to apply a fully quantitative approach to all metrics. Indeed, the evidence base for even the relative impacts of some mitigation measures remains weak for some metrics, as discussed below. Additionally, providing a fully comparable quantification of all ecosystem services/disservices requires a common unit of measurement, typically financial value, which is beyond the scope of the current project. A full financial accounting of the costs and benefits of different forms of peatland agriculture, including ecosystem services and disservices, was included in the Office for National Statistics Peatland Natural Capital Accounts (ONS, 2019, see Table 12). We anticipate that the new evidence generated during this project, and synthesised here, could inform a revision of the ONS peatland accounts in future.

## Results

The assessment of environmental impacts of mitigation for thick peat is shown in Table 6.8, and for thin/wasted peat in Table 6.9. Wherever possible we drew directly on the results obtained in other parts of the project including the review of paludiculture (Mulholland et al., 2020; Section 3), the assessment of the societal impacts of drainage (Page et al., 2020; Section 4), and results obtained from the flux tower network, irrigation trials and accompanying crop condition assessments (Section 5). The evidence base and methods used to classify each environmental impact are described below, along with a discussion of their implications and uncertainties.

***Table 6.8**. Environmental impacts of a range of current and potential future management options for remaining areas of thick lowland peat. Responses with a low level of confidence based on currently available evidence are shown in italics.*

| Management<br>WTD (cm) | Horticulture BAU<br>> 50 cm | Solar BAU<br>> 50 cm | Horticulture HWT<br>30-50 cm | Grass HWT<br>20-40 cm | Biomass crops<br>5-25 cm | Solar HWT<br>0-10 cm | Restoration<br>0-10 cm |
|---|---|---|---|---|---|---|---|
| $CO_2$ emission | V High | V High | High | Medium | Low | Zero/uptake | Zero/uptake |
| $N_2O$ emission | V high | Low | *V high* | Medium | Low | Zero | Zero |
| $CH_4$ emission (peat) | Zero | Zero | Zero | Zero/low | Low | Medium | High |
| $CH_4$ emission (ditches) | High | Medium | High | High | Medium | Medium | Medium |
| Wind erosion | V High | Low | V High | Low | Low | Low | Zero |
| Subsidence | V High | V High | High | Medium | Low | Zero | Zero |
| Eutrophication | *High* | Low | *High* | *High* | Zero | Low | Zero |
| Water demand | *High* | *Low* | *High* | *Medium* | *High* | *High* | *High* |
| Water storage | Low | Low | Medium | Medium | Medium | High | High |
| Flood attenuation | *Medium* | *Medium* | *Low* | *High* | *Medium* | *Medium* | *Medium* |
| Water quality regulation | *Low* | *Low* | *Low* | Medium | High | Medium | High |
| Carbon stock | Decreasing | Decreasing | Decreasing | Decreasing | Stable | Stable | *Increasing* |
| Biodiversity value | Low | Low | Low | Medium | Medium | *Low* | High |
| Recreation/tourism | Low | Low | Low | Medium | *Low* | *Low* | High |
| Aesthetic value | Medium | *Low* | Medium | High | Medium | *Low* | High |
| Food production (calorific) | Medium | Low | Medium | Medium | Zero | Low | Zero |
| Food production (value) | V High | Low | V High | High | Zero | Low | Zero |
| Fibre production | Zero | Zero | Zero | Zero | V High | Zero | Medium |
| Energy production | Low | V High | Low | Zero | High | V High | Zero |
| Farm income | V High | V High | High | Medium | Medium | V high | Low |

| Key | | | | | |
|---|---|---|---|---|---|
| ***Detrimental Impacts*** | Zero/low | Low | Medium | High | V high |
| ***Regulating and cultural services*** | | *Low* | Medium | High | |
| ***Provisioning services and economics*** | Zero | Low | Medium | High | V High |

***Table 6.9***. *Environmental impacts of a range of current and potential future management options for thin and wasted lowland peat. Responses with a low level of confidence based on currently available evidence are shown in italics.*

| **Management**<br>WTD (cm) | **Cereal BAU**<br>> 50 cm | **Solar BAU**<br>> 50 cm | **Cereal HWT**<br>30-50 cm | **Grass HWT**<br>20-40 cm | **Biomass crops**<br>10-30 cm | **Solar HWT**<br>0-10 cm | **Restoration**<br>0-10 cm |
|---|---|---|---|---|---|---|---|
| $CO_2$ emission | High | High | High | Medium | Low | Zero/uptake | Zero/uptake |
| $N_2O$ emission | High | Low | *High* | Medium | Low | Zero | Zero |
| $CH_4$ emission (peat) | Zero | Zero | Zero | Zero/low | Low | Medium | High |
| $CH_4$ emission (ditches) | High | Low | High | High | Medium | Medium | Medium |
| Wind erosion | High | Low | High | Low | Low | Low | Zero |
| Subsidence | Medium | Medium | Medium | Medium | Low | Zero | Zero |
| Eutrophication | *V High* | Low | *V High* | *High* | Zero | Low | Zero |
| Water demand | *Medium* | *Low* | *Medium* | *Medium* | *High* | *High* | *High* |
| Water storage | Low | Low | Low | Medium | Medium | Medium | Medium |
| Flood attenuation | *Medium* | *Medium* | *Low* | *High* | *Medium* | *Medium* | *Medium* |
| Water quality regulation | *Low* | *Low* | *Low* | Medium | High | Medium | High |
| Carbon stock | Decreasing | Decreasing | Decreasing | Decreasing | Stable | Stable | *Increasing* |
| Biodiversity value | *Low* | *Low* | *Low* | Medium | Medium | *Low* | High |
| Recreation/tourism | *Low* | *Low* | *Low* | Medium | *Low* | *Low* | High |
| Aesthetic value | Medium | *Low* | Medium | High | Medium | *Low* | High |
| Food production (calorific) | V High | Low | High | Medium | Zero | Low | Zero |
| Food production (value) | High | Low | Medium | High | Zero | Low | Zero |
| Fibre production | Zero | Zero | Zero | Zero | V High | Zero | Medium |
| Energy production | Medium | V High | Medium | Zero | High | V High | Zero |
| Farm income | High | V High | Medium | Medium | Medium | V high | Low |

| **Key** | | | | | |
|---|---|---|---|---|---|
| ***Detrimental Impacts*** | Zero/low | Low | Medium | High | V high |
| ***Regulating and cultural services*** | | *Low* | Medium | High | |
| ***Provisioning services and economics*** | Zero | Low | Medium | High | V High |

## $CO_2$ emissions

Relative $CO_2$ emissions for each land-management option were based on the empirical function derived from the UK flux tower synthesis published by Evans et al. (2021a) and updated with new data collected during the project as described in Section 5.5. This function predicts $CO_2$ fluxes as a linear function of effective water table depth (WTD*e*), i.e. whichever is the shallower out of the peat depth and the true water table depth. Fluxes are predicted to transition from negative (i.e. net $CO_2$ uptake) when the water table is within 10 cm of the peat surface, to increasingly positive values (net $CO_2$ emission) as $WTD_e$ increases. Note that the proposed non-linear relationship between $CO_2$ and WTDe described in Section 5.5 does not affect the relative emission classifications used in this qualitative assessment.

For the thick peat analysis in Table 6.8, $CO_2$ emissions decrease progressively from very high values under BAU horticulture or solar, through to near-zero values or net $CO_2$ uptake for the mitigation measures that involve the greatest amount of re-wetting. Horticulture or grassland management with higher water levels (HWL) offer some mitigation versus BAU levels, but emissions are expected to remain moderate to high based on the indicative annual mean WTD values applied. We assumed that wetland-adapted biomass crops could tolerate higher water levels but might require slightly greater drainage than HWL solar or full restoration to enable high growth rates. As a result, some $CO_2$ emissions remain likely, although these could be offset if harvested biomass is used for carbon storage, for example through conversion to biochar. As noted above we assumed that solar farms could (at least in theory) be operated with very high water levels, which might be sufficient to effectively halt $CO_2$ emissions. However net sequestration appears unlikely unless the land below the solar panels is managed in a way that can support peat-forming wetland species. Full re-wetting and restoration could enable new peat formation to occur, leading to $CO_2$ sequestration, although in practice the outcomes of restoration of former agricultural peatlands remain uncertain, with some risk of continued $CO_2$ emissions depending on the effectiveness of re-wetting, and degree to which wetland species are able to re-establish on physically modified and nutrient-enriched soils. Note that our assessment assumes that on average the water table remains below the peat surface, i.e. that large-scale inundation does not occur, which could have differing (and potentially worse) outcomes for GHG emissions due to high $CH_4$ emissions.

For thin/wasted peat (Table 6.9), rates of $CO_2$ emission are limited by the depth of the remaining peat layer, leading to the lower Tier 2 emission factor for cropland on wasted peat developed as part of the BEIS wasted peat project. As a result, additional drainage below the base of the peat will not further increase emissions from the peat (some $CO_2$ loss from drained mineral subsoils is possible, but expected to be smaller than that for highly organic peat soils). Conversely, raising water levels towards but not into the peat layer is unlikely to offer significant emissions mitigation. On this basis, $CO_2$ emissions were classed as 'high' (but not 'very high') for cereal crops on thin/wasted peat under BAU management, as well as for solar BAU. Cereals on thin/wasted peat with higher water levels were also classed as having high $CO_2$ emissions; these emissions are likely to be lower than those from cereals under BAU water level management, as demonstrated by new data from the Stretham high water table trial (Section 5.3), but may be unchanged if the water table is not raised to within the peat layer

### $N_2O$ emissions

Emissions of $N_2O$ show a high degree of spatial and temporal variability, are difficult to measure, and are affected by multiple environmental variables including soil moisture content (with maximum emissions associated with intermediate moisture levels), mineral nitrogen availability, the availability of organic substrates for microbial activity, and temperature. As such, predictions of $N_2O$ emissions as a function of land-management are intrinsically uncertain, and there are no simple empirically based response functions available. However, published $N_2O$ data collated for the UK National Atmospheric Emissions Inventory and Peatland Code (Evans et al., 2022a) shows clear differences in average emissions for key lowland peat categories in the order cropland > intensive grassland > extensive grassland, while $N_2O$ emissions from rewetted fen are considered to be zero. The emission factor dataset also suggests a noisy positive relationship between $N_2O$ emissions and drainage depth, with higher emissions tending to occur when WTD > 20 cm (Evans et al. 2023, Figure 3.9). While this might suggest that raising water levels under agricultural land could reduce $N_2O$ emissions, the results from the Skyline 2D experiment (Section 5.2) suggest that increasing the moisture content of (but not saturating) the topsoil via irrigation could increase $N_2O$ emissions. Results of eddy covariance $N_2O$ measurement at Stretham for the BEIS wasted peat project also suggest very high $N_2O$ emissions when the field was being used to grow potatoes, which involved a high level of irrigation and fertilisation. On this basis, we categorised $N_2O$ emissions as very high for both BAU and HWL horticulture, but with greater uncertainty for the higher water table option. $N_2O$ emissions were estimated to be medium for grassland (likely varying as a function of management intensity and the presence or absence of livestock), low for biomass crops and BAU solar, and near-zero for restoration and HWL solar.

For thin/wasted peat, we assumed similar rates of $N_2O$ emission for all categories except BAU and HWL cereals, which were considered to have 'high' emissions rather than the 'very high' emissions predicted for the horticulture on thick peat. This is primarily based on the fourfold lower $N_2O$ fluxes measured at Stretham under a wheat crop, compared to the potato crop in the preceding year, which we tentatively attribute to lower rates of fertilisation (limiting the supply of nitrate for denitrification to $N_2O$) and irrigation (leading to lower soil moisture levels, that are less favourable for $N_2O$ production).

### $CH_4$ emissions

To predict $CH_4$ emissions from the peat surface we applied the non-linear relationship between $CH_4$ fluxes and WTD derived by Evans et al. (2021a), partly based on previous research for Defra (Evans et al., 2016). This relationship suggests that $CH_4$ emissions from the peat itself will be negligible under any form of land management for which average WTD exceeds 20-30 cm. As the water table is raised to within around 10 cm of the peat surface, higher emissions are possible, although the magnitude of these emissions will likely depend on additional factors including the vegetation types present, notably the presence of wetland-adapted aerenchymatous, or 'shunt', species that transport $CH_4$ to the atmosphere via their stems, as well as the presence of areas of standing water (such as shallow pools in restored wetlands) and nutrient levels. As a result, the risk of elevated $CH_4$ emissions is highest from restored wetlands, although provided that large-scale permanent inundation does not occur the climate impact of higher $CH_4$ is unlikely to cancel out the benefits of reduced $CO_2$ emissions; even in a 'worst case' scenario involving extensive areas of eutrophic standing water, it is highly unlikely that $CH_4$ emissions would reach an equivalent level to the very high $CO_2$ emissions that result from deep-drained horticulture on thick peat.

We also considered $CH_4$ emissions from ditches, which were assumed to be present for all land-management categories. Ditch networks are typically used to transport water into restored fen wetlands, so remain an important component within restored peatland eocsystems. Data on $CH_4$ emissions from ditches remain sparse, but can be a major source of landscape-scale emissions, especially where ditches are nutrient-enriched and ditch densities are high (Peacock et al., 2021). On this basis, we categorised ditch $CH_4$ emissions as high from all forms of conventional agriculture (whether managed with high or low water levels), and medium from all other management options. Mitigation of ditch $CH_4$ emissions may be possible, for example by reducing nutrient loadings or removing eutrophic sediments, but this remains a challenge, for example high ditch $CH_4$ emissions were reported at the undrained Wicken Fen, possibly due to the effects of using eutrophic river water to maintain water levels (Peacock et al., 2017).

Given that $CH_4$ emissions from the peat surface are only observed when the water table is within 20-30 cm of the peat surface, we considered that emissions from any given management were likely to be the same regardless of whether the peat was thick or thin/wasted. There is some evidence that ditch $CH_4$ emissions may be lower from areas of thinner peat, in part simply because ditch networks tend to be less dense in these areas (Evans et al., 2017) but in the absence of more data we assigned the same ditch emission categories to thin/wasted peat as for the equivalent management option on thick peat.

Note that we did not account for $CH_4$ emissions from livestock in our assessment. These should be considered for any form of management that involves ruminant animals, notably grassland (whether present on the field or receiving fodder grown harvested from it) but is beyond the scope of the current assessment.

**Wind erosion**

Wind erosion fluxes have been measured at several sites in the Fens, as part of research projects linked to the Defra lowland peat programme (Cumming 2018; Newman 2022). As described in Section 4 and in the societal impacts report (Page et al., 2020), wind erosion risk is greatest where bare peat is exposed by farming operations during dry periods, leading to so-called 'fen blows', which can contribute significantly to overall C loss from individual fields. This risk is very high from thick peat (which tends to have the lowest bulk density), and from horticultural management which can leave field surfaces exposed for long periods, as well as during maize cultivation where fields are often poorly covered during the spring months. This risk can be partly mitigated via the use of shelter belts and cover crops. Any form of management which maintains a continuous or near-continuous vegetation cover should reduce wind erosion risk to low or zero, with any remaining risks associated with periods of vegetation or soil disturbance, such as biomass harvesting.

Thin and wasted peats tend to have a higher bulk density than remaining areas of thick peat, which may reduce their vulnerability to wind erosion. If cereal crops are on the field for longer than short-lived vegetable crops, or involve less disturbance of the soil (e.g. due to single rather than double-cropping within a year) then this could also reduce erosion risk. However, arable fields on thinner peat tend to be larger than horticultural fields on thick peat, with fewer shelter belts, increasing the erosion risk. Overall, we classed wind erosion risk as 'high' rather than 'very high' for both BAU and HWT cereals on thin/wasted peat

**Subsidence**

As discussed in the societal impacts report (Page et al., 2020) and summarised in Section 4, peatland subsidence is intrinsically linked to drainage and carbon loss. As such, predicted subsidence for each land-management option largely tracks predicted $CO_2$ emissions, with maximal rates for BAU horticulture on thick peat, reducing to near-zero values following restoration. Because subsidence tends to be highest for lower-density peat with a high organic carbon content, and only affects the remaining organic layer, predicted rates are generally higher for drained land-use categories on thick peat. As peat wastage progresses, increasing bulk density, declining organic carbon content and (until all organic matter is contained within the plough layer) a decreasing thickness of peat all result in slowing rates of subsidence. On this basis, we assigned a 'medium' rate of subsidence to deeper-drained land-use categories on thin and wasted peat.

## Eutrophication

We categorised eutrophication risk (i.e. risk of nutrient leaching to watercourses) based on the economic analysis model developed in Section 6.2, which was based on rates of estimated fertiliser usage. This analysis suggested a higher risk of nutrient leaching from cereals (wheat) and ley grasslands, with lower leaching from the vegetable crop (lettuce). It also suggested little or no reduction in leaching as a result of higher water levels, although the model did not incorporate possible effects of changing water levels on nutrient cycling and transport. Given that vegetable crops vary in their fertiliser requirements, and that changes in water level and associated redox status likely do alter nutrient dynamics, these results may not be generalisable, but clearly the risk of eutrophication will be higher for any form of agriculture requiring fertilisation. As an added complication, oxidising peat releases nitrogen and phosphorus via mineralisation of organic matter, which effectively reduces fertiliser demand from deeper, rapidly decomposing peat. Conversely, as peat wastage progresses and oxidation rates slow, the need for additional fertiliser is likely to increase, with a corresponding increase in eutrophication risk. This might also be the case for HWL management if it requires greater fertiliser application rates to maintain crop yields. On this basis, we classified the risk of eutrophication as 'very high' for both BAU and HWL cereal production on thin/wasted peat, and 'high' for the other agricultural management options, with a high associated uncertainty in all cases. Solar farms were assigned a low eutrophication risk based on the assumption that no fertiliser would be applied. Land managed for biomass production was assumed have a near-zero eutrophication risk, because biomass crops are likely to capture and retain all available nutrients. Similarly, restored peatlands are expected to be highly effective at retaining available nutrients.

## Water demand

Water demand was considered in terms of the anticipated requirements for water to support each form of land-management during summer, when water availability is lowest. Although it is often assumed that peatland re-wetting will require additional water input and therefore contribute to regional water scarcity, this is not inevitably the case; in the absence of continued pumped drainage to higher-elevation river systems, many of England's lowland peatlands would rapidly and permanently flood as a result of rainwater inputs alone. However, the hydrological disconnection of many remaining wetlands from their original water sources, such as groundwater seepage from adjacent uplands in areas such as the Norfolk Broads, together with the loss of normal floodplain functioning in other areas, means that maintaining higher water levels can be difficult. Furthermore, raising water levels in small areas of otherwise drained landscapes can be hard to achieve due to the degree to which artificial drainage networks (managed by Internal Drainage Boards and the Environment Agency) connect with adjacent areas of farmland. These issues of landscape-scale water management have been considered in depth by the Defra Lowland Agricultural Peat Task Force, and we do not wish to duplicate that work here. We therefore focus on the field-scale water demand required to maintain each of the land-management options considered, which largely relate to 1) the amount of water that needs to be added to support the crop and associated water levels (i.e. irrigation) and 2) the rate of water loss that will occur from the field as a result of this irrigation, and in the absence of active drainage, primarily via evapotranspiration.

As described in Section 5.3, the flux tower network provides a unique dataset of directly measured evapotranspiration (ET) from sites ranging from near-natural fen to intensive arable and horticulture. Of the sites for which data are available, the reedbed at Wicken Fen had the highest actual ET, of around 800 mm $yr^{-1}$, but in general we did not observe strong distinctions between sites as a function of vegetation/crop type or peat depth; the Anglesey Fen wetland sites had lower ET (albeit they are located in a cooler area), while ET rates approaching that of Wicken Fen were observed from deep-drained arable and horticultural sites on multiple occasions. Given the high water demand and resulting irrigation requirements of many horticultural crops in particular, this result is not surprising. There is some suggestion from the flux tower data that areas of thicker peat such as Rosedene, which can be irrigated using subsurface drains, may have lower ET rates than sites such as Redmere and Stretham which require surface irrigation. However, it is clear that further work is needed on water demand of different land-management options, and we therefore assign a low confidence to all predictions for this category. Overall, we consider that water demand is likely to be high for horticultural crops due to their high irrigation demand, and for re-wetted land-use categories due to the requirement for supplementary water to maintain high water levels during dry periods (although this demand would likely decrease as the scale of re-wetting increases). Water demand for cereals and grass are likely somewhat lower, in that these crops are more able to withstand drought, and low for BAU solar.

### Water storage

The water storage capacity of different land-management classes is considered here in terms of the total volume of water they can hold, and their resulting capacity to act as reservoirs that could theoretically release water during dry periods. Water storage capacity is effectively determined by the height of the water table, and the water holding capacity of the soil below the water table, and is therefore highest in re-wetted thick peat, which has a relatively high porosity. Conversely it will be lowest in deep-drained thin and wasted peat, where the water table is within the lower-porosity mineral soil.

### Flood attenuation

The capacity of different peatlands to reduce flood risk is not fully understood, and depends on factors such as geographic location as well as land-management *per se*. At present, water levels in many areas of lowland agricultural peat are lowered in winter to create 'freeboard' (i.e. short-term water storage capacity) within the field and ditch network to storage flood overflows from higher-level rivers. In other areas, wetlands and grasslands perform a water storage function by holding excess flood water, either as natural floodplains or engineered 'washlands'. Grasslands in areas like the Somerset Levels and Moors are often subject to large-scale inundation due to runoff from upland areas, which provides a service in terms of flood protection but can have detrimental impacts on land managed for agriculture. Furthermore, once land has become saturated or inundated it no longer offers any further flood protection function, and indeed may act as (in effect) an impermeable surface, exacerbating flood risk during subsequent rain events.

Overall, given the complexity and geographical variability of processes that determine flood risk it is difficult to generalise, and all classifications for this category have low confidence. We considered that conventionally drained cropland offered 'medium' flood protection, which was reduced to 'low' if water levels were raised. Grasslands in some settings such as washlands and connected floodplains can offer a high degree of flood attenuation, while land under high water-level management also offers 'medium' protection on the basis that it can withstand flooding, but may offer limited additional water storage potential.

### Water quality regulation

Water quality regulation is considered separately from the impact of management on eutrophication, as it relates to the wider capacity of the peatland system to retain pollutants, although the two clearly are linked. In general, drained peatlands have relatively little capacity to retain external pollutant inputs, whereas those with high water levels may be able to retain and store pollutants via natural processes of vegetation uptake and peat formation (see also Section 5.1 and Table 5.3 of Mulholland et al., 2020). Peatlands managed for biomass production under paludiculture-type conditions are likely to be particularly efficient at regulating water quality due to the high uptake and periodic removal of nutrients in the biomass, and may even be managed for this purpose, as is already the case for constructed water treatment wetlands. Peatland drainage can also expose sulphide-enriched subsoils to oxidation, resulting in highly acidic runoff enriched with sulphate, iron (Fe) and other metals. The subsequent oxidation of soluble $Fe^{2+}$ to insoluble $Fe^{3+}$ leads to the formation of ochre, an opaque orange substance which can block subsurface drains and have a highly deleterious impact on downstream ecosystems. Overall, we classified the water quality regulation function of all croplands and BAU solar as 'low', grassland and HWL solar as medium, and HWL biomass production and restoration as 'high'.

Note that, whilst dissolved organic carbon (DOC) is a significant issue for drinking water supplies derived from upland bogs, it is typically present in lower concentrations in runoff from lowland fen peat, and this runoff is rarely used for drinking water supply. We therefore omitted DOC from our assessment.

### Carbon stock

Carbon stock is one of the most widely reported natural capital metrics, and is notably high in all peatlands. However, a large and stable carbon stock has a limited environmental relevance, whereas a peatland which is losing carbon will be a $CO_2$ emission source, and one which is accumulating carbon will be a $CO_2$ sink. We therefore categorised carbon stocks as decreasing, stable or increasing, based on their estimated $CO_2$ emissions. On this basis all forms of drainage-based land-use (in both thick and thin/wasted peat) were considered likely to have decreasing C stocks, while HWL biomass production and solar were expected to have fairly stable C stocks. Some restoration projects may result in an increasing C stock, but this outcome is uncertain and depends on the effectiveness of the restoration.

### Biodiversity

There is no single metric of biodiversity that is applicable to all ecosystems, and some commonly used metrics such as overall species numbers may not be appropriate for peatlands, which (particularly in the case of bogs) tend to support a relatively small number of specialised – and consequently rare – species, adapted to live in waterlogged environments. Fens are typically more species-diverse than bogs, and remaining areas are often critically important habitats for some bird species such as waders. As a result, restoration was considered to result in high biodiversity, whereas drained monoculture crops and solar farms were considered to have low biodiversity (although this need not necessarily be the case for solar, depending on the vegetation grown beneath the panels). Moderately drained permanent grasslands can also support biodiverse flora, although this is not the case for more intensively managed or rotational grasslands, so overall this category was assigned a medium score. In line with Table 5.3 of Mulholland et al. (2020), biomass crops are also assigned a medium score, despite also typically being monoculture crops, because habitats such as reedbeds and coppice woodland can provide important habitat for some species. Raising ditch levels may also confer biodiversity benefits.

### Recreation and tourism

Mulholland et al. (2020) compared the recreational and tourism value of BAU and HWL cropland with paludiculture and restoration, along with education values which were assigned the same rankings. Restoration was expected to result in the highest recreational values, in line with the previous assessment of Bonn et al. (2010) for Thorne and Hatfield Moors, whilst cropland was considered to have the lowest values. For the extended range of land-management options included here we assigned low values to cropland and solar (regardless of water table management or peat thickness), medium values to grasslands, and high values to restored fen peatlands. We recognise that this is a simplification, with some areas of farmland having higher recreational values (e.g. where they support ecotourism activities) and some areas of conservation-managed wetland having restricted public access, and therefore a low value.

### Aesthetic values

As recognised by Mulholland et al. (2020), the aesthetic value of a landscape is highly subjective, and will vary from one person to another. We assigned the lowest values to solar farms, as these new and relatively industrialised forms of land-management are often considered controversial, while cropland was assigned a 'medium' score reflecting the cultural significance of farmed landscapes. Arguably the smaller fields and more frequent shelter belts in areas of remaining thick peat could be considered to have a higher landscape value than the large 'prairie' fields characteristic of thin and wasted peats, however we did not separate these in our classification. We also scored HWL biomass production as 'medium'. Restored wetlands and grasslands were classed as 'high'.

### Food production

We scored the provision of food on two metrics, calorific value and financial value. Financial values were derived from the economic analysis, and also refer to the analysis by ONS (2019), and to an ongoing review of vegetable production on organic soils being undertaken by UKCEH and NIAB for the WWF/Tesco Partnership (Rhymes et al., in prep). For HWL cereal management we also took account of the yield and crop health data obtained from the HWL wheat trial at Stretham, which suggested around a 25% reduction in yield, lower crop quality on some metrics and some evidence increased disease occurrence. For HWL horticulture we do not have direct data on yield and condition of crops grown under continuously high water levels, however the Skyline 2D irrigation trial did not suggest any yield reduction or increased disease incidence as a result of intensive water addition, in fact marketable yields increased. In a mesocosm experiment, Matysek et al. (2022) found that raising water levels from 50 to 30 cm below the surface did reduce lettuce yields, by around 30%, whereas a similar study with celery produced only a 19% yield reduction (Matysek et al., 2019) and one with radish showed a yield enhancement (Musarika et al., 2017). Therefore, the response of horticultural crops to water level changes may to some extent be crop-dependent, but on average we assumed that some reduction would occur. Calorific values were taken from literature information together with harvest yield and dry weight data obtained at the flux tower sites throughout the project.

Based on these data, as well as the ONS (2019) data, we assigned a 'very high' financial value to BAU horticultural production on thick peat and considered it likely that this could be maintained under (at least) moderately higher water levels. On the other hand, many horticultural products such as salad crops have very low dry weights and associated calorific value, and were therefore scored 'medium' overall on this metric. BAU cereals on thin/wasted peat was assigned a 'high' score in terms of financial return, but a 'very high' score on calorific value. As a result of lower yields measured under higher water levels we reduced these scores to 'medium' and 'high' respectively for HWL cereals, although data from additional sites, and in less climatically extreme years, are needed to provide more robust evidence. Grasslands were assigned a 'high' score in terms of financial value, assuming that they are used for intensive meat and dairy production, but a medium score in terms of calorific value given that livestock farming is a relatively inefficient means of producing food in general. Solar farms were scored 'low' on both metrics (assumed limited to low-intensity grazing, e.g. by sheep, beneath the solar panels) and HWL biomass production and restoration were both scored 'zero' on the basis that these areas are no longer managed for food production.

### Fibre production

We assigned zero scores for fibre production to all land managed for food production, as well as solar. It is possible that some crop residues could be used for fibre, but this is not common practice. Land that is actively managed for biomass production was scored 'very high', as fibre production (whether for bioenergy, carbon capture or other uses such as building materials) is the primary aim of land-management in these areas. Some restored areas can be managed to produce reeds or other forms of marketable biomass, so this land-management category was scored 'medium', although in some restored areas little or no off-site removal of harvested biomass (as opposed to on-site conservation management) takes place.

### Energy production

Solar farms on peat were scored 'very high' for energy generation. HWL biomass was also scored 'high' on the basis that water-tolerant biomass crops such as short-rotation coppice willow and *Miscanthus* are among the main UK-grown feedstocks for bioenergy production. We assumed that biomass produced in grasslands and restored wetlands was unlikely to be used for energy generation, and therefore assigned them a 'zero' value. Around 10% of the cereal production area on lowland peat is cultivated for maize (Rhymes et al., in prep.), most of which is believed to be used to produce biogas in anaerobic digesters. Maize cultivation for bioenergy on drained lowland peat is almost certainly detrimental in terms of its net climate impact, but nevertheless scored as 'medium' for both BAU and HWL cereal production on thin/wasted peat (acknowledging that energy yields may be somewhat lower for the HWL scenario as discussed above). Energy production from horticulture is limited to the potential use of crop residues in anaerobic digesters and was therefore scored 'low'.

### Farm income

Farm incomes under different land-management scenarios were evaluated in the preceding section, so we do not discuss them in detail here. Overall, we considered that incomes were likely to be highest for BAU horticulture on thick peat and solar farms, followed by BAU cereals on thin/wasted peat. Incomes were expected to decrease as a result of HWL cropland management, conversion to grassland, or implementation of HWL biomass production. Restoration to wetland was assumed to generate a low level of income from a farming perspective, although potential alternative income streams such as eco-tourism were not considered. Additionally, and in contrast to the ONS (2019) assessment, we did not include income from public subsidies such as agri-environment schemes in our calculations, which would generally favour more conservation-oriented management. On the other hand, subsidies embedded in the prices received by farmers for products (e.g. for renewable energy production) are included.

### Discussion

This assessment provides a qualitative overview of the environmental, societal and financial costs and benefits of alternative land-management options for lowland peatlands. Many of these costs and benefits are difficult to define or value, and in some cases evidence for the magnitude and even the direction of responses to a given land-management change remains weak. On this basis, a fully quantitative assessment would be difficult, and potentially misleading. Instead, we have attempted to link our qualitative rankings for individual environmental impacts and ecosystem services to empirical evidence, including new data collected during the project, wherever possible.

Overall, it is clear that there are no 'easy wins' for lowland peat management. Drainage-based farming has a broad range of detrimental environmental impacts including high GHG emissions, subsidence, erosion and nutrient leaching. It provides limited regulatory and cultural ecosystem services, and supports limited biodiversity. However lowland peat farming, particularly for crops, makes a nationally significant contribution to food supplies in terms of both calorific and financial value, reduces reliance on food imports, and supports the rural economy. Agriculture remains the dominant source of income from the land over the majority of lowland peat landscapes.

Solar farms offer comparable levels of farm income under current economic models, and contribute to national energy security and renewable energy targets, but currently at the expense of taking land out of food production. The economic analysis also assumes that farmers would earn income by leasing land for solar, rather than investing in the (considerable) capital costs themselves, which would result in a different conclusion. Solar farms also have a low (arguably negative) value in relation to cultural services such as recreation and landscape aesthetics. In addition, most if not all solar farms currently installed on lowland peatlands are located within agricultural landscapes, and are subject to BAU drainage, meaning that the peat beneath them is likely to be continuing to emit $CO_2$ at almost the same rate as adjacent farmland. There may be some reductions in $CO_2$ emissions due to the cooling effect of shading of soils by solar panels, and $N_2O$ emissions will likely be lower in the absence of fertilisation, but these reductions are likely to be marginal and must be considered in terms of the local biophysical (e.g. localised heating) impacts of large solar arrays. To the best of our knowledge, and bearing in mind that solar farms can be installed on the surface of lakes and reservoirs, there is no fundamental reason why solar farms could not be operated on partly or fully re-wetted peat, thereby reducing or halting $CO_2$ emissions. However, we are unaware of any solar farms currently operating on re-wetted peat, and in practice this would be difficult to achieve in situations where solar panels are being installed on individual fields within drained agricultural landscapes. A more planned approach comprising larger solar farms occupying contiguous areas, in which water levels could be managed independently of adjacent farmland, would be more effective from a climate mitigation perspective, but potentially controversial in terms of landscape impact. Establishing HWL solar farms as 'wet buffers' around areas of conserved or re-wetted fen would also offer an effective overall mitigation option, but again would likely have an impact on adjacent areas of high landscape and biodiversity value.

Apart from the (still theoretical) option of HWL solar, all other mitigation options would involve a decrease in provisioning services and farm incomes. Higher water level grassland management could be considered as a 'halfway house' between deep-drained cropland and full re-wetting, permitting continued productive use of the land with reduced GHG emissions. Indeed, many areas of lowland peat, including the Somerset Levels and Moors, Norfolk Broads and Lancashire Mosses, already support large areas of grassland, typically with higher water levels and correspondingly lower $CO_2$ emissions than croplands. However, increases in grassland extent and associated livestock numbers would be expected to lead to an increase in ruminant $CH_4$ emissions, as well as a potential increase in $N_2O$ emissions from urine patches, which were not accounted for in our assessment of soil-derived emissions, as they often occur away from the peatland itself. As a result, the overall emissions mitigation may be smaller than our field-scale assessment suggests. Furthermore, the UK's Net Zero Strategy, reflected in the 6$^{th}$ Carbon Budget (CCC, 2021) incorporates land-use scenarios based on anticipated dietary changes away from meat and dairy consumption. This implies a decreased requirement for grassland agriculture, freeing up some of this land for other land-uses such as afforestation, habitat restoration and biomass production for bioenergy, carbon capture and storage (Simon et al., 2021). In this context, expanding the area of drained grassland on lowland peat does not seem to be a viable mitigation strategy at a UK scale.

The remaining mitigation options, HWL biomass production and restoration, are considered effective in terms of climate change mitigation, as well as reducing other detrimental impacts such as subsidence and freshwater eutrophication. Mitigating GHG emissions via re-wetting and restoration, and to an extent also HWL via biomass production, would provide multiple ecosystem service co-benefits related to water quality and flow regulation, enhanced cultural values and higher biodiversity. Higher summer water demand may be a constraint on raising water levels, particularly for isolated areas within otherwise drained landscapes, however our data suggest that habitats managed with high water levels may not necessarily higher water demand relative to current agricultural practices. As a result, and as for solar farms, raising water levels over large, hydrologically contiguous areas may be more water-efficient then the fragmentary re-wetting of small areas.

As noted above, direct income to farming-based businesses from restored peatlands is typically low. Markets for biomass produced through paludiculture are either fairly small-scale (e.g. reed for thatch) or at an early stage of development (e.g. biomass for bioenergy or carbon capture, building materials such as fibre board) (Mulholland et al., 2021). Implementing high water level management at scale will therefore require additional or alternative income streams. Public funding, for example via ELMS or the Nature for Climate Fund, can support restoration and mitigation activities but tends to fund the cost of undertaking restoration works, or income forgone. In themselves, they do not necessarily provide a viable alternative business model, particularly in comparison to highly profitable cropland agriculture. There is some scope to address this financial imbalance by reducing current subsidies for unsustainable activities, such as cultivating environmentally-damaging maize on peat for biofuels. Area-based subsidies that pay higher rates for high-grade farmland than for the same land under wetland management also act as a barrier to change. Even with changes to current funding mechanisms, however, public funds alone may be insufficient to achieve climate change mitigation at the scale required to meet the targets in the 6$^{th}$ Carbon Budget, given the high profitability of existing farming systems. As a result, there would be benefits in aligning private sources of finance to support climate mitigation and wider environmental restoration measures in lowland peatlands. This could occur via the purchase of carbon credits for emissions reductions or carbon capture (e.g. via the Peatland Code, which now incorporates lowland fen peatlands), or via regulatory offsetting mechanisms such as Biodiversity Net Gain and Nutrient Neutrality (provided that peatland restoration is eligible in achieving the requirements of these schemes). Various mechanisms are also being developed to 'stack' payments for multiple ecosystem service benefits, such as the Landscape Enterprise Networks scheme. Price premiums paid by customers such as the supermarkets for more sustainably produced food may also help to drive change, although raising prices for consumers is likely to be difficult in the current economic climate. Farmer-led partnerships such as Fenland SOIL offer greater scalability and reduced transactional costs compared to farm businesses acting individually. Support for peatland restoration and paludiculture through the new, Defra-supported Big Nature Impact Fund could prove transformative by helping to direct green investment into the UK rather than the international voluntary markets.

While recent developments offer the promise of greater public and private investment into mitigating GHG emissions from lowland peatlands, it is vital that this does not simply displace the environmental costs and emissions associated with food production elsewhere. In particular, increased reliance on imported food is undesirable from the perspectives of both economics and food security, and risks simply 'offshoring' additional GHG emissions to other parts of the world. Ongoing work by members of the UKCEH project team, along with NIAB, for the WWF/Tesco partnership (Rhymes et al., 2022; in prep.) seeks to develop solutions to these challenges in relation to vegetable production, in particular. The Lowland Agricultural Peat Task Force has also addressed these questions in depth. While there is no single solution, the results of this project and the assessment described here suggest that a mixed model may be required, in which some areas continue to be managed intensively for food production, with mitigation measures applied, while other parts of the peatland landscape are managed for energy and biomass production, conservation, and carbon capture. Developing effective solutions will require a holistic approach with effective spatial planning to ensure that the right solutions applied in the right areas, that solutions are implemented at scale, that water is managed, stored and distributed more efficiently at a landscape scale, and that financial mechanisms are put in place to incentivise changes in management.

# 7. Other project-related activities

The work undertaken by the project, and members of the project team, have supported a broad range of related activities during the project. Many of these activities have been underpinned by data collected as part of the project. They include:

- **Defra Lowland Agricultural Peat Task Force**. Chris Evans was a member of the national group of the Task Force, Sue Page was a member of the Eastern England group, and Richard Lindsay was a member of both the national group and the Paludiculture subgroup. Ben Freeman, one of the PhD students affiliated to the project, was seconded to Defra to undertake a literature review in support of the Task Force. Project members provided scientific advice to the task force including latest data on the relationships between lowland peat management and GHG emissions, effectiveness of different mitigation options, the impacts of peatland management on water demand (based on a new analysis of flux tower evapotranspiration data, as described earlier), the identification of evidence gaps and research needs, and support for Defra and the Task Force Chair in finalising the recommendations.

- **Peatland Code 2.0**. Along with the James Hutton Institute, UKCEH led an updated analysis of emission factors used in the Peatland Code for Defra and the IUCN Peatland Programme. Project data were also used to produce a new methodology, including a carbon calculator tool, for estimating $CO_2$ and $CH_4$ emissions as a function of water table and peat depth, which have for the first time enabled fen peat restoration projects to become eligible for funding via the Peatland Code.

- **National Atmospheric Emissions Inventory update**. The analysis above was used to update emission factors for cropland and grassland on peat used in the UK's national emissions inventory. A new emission factor for cropland on wasted peat was also developed as part of the BEIS Wasted Peat project, which used flux tower data collected as in the Defra project to examine year-to-year variability, and as a source of some of the wasted peat data used.

- **House of Lords Science and Technology Committee Inquiry into Nature Based Solutions**. Chris Evans and Richard Lindsay provided evidence to the peatland session.

- **England Peat Strategy**. Chris Evans participated in the online launch event for the England Peat Strategy at the COP26 meeting in Glasgow.

- **England Peat Map**. Chris Evans participated in the Defra steering group for the England Peat Map, and provided advice on the data requirements to enable condition assessment and emissions reporting, including for lowland peat. Jennifer Williamson at UKCEH subsequently led the pilot survey for Natural England.

- **Fenland SOIL**. Several members of the project team (Chris Evans, Ross Morrison, Jörg Kaduk, Sue Page) are providing ongoing contributions to the work of the farmer-led Fenland SOIL group, providing scientific support and participating in a number of meetings including the forthcoming Fenland SOIL lowland peat conference.

- **PhD projects**. Three PhD projects (Sam Musarika, Tom Newman, Ben Freeman) were affiliated to the project and supervised by members of the consortium, providing additional data and helping to build capacity in this area.

- **Presentation of project results to external bodies**:

    i. Come hell or high water? Mitigation of greenhouse gas emissions from agriculturally drained peatlands. Chris Evans, Wetscapes Conference, Rostok, September 2019.

    ii. Greenhouse gas emissions from East Anglian and other lowland peatlands. Chris Evans, Tomorrow's Fenland: carbon, soil and livelihoods, Cambridgeshire, March 2020.

    iii. Quantifying and mitigating peatland greenhouse gas emissions: From sites to national and global inventories. Chris Evans, ICOS Conference, Finland, May 2022.

    iv. The role of hydrology in sustainable peatland management. Chris Evans, British Hydrological Society annual meeting, September 2021.

    v. Greenhouse gas removal potential of peat restoration. Chris Evans, IUCN Peatland Programme annual conference, September 2021.

    vi. Addressing the challenges of vegetable production on peat. Jenny Rhymes and Chris Evans, presentation to the UK retail sector for the WWF/Tesco partnership, January 2023.

    vii. Overview of UK peatland research. Chris Evans, National Trust peatland meeting, Derby, March 2023.

    viii. Overview of the challenges, evidence based and stakeholder networks for sustainable agricultural management of lowland agricultural peatlands. Chris Evans and Jörg Kaduk, UKRI Agrifood/Net Zero Nexus meeting, March 2023.

# 8. Project-related publications

[*Includes reports to other projects that drew to a significant extent on work undertaken during this project*]

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