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Saving our lowland peatlands

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

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

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

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

Signed: Benjamin W.J. Freeman

Date: 22nd November 2023

Thesis abstract

With global food demand expected to rise, there is a pressing need for climate change mitigation measures to limit food system greenhouse gas (GHG) emissions. Drained peatlands make an outsized contribution to agricultural emissions given their small area and contain a vulnerable store of irrecoverable carbon (C). This has justifiably made peatland GHG emission reductions a policy priority both in the United Kingdom (UK) and globally. However, peatland agriculture also makes important contributions to the production of high-value crops (e.g. fresh vegetables in the United Kingdom) and provides socioeconomic benefits. The programme of research presented in this thesis seeks to understand better how the environmental impacts of peatland agriculture can best be balanced with food production and economic benefits.

In reviewing the evidence base (Chapter 2), we observe that carbon dioxide emissions dominate the GHG balance of agricultural peatlands and are strongly influenced by drainage depth. We suggest that responsible management will require adaptation to the wetland character of peatlands by reducing the total volume of drained peat. The results of our expanded boundary life cycle assessment (Chapter 5) indicate that rewetting peat used for agriculture could provide substantial net climate benefits. However, with high-value agriculture, the climate benefits appear unlikely to offset the economic costs of relocating production at the current carbon price.

Responsible peatland management might therefore favour compromise, reduced drainage intensity strategies under high-value agriculture; aiming to continue production and partially reduce GHG emissions, through partial reductions in average annual drainage depth. However, the costs/benefits of such strategies appear highly sensitive to yield changes and further development is required before they can be recommended in practice. The cost/benefit ratio appears to be more favourable for rewetting under lower-value agriculture. Therefore, responsible management may favour relocating lower-value agriculture away from drained peatlands to create opportunities for full rewetting.

Wind erosion rates indicate that aeolian C losses from UK agricultural peatlands are lower than losses due to soil organic matter (SOM) mineralisation. However, the localised impacts of erosion can be substantial and resulting crop damage/contamination can reduce profitability (e.g. for high-value vegetable crops). Our field study (Chapter 3) found that bare peat surfaces are

vulnerable to erosion following the planting and irrigation of salad vegetable crops, regardless of SOM content. Therefore, erosion mitigation should be part of comprehensive responsible management strategies aiming to optimise the cost/benefit balance of agriculture on peatlands.

Our laboratory study (Chapter 4) indicates that the chemical soil stabiliser polyacrylamide (PAM) can stabilise high-SOM agricultural peat, making this a potential candidate mitigation option for areas/periods where bare soil is unavoidable. However, stabilisation required high application rates, so is likely to be expensive and cost-effective only where it provides clear economic benefits. These results were obtained under laboratory conditions, so further testing is required to examine the performance and persistence of PAM treatments under field conditions.

Responsible peatland management strategies will need to overcome agronomic, socio-economic and water management challenges, and workable solutions need to be developed alongside food producers. The communities occupying agricultural peatlands have a long history of overcoming environmental management challenges and this adaptive capacity should be drawn upon to ensure the success of future peatland management strategies.

The challenge of responsible use presents an exciting opportunity to rethink peatland management; to increase the resilience of food production systems, deliver environmental benefits, protect valuable peat resources, and invest in our future. The UK has an opportunity to be a global leader by creating thriving, innovative, green, and profitable peatland landscapes, delivering an important contribution to international climate change mitigation efforts.

Acknowledgements

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I am grateful to Andrew Newton and the staff at the Ely Group of Internal Drainage Boards for giving up their time to explain Fenland drainage management. Thanks are also due to the staff at the companies producing/distributing chemical soil stabilisers for their guidance and for providing product samples for testing. Finally, thank you to Saskia Boardman and everyone at Defra for the opportunity to undertake an extremely engaging internship and for giving me an insight into UK peatland policy.

Specific acknowledgements are detailed in the relevant sections of individual chapters. However, over the course of a PhD one crosses paths with many people. I hope I managed to convey my gratitude along the way to everyone who lent me a hand. Nonetheless, I know that some of our paths will not cross again, so I want to take this chance to reiterate my heartfelt thanks for every act of kindness shown to me over the course of the last few years. Whether you helped with one small thing or several big things, I simply couldn't have made it to this point without your collective generosity. Thank you very much indeed. I will strive to pay it forward.

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Chapter 1

Introduction

1.1 Global research context

Peat forms under persistent waterlogged conditions resulting from impeded drainage or precipitation exceeding drainage capacity. Water excludes air from soil pore spaces, creating anaerobic conditions and lowering soil redox potential. In the relative absence of oxygen as a terminal electron acceptor, soil microorganisms must utilise alternative electron acceptors with smaller reduction potentials. Soil organic matter (SOM) is therefore degraded relatively slowly by anaerobic processes (e.g. fermentation, methanogenesis) in wet peatlands.

As a result, the rate of accumulation of SOM due to inputs from vegetation (e.g. from litter fall, root exudates or mortality) can exceed the rate of SOM decomposition in wet peatlands. The resulting peat soils – also known as organic soils or histosols – are therefore characterised by their high SOM content (Table 1.1). The net accumulation of SOM allows undisturbed peatlands to act as a net carbon (C) sink of around -0.3 t C ha^{-1} yr^{-1} and to have a net climate cooling impact on millennial timescales (Frolking et al., 2011).

However, the long-term accumulation of SOM in peat deposits also creates an energy and nutrient dense natural resource, which in the case of lowland peatlands has relatively low access costs. Consequently, there is a long history of anthropogenic peatland drainage to facilitate the productive use of peat deposits (Sly, 2010). The main productive uses of peat *in situ* are agriculture and forestry, whilst extractive use has historically been dominated by use for fuel and more recently for use as a horticultural growing medium. Global peatlands cover an estimated area of 463 Mha and store ~600 Gt C but ~11% of the global peatland area is now subject to drainage associated with productive uses (Leifeld and Menichetti, 2018).

Drainage aerates the soil, increasing the availability of oxygen as a terminal electron acceptor. Consequently, more efficient aerobic microbial respiration pathways become dominant in drained peatlands. This leads to: (i) increased rates of SOM mineralisation, associated heterotrophic carbon dioxide (CO_2) emissions and C losses. (ii) methanotrophy in upper aerobic peat layers exceeding methanogenesis in deeper anoxic layers, leading to reduced methane (CH_4) emissions. (iii) stimulation of the nitrogen (N) cycle, with increased rates of ammonification and nitrification leading to higher emissions of nitrous oxide (N_2O).

As a result of these changes, drained peatlands are estimated to release annual greenhouse gas (GHG) emissions of 1.91 Gt CO_2 eq. globally (on a 100-year global warming potential basis; GWP; Leifeld and Menichetti, 2018). This represents approximately 3% of global anthropogenic net GHG emissions, which is an outsized contribution given drained peatlands occupy only ~0.4% of the global land area (IPCC, 2022).

Drained peatlands also experience substantial surface subsidence due to mineralisation of SOM, compaction of soil pore spaces, shrinkage of aggregates and erosion of surface particles (Hutchinson, 1980). Peatland drainage can therefore have substantial impacts at more local scales by increasing flood risk, damaging infrastructure and depleting soil nutrient stocks (Page et al., 2020), whilst wind erosion can damage crops and negatively affect air quality (Genis et al., 2013; Staffogia et al., 2016).

Approximately half of the drained peatland area is under agricultural use as drained cultivated organic soils are highly productive (Joosten and Clarke, 2002). Peatland agriculture benefits from the release of N (and other nutrients) by SOM mineralisation, relatively favourable water availability and the excellent water storage capacity of peat (Rochette et al., 2010; Liu et al., 2022). These factors provide substantial natural assistance in overcoming the key abiotic crop growth constraints of water and nutrient availability (Liliane and Charles, 2020). Consequently, drained organic soil can support highly profitable agricultural enterprises, and can play an important role in sustaining rural economies and communities (Rawlins and Morris, 2010; Rebhann et al., 2016).

However, the high profitability of peatland agriculture does not reflect the long-term and large-scale external environmental costs imposed by peatland drainage (Pieper et al., 2020). Depleted peatland C stocks are also practically irrecoverable, as the C lost cannot be replenished over human-relevant timescales (Goldstein et al., 2020; Noon et al., 2021). Therefore, conventional agricultural systems on organic soil are effectively both highly extractive (Anderson and Rivera-Ferre, 2021) and unsustainable (economically and environmentally; Wijedasa et al., 2016).

Against the backdrop of an increasing population and climate change mitigation targets, balancing food production and economic benefits against the environmental impacts of peatland agriculture presents a major challenge for policymakers (UNFCCC, 2015). Land managers, meanwhile, face difficult and risky decisions regarding the optimal use of a finite resource against a rapidly changing social and regulatory background (Ferré et al., 2019). This context produces a strong need for research to build a robust evidence base, which can support decision makers to develop responsible management strategies for agricultural peatlands.

Table 1.1. Characteristics of peat soil as described by classification systems. Sources: 1 = USDA (1999); 2 = IUSS (2022); 3 = Avery (1980).

Variable	Description
US soil taxonomy (Histosol)[1]	More than half of the upper 80 cm is organic material. Depth >40 cm unless bulk density very low (<0.1 g cm^{-3}), in which case depth >60 cm. Alternatively, shallower layers of organic material resting directly on bedrock or underlying volcanic material. Soil not affected by permafrost.
World reference base for Soil Resources (Histosol)[2]	Organic material starting ≤40 cm from soil surface and totalling 40-60 cm thickness (depending on moss fibre content). Alternatively, organic material starting at soil surface with depth ≥10 cm over ice/bedrock or ≥40 cm if organic material fills interstices between coarse fragments of mineral material.
Soil survey of England and Wales (Peat soil)[3]	Soils with >40 cm depth of organic topsoil.

1.2. Local research context

There are approximately 2.9 Mha of peatland in the UK, with two thirds of this area in Scotland, including large areas of near-natural, modified and afforested bog (Evans et al., 2017; Figure 1.1). However, agricultural land uses represent notable peatland GHG emissions hotspots when balancing emissions of CO_2, CH_4 and N_2O on a per ha basis (Evans et al., 2017).

Using recent emission factors (EFs; Evans et al., 2017, 2023a, 2023b) and the IPCC AR5 GWP values (Mhyre et al., 2013), UK agricultural peatlands can be estimated to produce 9.3 Mt CO_2 eq. yr^{-1}, which equates to approximately 55% of UK peatland GHG emissions, from only 15% of the UK's peatland area (0.43 Mha; Evans et al., 2017). By far the largest single contribution to agricultural peatland emissions is made by deeply drained and intensively cultivated cropland in England, which produces ~28% of UK peatland GHG emissions whilst occupying only ~6% of the total area.

The East Anglian Fens represent the largest area of agriculturally managed lowland peatland in the UK (Figure 1.1; Rhymes et al., 2023). The Fens were formed as the result of raised and fluctuating sea levels following the last glacial period. Repeated marine flooding and the resulting sediment deposits, impeded drainage of the rivers in the Fen Basin, producing highly favourable conditions for peat formation (Waller and Kirby, 2021).

It is likely that the peat area in the Fens was once ~150 kha, with large areas >5 m in depth (Hutchinson, 1980; Waller and Kirby, 2021). However, since the 17th century there has been a determined and progressively intensifying program of drainage in the region, predominantly for agricultural purposes (Sly, 2010). Consequently, the area has experienced significant subsidence and wastage, and the remaining area of peat soil >40 cm has been estimated at ~17 kha (Holman, 2009), with much of the former peat area now 'wasted' peat (<40 cm in depth, typically intermixed with mineral soil) or entirely lost.

Drainage has also made the Fens a highly productive industrial agricultural landscape. The Fens account for approximately 7% of England's total agricultural production, 33% of England's fresh vegetable production, and support a combined agricultural and food value chain industry worth £2.3 billion (GBP; gross value added), which employs 44,000 people (NFU, 2019). Fenland vegetable production alone is worth £357 million annually (NFU, 2019) and the region is associated with high quality fresh produce; for example, Fenland celery has Protected Geographical Indication status.

Addressing the challenge of balancing food production and economic benefits against the environmental impacts of peatland drainage in the Fens has become a UK policy priority (Defra, 2021, 2023). This has created a demand for high quality research to support decision makers in both government and the private sector. The program of research presented in this thesis was supported by G's Fresh, one of Europe's largest fresh produce businesses, which was founded in the Fens and continues to have a strong presence in the region. The research also benefited from alignment with major research council and government funded projects including the UK Natural Environment Research Council (NERC) funded SEFLOS project (Securing long-term ecosystem function in lowland organic soils) and Defra funded Lowland Peat 2 project (SP1218). This research environment ensured that (i) research objectives were tailored to the needs of food producers in the Fens, (ii) the research was informed by the most recent advances in the field and (iii) the research was grounded in the wider and evolving policy context.

Given this context, the overarching research question that the presented programme of research will seek to address is:

How can the environmental impacts of peatland agriculture best be balanced with food production and economic benefits?

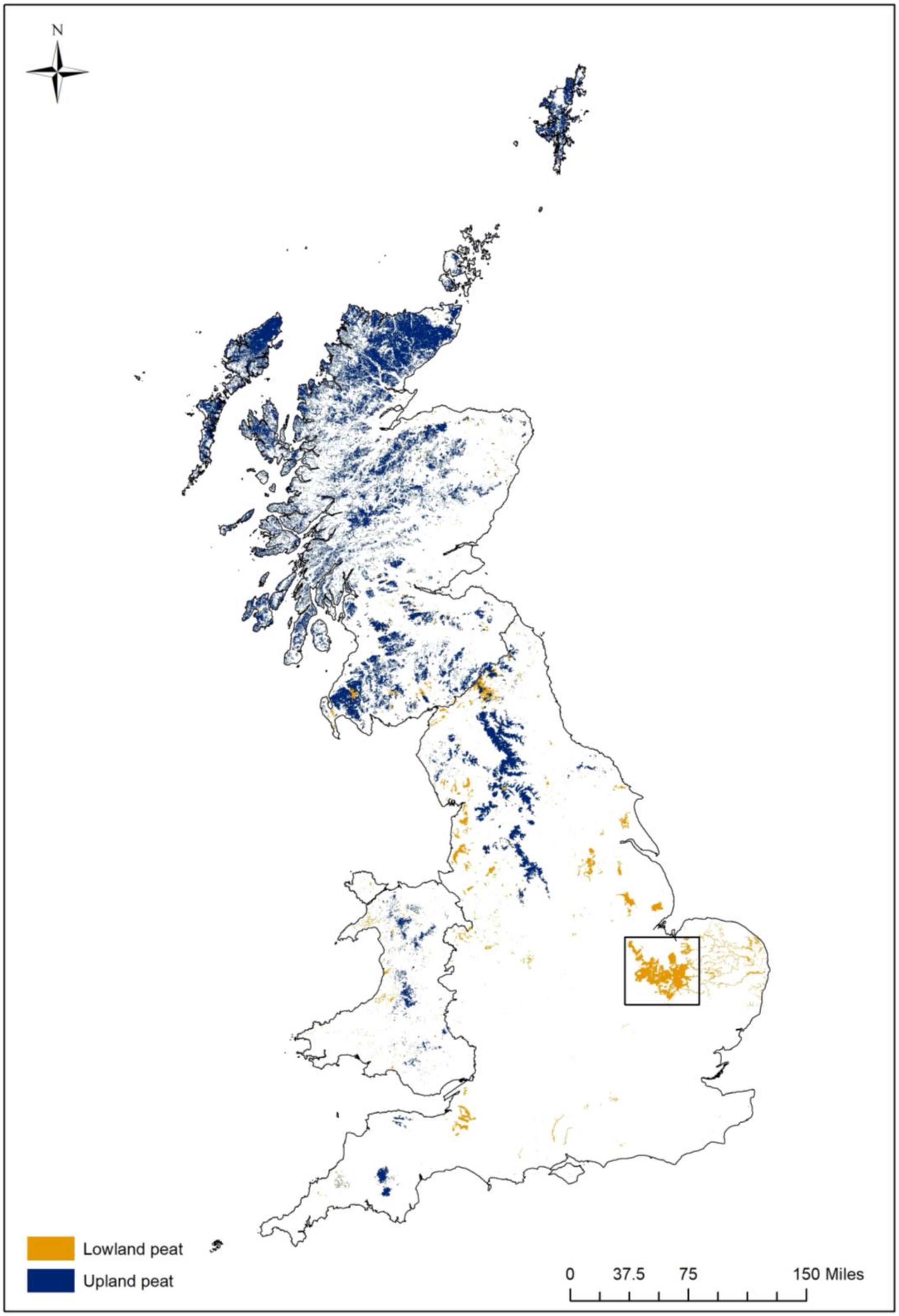

Figure 1.1. Distribution of peat soil for England, Scotland and Wales. The small black box indicates the location of the East Anglian Fens. Peat distribution data set: Evans et al. (2017). Boundaries data: Office for National Statistics licensed under the Open Government Licence v.3.0. Contains OS data © Crown copyright and database right 2024.

1.3. Research objectives

1.3.1. Evidence review

The overall structure of the thesis is summarised for the reader's convenience in Figure 1.2. Our first research objective was to review the available published evidence, in order to synthesise the current understanding of agricultural peatland systems and evaluate the potential challenges and opportunities for responsible management. We specifically aimed to: (i) critically assess the environmental impacts resulting from the agricultural use of peatlands; (ii) evaluate strategies presenting potential opportunities for responsible management (balancing food production/economic benefits against environmental impacts); (iii) highlight the main challenges facing the implementation of responsible management strategies. This review (presented in Chapter 2) was intended both to inform the program of research presented in this thesis and to provide a resource to support stakeholders and decisionmakers navigating the challenges of agricultural peatland management.

1.3.2. Characterisation of erosion processes

Research to estimate agricultural peatland GHG EFs and explore the underlying biochemical processes is now relatively well established. However, the evidence review (Chapter 2), highlighted a substantial knowledge gap around lowland agricultural peatland wind erosion processes and the associated C/nutrient losses. Wind erosion not only contributes to soil/nutrient loss and subsidence but can cause costly damage/contamination of crops. This is particularly relevant for fresh salad vegetables, which are both fragile/vulnerable in the early stages of growth and are also often packed fresh in the field. Chapter 3 presents a field study conducted to characterise the vulnerability to wind erosion of agriculturally managed UK lowland peat surfaces in the East Anglian Fens. We specifically aimed to: (i) examine the interactions between farm management practices, soil physical properties, and erosion vulnerability during bare soil periods; (ii) identify potential spatiotemporal peaks of erosion vulnerability and highlight the predisposing factors; (iii) guide the development and targeting of wind erosion mitigation strategies to assist food producers in minimising soil loss and crop damage/contamination.

1.3.3. Evaluation of an erosion mitigation option

The results from Chapter 3 showed that during bare soil periods, agriculturally managed peatlands can have high potential vulnerability to wind erosion. Wind erosion may be mitigated by increasing soil cover, either through vegetation or artificial cover (e.g. fleecing). However, current crop management practices would still unavoidably result in periods of bare soil due to tillage and on field vehicle activity. Chemical soil stabilisers represent a potential option for targeted mitigation of wind erosion during bare soil periods but have generally been considered unsuitable for use on organic soils. Therefore, Chapter 4 presents a laboratory study to evaluate the erosion mitigation potential of chemical soil stabilisers for high-SOM content agricultural peat soil. We specifically aimed to: (i) evaluate the performance of several commercially available products with different chemical compositions; (ii) identify products with the capacity to stabilise and reduce the erosion vulnerability of agricultural peat surfaces; (iii) take a first step on the path towards potentially developing chemical soil stabilisers as an erosion control measure for agricultural peatlands.

1.3.4. Evaluation of peatland rewetting for climate change mitigation

The evidence review (Chapter 2) indicated that the single most efficacious strategy for reducing total net GHG emissions from drained peatlands is to reduce the depth to which peat is drained. However, when evaluating the potential effects of responsible management strategies, it is essential to assess the wider consequences of land use change across the full life cycle, and to balance climate benefits against the potential loss of food production/economic benefits resulting from peatland drainage. Chapter 5 therefore presents an expanded boundary life cycle assessment (LCA) of UK lettuce production, with a focus on the potential consequences of rewetting cultivated organic soils. We specifically aimed to: (i) produce a comparative assessment of the environmental impacts of the major supply chains supplying the UK lettuce market; (ii) evaluate the net climate change impact (NCCI) of rewetting cultivated organic soil and relocating this production to mineral soil; (iii) upscale our LCA results to make an initial assessment of the potential NCCI of rewetting policies in the UK; (iv) contextualise our NCCI estimates by considering the financial implications of rewetting agricultural peatlands.

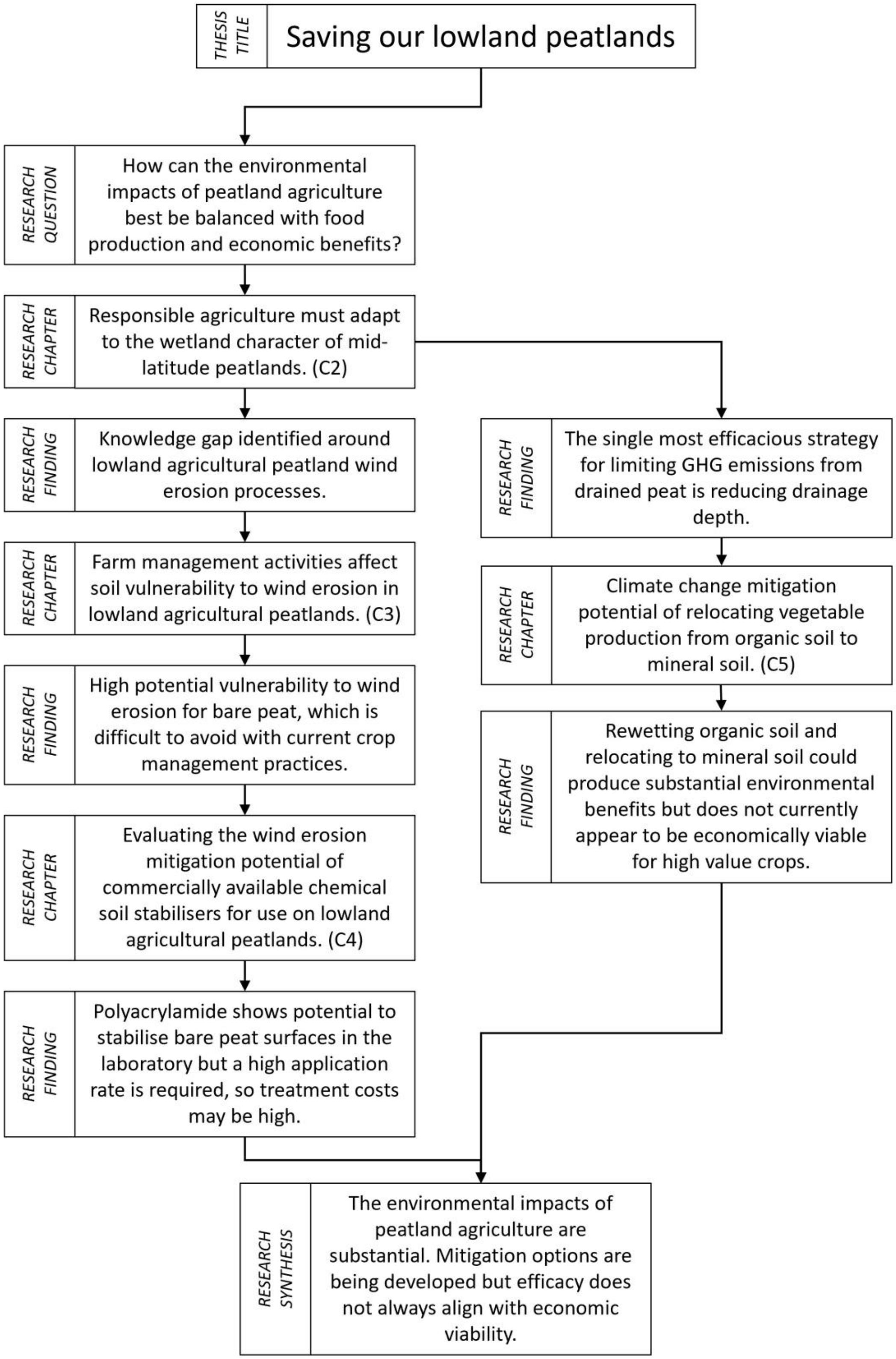

Figure 1.2. Schematic of the research programme presented in this thesis.

1.4. Further research contributions

Beyond the chapters presented in this thesis, the program of research undertaken also resulted in contributions to the following reports and peer-reviewed articles:

- Wen, Y., Zang, H., Freeman, B., Musarika, S., Evans, C.D., Chadwick, D.R., Jones, D.L., 2019. Microbial utilization of low molecular weight organic carbon substrates in cultivated peats in response to warming and soil degradation. Soil Biology and Biochemistry, 139, 107629.

- Wen, Y., Zang, H., Freeman, B., Ma, Q., Chadwick, D.R., Jones, D.L., 2019. Rye cover crop incorporation and high water table mitigate greenhouse gas emissions in cultivated peatland. Land Degradation and Development, 30, 1928-1938.

- Wen, Y., Zang, H., Ma, Q., Freeman, B., Chadwick, D.R., Evans, C.D., Jones, D.L., 2020. Impact of water table levels and winter cover crops on greenhouse gas emissions from cultivated peat soils. Science of the Total Environment, 719, 135130.

- Wen, Y., Freeman, B., Ma, Q., Evans, C.D., Chadwick, D.R., Zang, H., Jones, D.L., 2020. Raising the groundwater table in the non-growing season can reduce greenhouse gas emissions and maintain crop productivity in cultivated fen peats. Journal of Cleaner Production, 262, 121179.

- Mulholland, B., Abdel-Aziz, I., Lindsay, R., McNamara, N., Keith, A., Page, S., Clough, J., Freeman, B., Evans C., 2020. An assessment of the potential for paludiculture in England and Wales. Report to Defra for Project SP1218, 98 pp.

- Wen, Y., Freeman, B., Hunt, D., Musarika, S., Zang, H., Marsden, K., Evans, C.D., Chadwick, D.R., Jones, D.L., 2021. Livestock-induced N2O emissions may limit the benefits of converting cropland to grazed grassland as a greenhouse gas mitigation strategy for agricultural peatlands. Resources, Conservation and Recycling, 174, 105764.

- Casey, L., Freeman, B., Francis, K., Brychkova, G., McKeown, P., Spillane, C., Bezrukov, A., Zaworotko, M., Styles, D., 2022. Comparative environmental footprints of lettuce supplied by hydroponic controlled-environment agriculture and field-based supply chains. Journal of Cleaner Production, 369, 133214.

- Evans, C., Artz, R., Burden, A., Clilverd, H., Freeman, B., Heinemeyer, A., Lindsay, R., Morrison, R., Potts, J., Reed, M., Williamson, J., 2023. Aligning the peatland code with the UK peatland inventory. Report to Defra and the IUCN Peatland Programme, March 2022 (Updated January 2023).

- Evans, C., Freeman, B., Artz, R., Burden, A., Burton, R., Chadwick, D., Clilverd, H., Cooper, H., Cowan, N., Cumming, A., Hudson, M., Jones, D., Kaduk, J., Newman, T., Page, S., Potts, J., Morrison, R., 2023. Wasted Peat Emission Factor Assessment. Report to BEIS, UK Centre for Ecology and Hydrology.

- Evans, C., Morrison, R., Cumming, A., Bodo, A., Burden, A., Callaghan, N., Clilverd, H., Cooper, H., Cowan, N., Crabtree, D., D'Acunha, B., Freeman, B., Rhymes, J., Jovani-Sancho, J., Keith, A., McNamara, N., Musarika, S., Rylett, D., Page, S., Kaduk, J., Mills, M., Newman, T., Chadwick, D., Hardaker, A., Gibbons, J., Jones, D., Abdul-Aziz, I., Eyre, C., Mulholland, B., Baird, A., Lindsay, R., Clough, J., Hudson, M., Palmer, L., Burton, R., 2023. 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, UK Centre for Ecology and Hydrology.

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Chapter 2

Responsible agriculture must adapt to the wetland character of mid-latitude peatlands.

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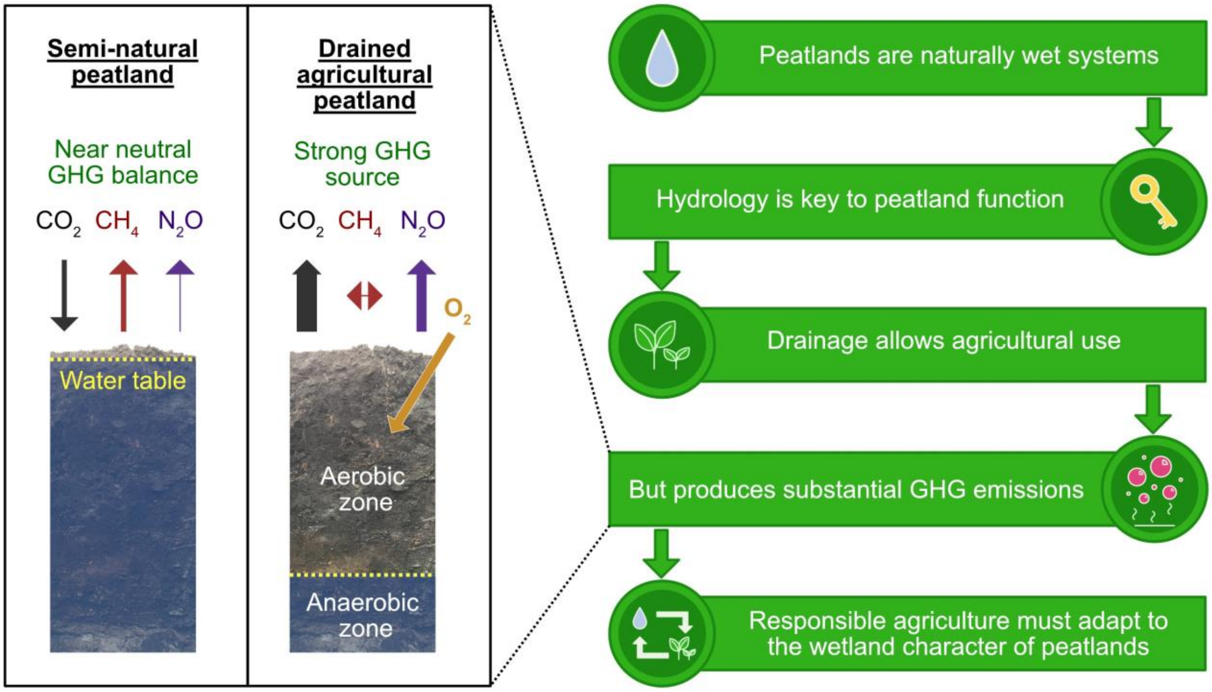

Graphical Abstract. Peatlands are a globally important but diminishing and irrecoverable carbon store. Hydrology is a dominant driver of peatland ecosystem function, and water table depth is a strong predictor of peatland greenhouse gas emissions. Responsible management requires agriculture to adapt to the wetland character of peatlands. Wetland agriculture strategies could increase the resilience of production systems, deliver a wider range of environmental benefits and protect these valuable ecosystems for everyone's future benefit, whilst making a vital contribution to global climate change mitigation efforts.

Abstract

Drained, lowland agricultural peatlands are greenhouse gas (GHG) emission hotspots and a large but vulnerable store of irrecoverable carbon. They exhibit soil loss rates of ~2.0 cm yr^{-1} and are estimated to account for 32% of global cropland emissions whilst producing only 1.1% of crop kilocalories. Carbon dioxide emissions account for >80% of their terrestrial GHG emissions and are largely controlled by water table depth. Reducing drainage depths is therefore essential for responsible peatland management. Peatland restoration can substantially reduce emissions. However, this may conflict with societal needs to maintain productive use, to protect food security and livelihoods. Wetland agriculture strategies will therefore be required to adapt agriculture to the wetland character of peatlands, and balance GHG mitigation against productivity, where halting emissions is not immediately possible. Paludiculture may substantially reduce GHG emissions but will not always be viable in the current economic landscape. Reduced drainage intensity systems may deliver partial reductions in the rate of emissions, with smaller modifications to existing systems. These compromise systems may face fewer hurdles to adoption and minimise environmental harm until societal conditions favour strategies that can halt emissions. Wetland agriculture will face agronomic, socio-economic and water management challenges, and careful implementation will be required. Diversity of values and priorities amongst stakeholders creates the potential for conflict. Successful implementation will require participatory research approaches and co-creation of workable solutions. Policymakers, private sector funders and researchers have key roles to play but adoption risks would fall predominantly on land managers. Development of a robust wetland agriculture paradigm is essential to deliver resilient production systems and wider environmental benefits. The challenge of responsible use presents an opportunity to rethink peatland management and create thriving, innovative and green wetland landscapes for everyone's future benefit, whilst making a vital contribution to global climate change mitigation.

Keywords: Climate Change Mitigation, Peatlands, Carbon, Soil Loss, Greenhouse Gases, Paludiculture, Hydrology, Wetland Agriculture.

2.1. Introduction

The Agriculture, Forestry and Other Land Use (AFOLU) sector contributes ~24% of global greenhouse gas (GHG) emissions (Smith et al., 2014). Agricultural production will need to rise by ~60% to meet global food demands by 2050 (Alexandratos and Bruinsma, 2012), and mitigation measures will be required to limit associated increases in agricultural GHG emissions (Bennetzen et al., 2016). The challenge of balancing climate change mitigation and adaptation with achieving food security has been formally recognised by policymakers in the Paris Agreement (UNFCCC, 2015) and is particularly acute for agriculturally managed peatlands.

Global peatlands store >600 Gt of carbon (C) in an estimated area of 4.23 million km^2 (Yu et al., 2010; Xu et al., 2018). This represents more C than was added to the atmosphere by total anthropogenic carbon dioxide (CO_2) emissions between 1750 and 2011, stored on less than 3% of the global land area (IPCC, 2014). Many peatlands have been drained to enhance the delivery of economically valuable provisioning services (e.g. food and timber). Consequently, there are an estimated 500,000 km^2 of heavily modified, diminishing peatlands globally, emitting 1.2 – 1.9 Gt CO_2-e yr^{-1} and contributing ~14% of AFOLU GHG emissions (Joosten, 2010; Leifeld and Menichetti, 2018). Halting these emissions will be integral to achieving climate stabilisation (Günther et al., 2020). This review focuses on mid-latitude (non-tropical and non-polar; Appendix 2.1) peatlands, which contain ~75% of peatland C and account for ~22% of modified peatland GHG emissions (Leifeld and Menichetti, 2018).

Agriculture is responsible for ~50% of the peatland conversion that has occurred in mid-latitude areas, with the greatest impacts seen on relatively accessible lowland fens and raised bogs (Joosten and Clarke, 2002). Forestry and peat extraction account for much of the remainder. Drainage has facilitated the application of conventional agricultural systems, which originated in dryland regions, to naturally wet lowland peatlands, producing highly productive agroecosystems. However, it is estimated that peatland agriculture accounts for 32% of global cropland GHG emissions, despite producing only 1.1% of total crop kilocalories (Carlson et al., 2017). Peatland C stocks are irrecoverable, as C depleted by drainage cannot be replenished over human-relevant timescales (Noon et al., 2021). Therefore, conventional agriculture on peatlands is neither economically nor environmentally sustainable (Wijedasa et al., 2016). Given the cultural and economic significance of peatland agriculture to many of the regions where it occurs, high levels of private land ownership and the absence of mechanisms to reflect the high external costs of peatland GHG emissions, full restoration to pre-drainage condition and function is unlikely to be immediately viable in all cases. Full rewetting and complete halting of peatland GHG emissions should remain a long-term goal. However, it is imperative that responsible peatland management strategies are developed, which adapt productive agricultural management to the wetland character of peatlands and slow peat loss/emission rates; minimising harm until societal conditions favour halting emissions (Clarke and Rieley, 2019). This review aims to: (i) critically assess the impacts resulting from drainage-based agriculture on mid-latitude lowland peatlands, (ii) evaluate the potential for wetland agriculture systems to enable responsible peatland management and, (iii) highlight key challenges, which must be addressed to inform research priorities, and support land managers and policymakers in this vital undertaking.

2.2. Agricultural drainage impacts

Peat soils (histosols) are those with more than 50% organic material in the top 80 cm, or with shallower organic deposits (C content >12-18%) resting directly on bedrock and not influenced by permafrost (USDA Soil Survey Staff, 1999). Peatlands form when impeded drainage produces waterlogged conditions and the rate of organic matter accumulation exceeds the rate of decomposition. As a result, undisturbed mid-latitude peatlands act as a net C sink (~ -0.32 t C ha^{-1} yr^{-1}) and have a net climate cooling impact on millennial timescales; on a 100-year global warming potential (GWP) basis, the balance of CO_2, methane (CH_4) and nitrous oxide (N_2O) results in a small net GHG source (Frolking et al., 2011).

Conversion to the deeply drained agricultural landscapes, characteristic of today's mid-latitude lowland peatlands began with gravity drainage and accelerated sequentially with the harnessing of wind-power (11th Century AD), the advent of steam power and the centrifugal water pump (1800s), and most recently modern diesel engines and electric pumps (Sly, 2010). Drainage has a range of impacts on peatland function (Figure 2.1). The most visible impact is subsidence of the land surface, which enhances flood risk and causes costly damage to infrastructure (Page et al., 2020). Subsidence results from physical shrinkage of organic matter, compaction of the peat pore spaces, microbial mineralisation of soil organic matter (SOM) and increased vulnerability to erosion. Measured subsidence rates from the literature are relatively consistent across mid-latitude drained agricultural peatlands, with a median rate of 2.0 cm yr^{-1} (Quartiles = 1.3 – 2.7, n = 48; Appendix 2.2).

Mineralisation of SOM has been estimated to account for 28 – 64% of subsidence in a temperate climate (Leifeld et al., 2011). In the early years following drainage, primary subsidence is rapid, dominated by shrinkage and compaction, and results in large volumetric losses. However, it is subsequent, more gradual, secondary subsidence, with larger contributions from mineralisation, which results in the depletion of SOM/C stocks, until eventually, the loss of peat depth and C content proceeds to such an extent that the soil is no longer classifiable as peat (Hutchinson, 1980).

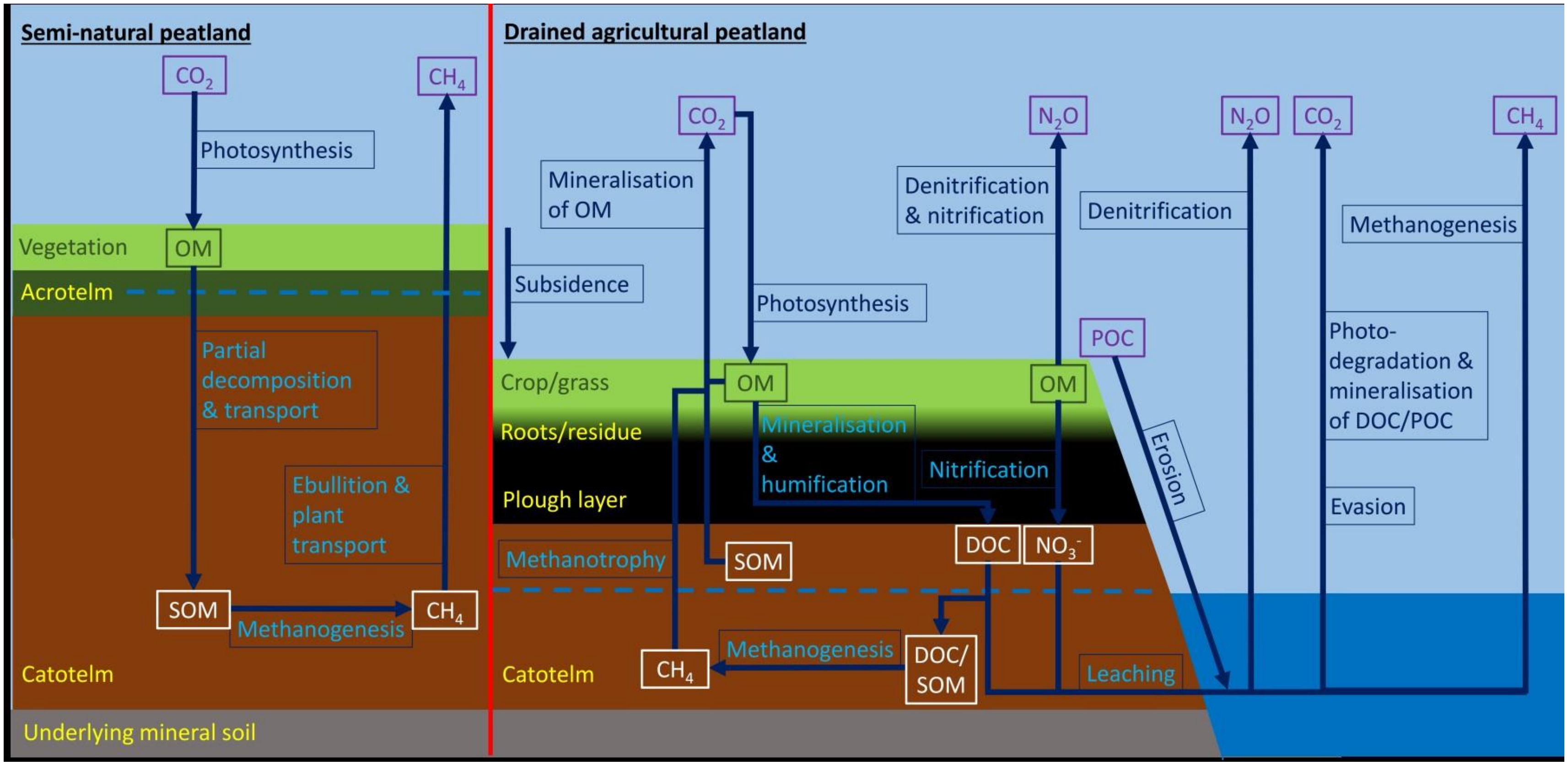

Figure 2.1. Impacts of drainage for agriculture on fundamental peatland processes. Blue dashed lines indicate the average water table depth (WTD) with fluctuation around this level assumed. Semi-natural peatlands are approximately carbon neutral but can be slight net sinks or sources of greenhouse gas (GHG) emissions depending on methane emissions. Drained peatlands are strong sources of GHG emissions from both fields and ditches. The acrotelm is the partially aerated upper layer of semi-natural peatlands, whilst the catotelm is the submerged, anaerobic, lower peat layer. Fluctuations in the WTD produce a dynamic mesotelm layer between these, which has been omitted for clarity. Additions of fertiliser and livestock excreta increase labile carbon (C) and nitrogen (N) stocks in agricultural peatlands, exacerbating changes in C and N cycling. The fate of dissolved organic material leached from semi-natural peatlands to streams and rivers is similar to that shown for drained peatlands and is omitted from the diagram in the interest of space. CH_4 = methane, CO_2 = carbon dioxide, DOC = dissolved organic carbon, DOM = dissolved organic matter, N_2O = nitrous oxide, OM = organic matter, POC = particulate organic carbon, SOM = soil organic matter.

Drainage results in increased oxygenation and thus soil redox potential, which facilitates aerobic respiration of soil microbes and increases CO_2 emissions. This emission is substantial and dominates the overall GHG balance of the ecosystem (Table 2.1). Using available emission factors (EFs) for temperate agricultural peatlands, the estimated contribution of soil CO_2 emissions to the terrestrial GHG balance is 82.9% for cropland, 89.7% for intensive grassland and 83.9% for extensive grassland (Appendix 2.3). It will therefore be necessary to reduce terrestrial CO_2 emissions if substantial mitigation of GHG emissions is to be achieved.

Mineralisation of SOM also releases nitrogen (N), resulting in high mineral-N supplies (e.g. 250-571 kg N ha^{-1} yr^{-1} for cropland; Rochette et al., 2010). Where these exceed crop N demands and provide substrate for microbial metabolism, substantial N_2O emissions can occur (Table 2.1; Poyda et al., 2016). Emissions of N_2O can represent an important component of the terrestrial GHG balance for cropland (17%; Appendix 2.3) and intensively managed grassland (9.2%; Appendix 2.3).

In drained peatlands, CH_4 oxidation potentials are high throughout the soil profile (Jerman et al., 2017), which results in near-zero terrestrial CH_4 emissions from croplands (Table 2.1). Grassland soil CH_4 emissions are also generally low but can be high during periods of inundation and rapid anaerobic decomposition of flood-intolerant grassland plant species (Tiemeyer et al., 2016). These conditions are mostly observed on shallow-drained, extensive grassland sites, where CH_4 emissions can constitute an important portion of the terrestrial GHG balance (9.8%; Appendix 2.3).

Methanogenesis is a major catabolic process in anoxic environments and can proceed rapidly in drainage ditches bordering fields. Observed ditch CH_4 fluxes are highly variable and currently poorly understood but can be substantial (Table 2.1) and may make an important contribution to overall site GHG budgets (Peacock et al., 2021).

Ditches also represent an export pathway for dissolved and particulate organic C (DOC and POC, respectively). In watercourses, this C is vulnerable to photodegradation and can be transformed by microbial activity, resulting in indirect CO_2 emissions, which may occur far from the peatland (e.g. in rivers, lakes or coastal seas), whilst dissolved CO_2 exported directly from the peat is generally released rapidly (Evans et al., 2016a). Indirect CO_2 emissions from DOC and POC export are generally low relative to terrestrial emissions in drained systems (Table 2.1).

Table 2.1. Tier 1 emission factors for mid-latitude peatlands under agricultural management. LCI/UCI = Lower/Upper 95% Confidence Intervals, n = number of studies included in deriving estimate, * = single value not available for composite metric. Values collated from Drösler et al. (2014).

Emission Factor	Land Use	Climate Zone	Nutrient status	Drainage Depth	Value	LCI	UCI	n
CO_2 ($t\ CO_2$-C $ha^{-1}\ yr^{-1}$)	Cropland	Boreo-temperate			7.9	6.5	9.4	39
	Grassland	Boreal			5.7	2.9	8.6	8
		Temperate	Low		5.3	3.7	6.9	39
			High	Deep	6.1	5	7.3	7
				Shallow	3.6	1.8	5.4	13
Field CH_4 (kg CH_4 $ha^{-1}\ yr^{-1}$)	Cropland	Boreo-temperate			0	-2.8	2.8	38
	Grassland	Boreal			1.4	-1.6	4.5	12
		Temperate	Low		1.8	0.72	2.9	9
			High	Deep	16	2.4	29	44
				Shallow	39	-2.9	81	16
N_2O (kg N_2O-N $ha^{-1}\ yr^{-1}$)	Cropland	Boreo-temperate			13	8.2	18	36
	Grassland	Boreal			9.5	4.6	14	16
		Temperate	Low		4.3	1.9	6.8	7
			High	Deep	8.2	4.9	11	47
				Shallow	1.6	0.56	2.7	13
Ditch CH_4 (kg CH_4 $ha^{-1}\ yr^{-1}$)	Agriculture	Boreo-temperate		Deep	1165	335	1995	6
	Grassland	Boreo-temperate		Shallow	527	285	769	5
DOC (t C $ha^{-1}\ yr^{-1}$)	Agriculture	Boreal			0.12	0.07	0.19	*
	Agriculture	Temperate			0.31	0.19	0.46	*

Secondary humification produces small, light particles, which are highly susceptible to wind erosion when exposed. The severity of wind erosion events on cultivated peatlands has long been known (Thompson, 1957) but measurements of their magnitude and dynamics are extremely rare. Cumming (2018) recorded sediment movements at the border of an arable UK peatland field and observed sediment fluxes of 0.87-4.88 t C ha^{-1} yr^{-1}. Peak fluxes coincided with periods of bare soil and high wind speed, suggesting losses may be much lower from permanent grassland. The ultimate fate of eroded material remains unclear, so it is not currently possible to estimate the contribution of aeolian losses to depletion of C stocks or indirect GHG emissions.

Anthropogenic drainage reduces peatland moisture content, increasing their flammability and the depth of peat available for combustion (Turetsky et al., 2011). Fire impacts can therefore be severe, with a typical uncontrolled fire estimated to emit 122 t CO_2-C ha^{-1} (Drösler et al., 2014). Controlled burning still takes place on some mid-latitude peatlands but the practice is rare, largely due to the air pollution impacts, demonstrated by recent fires in Russia and the UK (Chubarova et al., 2009; Graham et al., 2020). Accidental fire risks remain on abandoned sites where drainage may be poorly managed but these can be reduced by rewetting (Sirin et al., 2020).

Drainage-induced mineralisation of SOM is the dominant factor driving long-term subsidence, C stock depletion and GHG emissions in agriculturally managed peatlands. Consequently, suppressing rates of SOM mineralisation and CO_2 emissions is the primary pro-environmental objective required for responsible peatland management. N_2O and CH_4 emissions can also make important contributions to GHG balances and must be considered, along with provisions to mitigate erosion losses and fire risk.

2.3. Water table control of emissions

Average annual peatland subsidence rates are linearly related to the average annual water table depth (WTD), increasing by an estimated 0.2 cm yr^{-1} for every 0.1 m of additional drainage in non-tropical peatlands (Evans et al., 2019). This result is derived from long-term subsidence data, including sites with long drainage histories. Very long-term studies show decreasing subsidence rates over time (Hutchinson, 1980; Stephens et al., 1984) and rapid primary subsidence immediately following drainage was not included in this analysis. Average annual WTD strongly influences the volume of aerated organic matter and thus microbial activity, making it a very convenient indicator of peatland function. It is a simple measure, which cannot capture site-specific differences in moisture content, oxygen concentration and C density in the unsaturated zone. Variables such as average summer WTD (Weideveld et al., 2021) and hydrograph skewness (Tiemeyer et al., 2016) may offer more nuance. However, average annual WTD has been widely used in the literature to date and data are more often available. We therefore adopt this measure of WTD unless otherwise stated.

Several data syntheses have shown a positive relationship between CO_2 emissions and peatland WTD (Figure 2.2a; Couwenberg et al., 2011; Tiemeyer et al., 2020; Evans et al., 2021). The slope and shape of the fitted relationships vary between these studies. However, there is strong agreement that (i) CO_2 emissions are high on drained agricultural peatlands (mean predicted value = 24.7±5.9 t CO_2 ha^{-1} yr^{-1} when WTD = 0.5 m) and (ii) surface-level WTDs on semi-natural sites result in net CO_2 uptake (mean predicted value = -5.2±0.7 t CO_2 ha^{-1} yr^{-1} when WTD = 0 m). Experimental manipulations of WTD remove potentially confounding differences between land uses and still generally support this trend for WTDs ≤0.7 m (e.g. Karki et al., 2014; Regina et al., 2015). Overall, the clear implication is that WTDs nearer the surface are linked to lower rates of SOM mineralisation and CO_2 emissions in peatlands. This analysis also indicates that no new peatland drainage should occur if overall peatland GHG emissions are to be reduced.

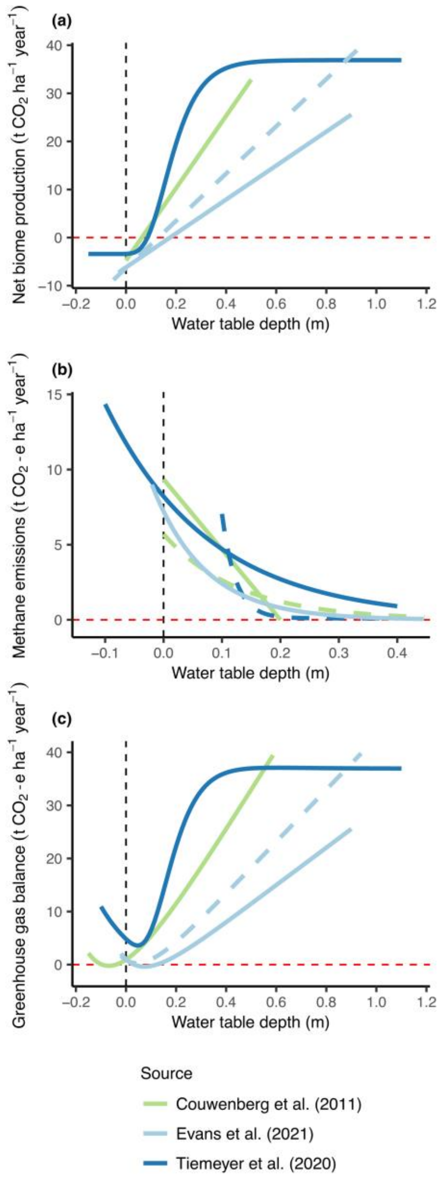

Figure 2.2. Relationships between peatland water table depth (WTD) and carbon-derived greenhouse gas emissions. (a) Net biome production (NBP; sum of ecosystem respiration, gross primary productivity and carbon import/export). Dashed and solid light blue lines represent UK and global relationships, respectively, in Evans et al. (2021). (b) Terrestrial methane emissions (CH_4; excluding ditch emissions and converted to CO_2 equivalent using a 100-year global warming potential of 28). Dashed and solid dark blue lines represent relationships for agricultural and rewetted sites, respectively, in Tiemeyer et al. (2020). Solid and dashed green lines indicate the published relationship from Couwenberg et al. (2011) and an exponential function fitted to a digitised subset of this data (see Appendix 2.4 for detailed description). (c) Terrestrial GHG balance of CO_2 (NBP) and CH_4. Functions for Tiemeyer et al. (2020) and Couwenberg et al. (2011) produced using rewetted site and exponential CH_4 functions, respectively. Dashed and solid light blue lines as for (a). Vertical dashed black lines indicate the peat surface (WTD = 0 m). More positive WTD values indicate deeper drainage and negative values indicate inundation. Horizontal red dashed lines indicate emission values of zero.

Terrestrial CH_4 emissions from peatlands are consistently very low at WTDs deeper than 0.25 m (Figure 2.2b). However, a sharp increase is observed as WTDs are reduced beyond this level. There is agreement across several data syntheses regarding this exponential response (Couwenberg et al., 2011; Tiemeyer et al., 2020; Evans et al., 2021). In balancing CO_2 against CH_4, C-derived GHG emissions appear to be minimised when the WTD is close to the peat surface (Figure 2.2c). The mean optimal WTD indicated for C-derived-GHG mitigation is 0.04±0.03 m based on a 100-year GWP of 28 for CH_4 (mean prediction of the four functions shown in Figure 2.2c plus the Tiemeyer et al. (2020) GHG balance using agricultural site CH_4 emissions not shown in Figure 2.2c; Myhre et al., 2013). This is slightly deeper (0.08±0.02 m) when estimated using a 100-year sustained GWP of 45 for CH_4 (Neubauer and Megonigal, 2015). Further inundation increases CH_4 emissions, offsetting CO_2 reductions and indicating that flooded peatlands would be GHG sources.

Deeply drained agricultural sites exhibit higher N_2O emissions than near-natural sites and so the overall pattern is of increasing N_2O emissions with deeper peatland WTDs (Figure 2.3; Leppelt et al., 2014). However, as for CO_2, management practices (e.g. N inputs and vegetation) potentially confound this relationship and the situation appears more complex within land use categories. Tiemeyer et al. (2016) found that the extent of WTD fluctuations and topsoil N stocks best predicted N_2O emissions from German peat grasslands, whilst rainfall (Taghizadeh-Toosi et al., 2019) and irrigation (Rochette et al., 2010) have also been observed to stimulate N_2O emissions from agricultural peatlands. Hot moments, driven by the onset of winter flooding, irrigation and fertilisation, accounted for 45% of annual N_2O emissions from a peat cropland in California (Anthony and Silver, 2021). These studies highlight the risk of large, denitrification-driven N_2O pulses when drained soils are subject to acute wetting events and WTD fluctuations. This is especially the case following prolonged periods of drainage, when mineralisation and nitrification of SOM, along with anthropogenic N inputs, lead to nitrate (NO_3^-) accumulation, providing plentiful substrate for denitrification upon subsequent wetting (Taghizadeh-Toosi et al., 2019). N_2O emissions are likely to be low from consistently fully saturated peat, due to inhibition of nitrification and complete denitrification to N_2.

The evidence implies that minimising GHG emission rates from agricultural peatlands will require maintenance of shallower and more stable WTDs, alongside reduced soil mineral-N concentrations. Whilst near-surface WTDs would be optimal for GHG mitigation, there is some indication that smaller reductions in WTD, deeper in the peat profile, may partially mitigate GHG emissions (Evans et al., 2021). However, such partial changes could only be expected to slow peat loss rates, with mineralisation continuing in the aerated layer and leading to eventual loss of the peat. Excessive inundation has the potential to induce substantial CH_4 emissions, and may also constrain primary productivity, peat formation and associated CO_2 uptake. Because of the strong control that WTD exerts over C-derived GHG emission rates (the majority of the GHG budget), WTD management clearly represents the most efficacious tool available to slow the rate of peat loss. Responsible WTD management is therefore essential for responsible peatland management. Development of wetland agriculture systems will be necessary, in order to adapt agricultural production to the wetland character of peatlands and ensure continued delivery of provisioning services alongside improved environmental outcomes.

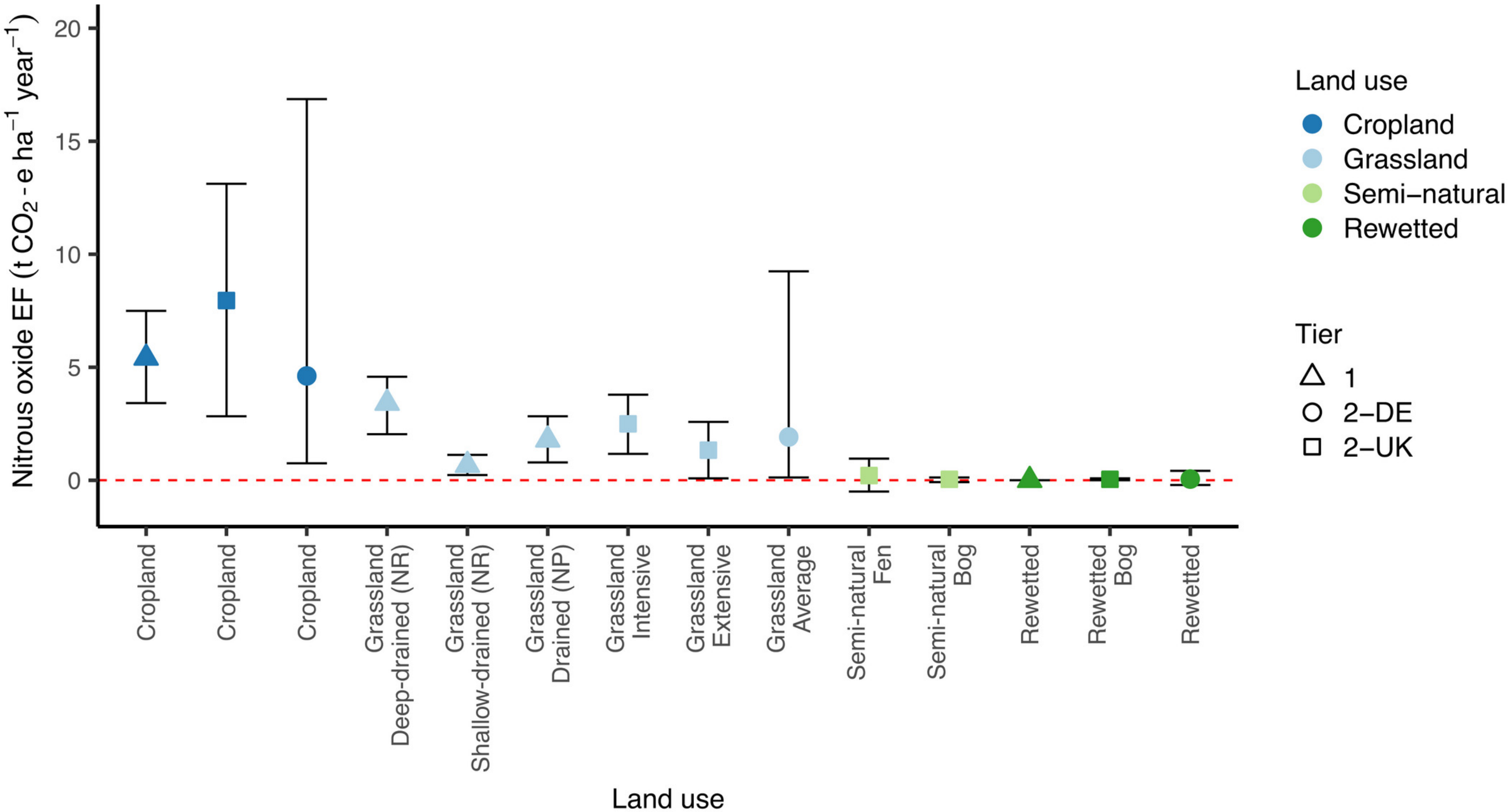

Figure 2.3. Nitrous oxide emission factors for selected land use categories. Error bars indicate 95% confidence intervals (CIs). The horizontal red dashed line indicates zero emissions and is included to highlight that the CIs for cropland and grassland sites exclude zero, whilst the CIs for semi-natural and rewetted sites include zero. N_2O was converted to CO_2 equivalent using a 100-year global warming potential of 265 (Myhre et al., 2013) to aid comparison with carbon-derived greenhouse gas emissions. Land use categories are presented in approximate order of decreasing water table depth (WTD), with deeper drained agricultural sites on the left and near-surface WTDs on the right. NR = nutrient-rich and NP = nutrient-poor. Tier 1 (Default) emissions factors (EFs) were sourced from Drösler et al. (2014), Tier 2 (Germany; DE) EFs from Tiemeyer et al. (2020) and Tier 2 (United Kingdom; UK) EFs from Evans et al. (2017).

2.4. Wetland agriculture systems

2.4.1. Paludiculture systems

Responsible peatland management requires a balance to be struck between GHG mitigation, food security and economic productivity. Rewetting and restoring peatlands to semi-natural conditions can achieve substantial reductions in the rate of GHG emissions (Nugent et al., 2019) and will represent an important component of national and international climate strategies. However, it greatly reduces delivery of provisioning services from peatlands, which will not be desirable in all cases. Paludiculture is a form of wetland agriculture, which pairs near-surface WTDs with production systems compatible with the resulting wet conditions (Wichtmann et al., 2016; Table 2.2). It therefore represents a compromise approach, weighted strongly towards GHG mitigation but without complete loss of the provisioning services prioritised under the currently dominant paradigm of drainage-based, conventional agriculture.

Several paludiculture systems have been identified that can substantially reduce soil loss rates. Cultivation of biomass species for the production of bioenergy and biofuels on rewetted peatlands in northern Europe has shown potential to produce near-neutral onsite CO_2 balances even after biomass removal (Günther et al., 2015; Kandel et al., 2017). Bioenergy from Reed Canary Grass (*Phalaris arundinacea*) has been estimated to produce ~40% lower GHG emissions per unit of energy generated than coal combustion and to result in a negative overall GHG balance (Shurpali et al., 2010; Järveoja et al., 2013). Wet conditions favoured by paludiculture crops inhibit methanotrophy, leading to net-emission of CH_4, reducing the favourability of their GHG balance on a 100-year GWP basis and requiring careful management (e.g. Günther et al., 2015). However, the suitability of using 100-year GWP to assess the warming equivalence of short-lived pollutants is questionable (Cain et al., 2019). The relatively short atmospheric lifespan of CH_4, means that adoption of these systems may still be favoured as it would result in a beneficial climatic effect over time, due to avoided emissions of more atmospherically persistent CO_2 (Günther et al., 2020).

Peat moss (*Sphagnum* spp.) production sites can be net CO_2 sinks during cultivation and may be essentially C neutral after accounting for biomass harvest (Beyer et al., 2015; Günther et al., 2017). The GHG balance over the full life cycle may be negative due to avoided emissions if horticultural peat extraction were replaced by *Sphagnum* cultivation. However, commercially viable *Sphagnum* cultivation faces practical challenges. Farmed *Sphagnum* is liable to be more expensive than extracted peat and it is currently unclear whether it can fully replicate peat's properties as a growing medium (Mulholland et al., 2020). Biomass and *Sphagnum* cultivation systems are both subject to high initial investments and harvesting costs, as specialist machinery is required to manage wet peatlands (Wichmann, 2017; Mulholland et al., 2020).

Food production options on wet peatlands are limited for mid-latitude regions. Wet meadows and pasture can provide fodder and grazing for livestock production, and extensively managed grassland sites can have GHG balances close to semi-natural peatlands (Beetz et al., 2013). However, they have low biomass production rates and therefore only support low stocking rates, so profitability is generally low in the current economic landscape. Given enteric CH_4 emissions from ruminant livestock and generally high GHG emissions from meat production, there is a risk of substituting GHG emission sources if additional livestock production results from this change. The UK Committee on Climate Change recently factored reduced meat consumption into its projections of future land-use sector emissions (Stark et al., 2020), suggesting that changes which increase livestock production may not always be commensurate with wider societal goals. Few conventional crops are tolerant of flooded conditions. Paddy rice (*Oryza sativa*) production is well-established, suitable for fen peatlands and can reduce GHG emissions by ~75% relative to drained cropland (Knox et al., 2015). Its viable geographic range is climate dependent and water demands can be substantial, which currently constrains its applicability within the mid-latitudes.

Paludiculture adoption in the mid-latitudes would likely reduce food production on the peatland area. This is an important consideration, given global food security goals. Adoption also risks displacing food production elsewhere, entailing higher agrochemical or transportation burdens, and counterbalancing direct benefits. However, due to the finite lifespan of drained agricultural peatlands, their high productivity is only temporary. Sustainable intensification of agriculture on mineral soils, alongside demand-side measures and waste reductions will be essential to meet global food requirements (Springmann et al., 2018). Where possible, proactive peatland restoration and transition to mineral soil production would allow protection of peatland C stocks and retention of regulating/supporting ecosystem services of substantial value (Wichmann et al., 2016).

The limited available evidence suggests that paludiculture would generally result in lower agricultural profitability than conventional agriculture within the current economic landscape and so it will not be immediately viable in all circumstances (Wichmann, 2017; Mulholland et al., 2020). However, paludiculture offers an important strategy for mitigation of GHG emissions and the protection of peatland C stocks, without the complete loss of agricultural profitability associated with full restoration. With a favourable economic environment including efficient C markets for emissions reductions/sequestration and an adequate C price, such systems could play an important role in responsible peatland management (de Jong, 2020). The possibility also exists that peatlands could be managed as active C/GHG sinks (‘C farms’) based on paludiculture-type approaches without biomass harvest to maximise net-uptake (Element Energy and UKCEH, 2021). Development of functioning paludiculture demonstration systems could support identification of economic potential and development of appropriate GHG EFs at field and product level (Tanneberger et al., 2020). Delivering these research outcomes is vital to support decision-makers in both policy and industry to invest in the development of commercially viable paludiculture options.

2.4.2. Reduced drainage intensity systems

Where socio-economic circumstances are currently incompatible with restoration or paludiculture adoption, alternative wetland agriculture systems will be required. These would need to impose smaller reductions in the delivery of provisioning services but would require a compromise on their GHG mitigation potential. Reduced drainage intensity systems may deliver such a compromise, and would involve some combination of reducing the drainage depth, the duration of drainage and the area drained (Table 2.2). The aim would be to maximise the proportion of the peat layer at a site that is saturated over the course of the year and thus produce partial, but potentially significant, reductions in rates of subsidence and GHG emissions, whilst minimising impacts on agricultural production.

The evidence suggests that bringing the average WTD closer to the peat surface might incrementally reduce C-derived GHG emissions up to a WTD of ~0.05 m. Several mesocosm studies have confirmed that WTD reductions from 0.5 m to 0.3 m can significantly reduce CO_2 emissions from agricultural peat soils, without significantly increasing CH_4 emissions (Musarika et al., 2017; Matysek et al., 2019; Wen et al., 2020). Of these studies, only Wen et al. (2020) measured N_2O emissions, observing a 41% reduction. However, this study did not add N, which can exacerbate N_2O emissions in wet peatland systems (Kandel et al., 2019; Wen et al., 2021). The effect of WTD change on N_2O emissions from active agricultural peatlands remains poorly quantified and represents an important weakness in our understanding.

Table 2.2. Overview of relationships between conventional systems and wetland agriculture systems. There is a subdivision between paludiculture and reduced drainage intensity approaches in the extent of modification. Reductions in overall drainage intensity are separated into those decreasing the depth, duration or area of drainage. However, in practice some combination of these may also be used. These are broad characterisations intended to highlight differences. Water table depths and management practices on specific sites may be less clear-cut. BAU = Business as usual.

Category	Sub-category	Summer WTD	Winter WTD	Area under modified WTD	Land use on modified WTD area	Land use on unmodified WTD area
Conventional agriculture	Conventional agriculture	Deep drained (BAU)	Deep drained (BAU)	None	N/a	Conventional agriculture
Wetland agriculture	Paludiculture	Near-surface	Near-surface	Whole site	Paludiculture	N/a
	Reduced drainage depth	Intermediate	Intermediate	Whole site	Adapted agriculture	N/a
	Reduced drainage duration	Deep drained (BAU)	Near-surface	Whole site	Adapted agriculture	N/a
	Reduced drainage area	Near-surface	Near-surface	Part of site only	Paludiculture or restoration	Conventional agriculture

Current agricultural practices rely predominately upon crops that originated in dryland regions and are poorly suited to wet conditions. Varying, yield effects were observed in the aforementioned mesocosm studies for celery (-19%; *Apium* graveolens; Matysek et al., 2019), lettuce (-37%; *Lactuca* sativa; Wen et al., 2020) and radish (+33%; *Raphanus raphanistrum*; Musarika et al., 2017). Where yield decreases occur, they are likely to be economically significant and in the absence of external costs being borne by producers, either compensation (e.g. payments for avoided emissions) or income from the sale of C credits (for emission reductions) may be required to offset losses (Buschmann et al., 2020). Robust studies of the impact of WTD management on crop yield and quality are required to support decision-makers with implementation of reduced drainage depth systems on cropland. Alternative crops, which can tolerate wetter soil conditions may be necessary to adapt agriculture to reduced drainage depths (e.g. Cranberries and Blueberries – *Vaccinium* spp.; Abel, 2016).

Land-use change from cropland to grassland is an option to reduce the WTD to at most 0.5 m, whilst limiting reductions in agricultural profitability. Intensive grasslands do require drainage but shallower WTDs are possible under grass than most conventional crops. WTDs around 0.5 m support grassland biomass production (Campbell et al., 2015), whilst producing bearing capacities generally suitable for vehicle access (Schothorst, 1982). Consequently, such systems are already widespread but as noted above, GHG emissions resulting from any additional livestock production may offset reductions in peat-derived GHG emissions, potentially limiting the net benefits of such land-use change.

Long-term drainage ultimately causes sufficient loss (or 'wastage') of peat depth and SOM content that the soil no longer meets the definition of a histosol. In areas of Northern Europe that have been drained for centuries, such soils are widespread, and in some regions may comprise the majority of the agricultural 'peatland' area. Robust EFs for wasted peat soils represent an important knowledge gap, with studies indicating that CO_2 emissions decrease with declining SOM content (Taft et al., 2017), are similar to those from deeper peat soils (Tiemeyer et al., 2016) or are higher from lower SOM peat soils (Leiber-Sauheitl et al., 2014). This uncertainty extends to the scale, and nature of, mitigation measures required.

Reducing WTDs on agriculturally active wasted peatlands would be extremely challenging. Near-surface WTDs would be needed in order to saturate a substantial proportion of the remaining peat layer and wastage often reveals topographically uneven subsoils. One option for reducing drainage intensity on these sites may be to reduce drainage duration, and thus the time for which the peat layer is aerated. Approximately 23-41% of net CO_2 emissions occur during the winter (October – March; Evans et al., 2016b), when farm activity is reduced, potentially providing an opportunity to reduce drainage depths. Wen et al. (2020) observed 33% lower GHG emissions from mesocosms during the winter at a WTD of 0.3 m compared to 0.5 m. Arable production is possible on seasonally flooded peat in California (winter WTD up to -0.3 m), though viability may rely on adequate evapotranspiration rates, which will vary with climate (Anthony and Silver, 2021). Shallower winter WTDs may restrict vehicle access and interfere with field preparation. They could also lead to shifts in grassland plant community composition but this requires prolonged wet conditions and may be avoidable with appropriate management (Toogood and Joyce, 2009).

Where reductions in neither drainage depth nor intensity are deemed possible (e.g. wasted peat cropland in cool climates), reducing the area under drainage may be the only option to achieve substantial GHG emission reductions. This could be achieved by placing some portion of a site under rewetted conditions (e.g. paludiculture/restoration), whilst continuing conventional production on the remainder.

Reduced drainage intensity systems are likely to offer less GHG mitigation, but lead to smaller reductions in agricultural profitability, when compared to paludiculture in the current regulatory landscape. However, we currently lack robust field and experimental evidence of the effects of reduced drainage depth and duration on GHG emissions and crop yield/quality. Field trials at plot and field scale will be necessary to evaluate the benefits of these approaches, to allow the identification and resolution of any management issues, and to enable optimal management of the trade-off between GHG mitigation and economic productivity.

Development of reduced drainage intensity systems is an important step in producing robust wetland agriculture options for responsible management of mid-latitude peatlands. These options do not equate to truly sustainable management, because decomposition will continue in the remaining aerated peat layer. Therefore, they do not argue against peatland restoration or paludiculture adoption where these are suitable. However, reduced drainage intensity systems are closer to the status quo than restoration/paludiculture adoption, and may therefore meet with fewer practical, socio-economic and political barriers. If solutions to challenges can be found, they may therefore make a significant contribution to the overall mitigation of agricultural peatland GHG emission rates and provide a transitional option where circumstances preclude immediate restoration or paludiculture adoption.

2.4.3. Drainage and water resource management

The drainage infrastructure and technology required for management of wetland agriculture systems has not yet been fully developed and is of varying quality in different regions. The management of drainage and water resources is likely to pose challenges for wetland agriculture adoption at both field and regional scales. Reduced drainage intensity systems will require close regulation of WTDs within fields to minimise agricultural risks. Traditional drainage ditch networks were designed for removal of excess water in winter and are usually unsuitable for precise WTD control. Submerged drainage systems involve the installation of drainage pipes within the peat layer to improve drainage in winter and limit WTD drawdown in summer (Weideveld et al., 2021; Figure 2.4a/b). This can provide a more stable WTD over the course of the year, facilitating both sub-irrigation of crops and winter vehicle access. However, they may perform better in aiding drainage than in producing infiltration of water into the field (Hoving et al., 2008). Their performance can be enhanced by manipulating ditch water levels relative to the drain depth (dynamic WTD management; Hoving et al., 2013, 2015) and by using pumps to adjust water levels in wells attached to submerged drainage pipes (Jansen et al., 2017). However, pumping will incur energy costs and indirect GHG emissions.

Submerged drainage systems almost certainly offer a valuable tool for improvement of field-scale WTD control. It has been suggested that they can also reduce CO_2 emissions by as much as 50% (van den Akker and Hendriks, 2017). However, this is based on an assumption of a linear relationship between subsidence rates and CO_2 emissions and not direct GHG measurements (Couwenberg, 2018). A recent study suggests that submerged drainage alone does not lead to reductions in GHG emissions (Weideveld et al., 2021). We know of no studies that have measured GHG emissions whilst both (i) installing submerged drains and (ii) attempting to reduce the average annual WTD by simultaneously raising ditch levels relative to controls. Robust measurements of GHG emissions (including N_2O) under such experimental conditions will be essential to understanding the GHG mitigation potential of wetland agriculture systems.

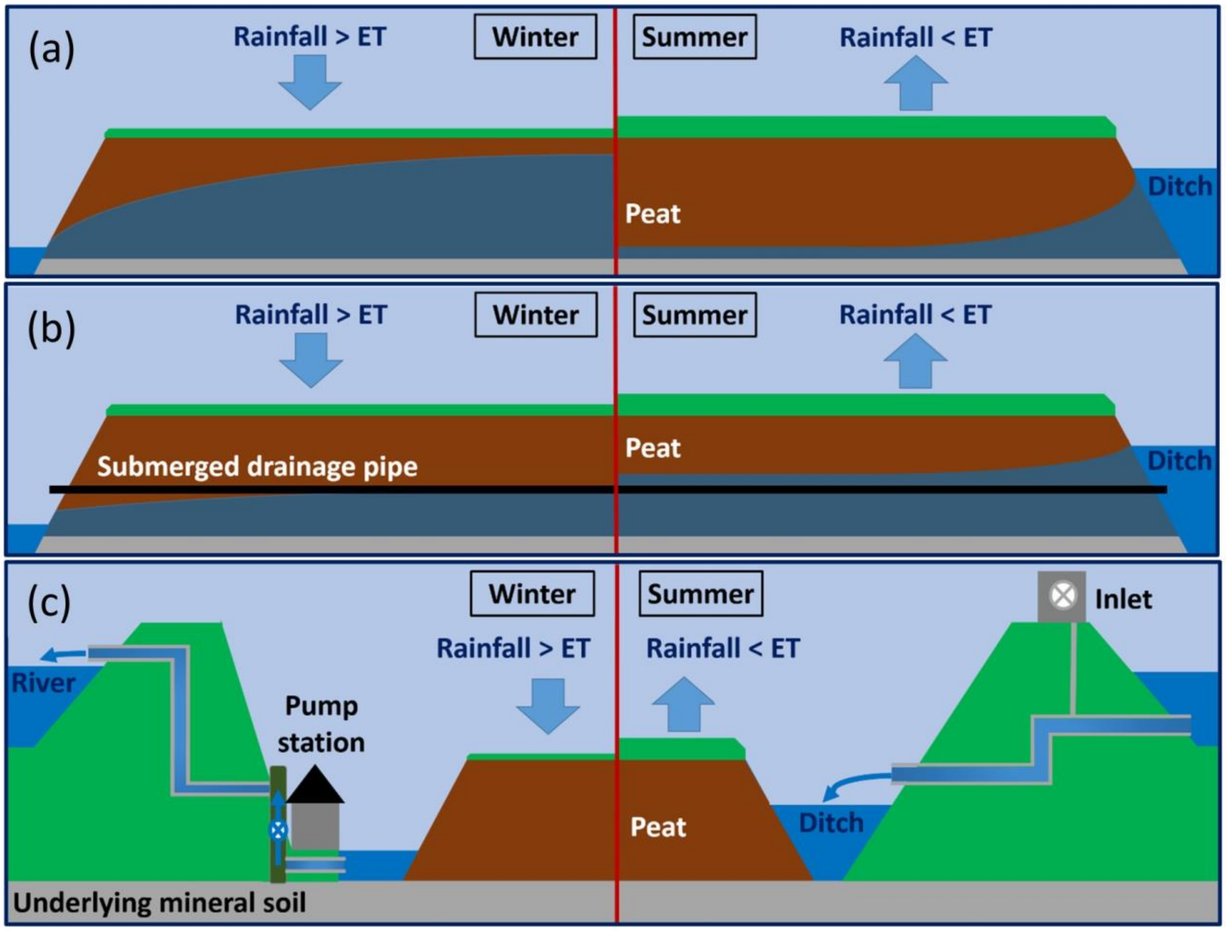

Figure 2.4. Seasonal water management in agricultural lowland peatlands. (a) Field water table conditions with drainage ditches alone, (b) Theoretical field water table conditions with submerged drains, (c) Drainage management on agricultural peatlands subject to extensive subsidence. In winter, water is pumped from ditches up to rivers, in order to drain the fields and limit flood risk. In summer, water is allowed to flow down from rivers to ditches to aid irrigation of the crop/sward. Sub-images a) and b) developed from Hoving et al. (2015).

Many mid-latitude agricultural peatland areas experience substantial seasonal variation in water availability. Subsidence means that many lowland peatlands lie below the level of local river channels and/or the sea, requiring major watercourses to be embanked in order to prevent river flooding. When regional precipitation exceeds the sum of evapotranspiration and available water storage capacity (e.g. in winter), surface flooding of fields will occur unless excess water is discharged by pumping (Fig. 4c). Drainage pumping would continue to be necessary under paludiculture (Mulholland et al., 2020) and reductions in free storage capacity may also necessitate more rapid pumping to keep pace with incident rainfall and control flood risk. The rate of discharge from fields themselves may also present a challenge for reduced drainage depth strategies. Drainage and infrastructure improvements at both field and regional scale would almost certainly be required for wetland agriculture adoption. A better understanding of the hydrology of these systems will be needed to assess the scale of the resulting risks and constraints.

During the summer, evapotranspiration exceeds rainfall and water supply to meet crop/sward demands becomes the dominant challenge on agricultural peatlands. Water can be released from embanked watercourses to supplement agricultural requirements (Figure 2.4c). However, high demand and relatively low supply can create short-term conditions of water stress even in regions where annual rainfall is abundant. These challenges would be likely to continue under wetland agriculture and additional water supply may be necessary (Querner et al., 2012). Reservoir construction may help to supplement the summer water supply by holding winter rainfall until it is needed and smoothing out seasonal trends in availability. However, the scale of infrastructure that would be required is unclear and may be substantial. Areas set aside for paludiculture, may form a dispersed network of emergency reservoirs; they could be managed for near-surface WTDs normally, with allowances for drawdown to supplement crop irrigation under drought conditions. These areas could also perform a flood regulation role by holding excess water during winter.

Inevitably, different rates of adoption of wetland agriculture systems would risk conflict between land managers (Buschmann et al., 2020). In practice, hydrological isolation of sites will be challenging, because water management in agricultural peatlands typically occurs at the large (i.e. multiple farm) scale. Bunds and impermeable membranes have been used to reduce lateral water movement onto/off wet sites within agricultural peatland landscapes. However, these are rarely fully effective, and WTDs on adjacent sites are likely to influence each other. Power imbalances may exist due to proximity to water inputs or drainage pumps, though this might be manageable to some extent using bypass channels or sluice gates (Ferré et al., 2019). Managing the resulting patchwork of systems with varying WTDs and requirements will require land manager cooperation, coordination through administrative/regulatory bodies at larger scales and state support for the landscape-scale infrastructure required. Regulators themselves will face challenges where conflict exists between various aspects of their role, suggesting that coherent policy will be essential to facilitate successful implementation.

Whilst the changes in drainage and water resource management required by wetland agriculture are substantial, they are currently poorly studied and there is a clear need to establish their efficacy and practicality. Our understanding of agricultural peatland hydrology is currently limited at both site and regional scales, but this is essential to understanding GHG mitigation potential in what are ultimately wetland systems. Addressing these limitations will be vital to evaluate the practical viability of establishing wetland agriculture systems, understand additional management challenges that may result from climate change and design appropriate land use policies.

2.4.4. Socio-economic considerations

Agricultural peatlands are anthropogenic landscapes, providing both human habitation and livelihoods. Consequently, adoption of responsible management strategies cannot be considered in isolation from socio-economic and political challenges (Table 2.3). Stakeholders experience a range of different pressures and exhibit differing preferences, resulting in value plurality and conflicting interests (Rawlins and Morris, 2010; Buschmann et al., 2020). Stakeholder analysis in the UK shows wide agreement on the important role of hydrological management on peatlands but a divergence on the importance of use (e.g. agriculture) and non-use (e.g. pro-environmental) values (Rawlins and Morris, 2010). This, along with the challenges of using peatlands without depleting them, has resulted in the current polarisation between heavily drained agricultural systems and wetland conservation/restoration sites. Wetland agriculture systems represent a compromise, balancing economic productivity and pro-environmental outcomes, through a focus on improved hydrological management.

Table 2.3. Socio-economic challenges facing wetland agriculture adoption on mid-latitude peatlands. C = carbon, WTD = water table depth. Sources: 1) Ferré et al. (2019), 2) Schaller et al. (2011), 3) Buschmann et al. (2020), 4) Reed et al. (2020), 5) Mulholland et al. (2020), 6) Rawlins and Morris (2010), 7) Regina et al. (2016), 8) Bonn et al. (2014).

Challenge	**Details**
Opportunity costs	Agricultural use can be highly profitable[1,2]. Where wetland agriculture is less profitable, this represents a loss if income is not replaced[2,3]. Conversion costs borne by land managers cannot be invested in future productivity gains[2]. Where changes are irreversible, perceived opportunity costs may be substantial[4].
Uncertain time horizons	Remaining lifespans of agricultural peatlands vary between sites and are often uncertain[1]. Uncertainty and poor visibility of soil loss rates on deep peat may affect perceptions of the urgency of response required[1].
Uncertain costs of business as usual	Underlying mineral soils are variable and define the income generating potential after peat loss[1]. Expectations of future yield enhancing technologies may mitigate concerns about transitioning to less productive underlying soils, reducing the perceived costs of continuing current practice.
Regional cost-benefit disparities	Spatial variation in productivity and C stocks will cause spatial variation in cost-benefit assessments around adoption. For example, on highly productive systems with low remaining C stocks, the costs of offsetting production losses may outweigh the perceived benefits of adoption[3].
Cultural identities	In many areas, agricultural use is long established and local communities have invested heavily in building rural economies[1,4]. Pride in local culture and traditions may favour agricultural solutions, and impede adoption of externally imposed novel solutions[4].
Stakeholder networks	Highly connected networks, including both scientific expertise and local actors positioned to implement solutions, appear to enhance potential for adoption[2]. Poorly connected networks are associated with low acceptance and potential for conflict[2].
Stakeholder conflict	Local conflict may arise when adoption affects water levels on neighbouring land. Land use heterogeneity and high productivity can increase conflict potential[3,5]. Larger-scale conflict may arise over the importance of production and pro-environmental ecosystem services[6].
Economic pressures	Agricultural producers face pressures from retailers on both the quantity and timeframe of production[1]. Producers unable to meet these demands under less productive/reliable systems may lose contracts or favourable terms, exacerbating profitability reductions.

Economic competition	Reduced production potential under wetland agriculture may diminish the comparative advantage of peatland production and expose producers to competition with mineral soil producers, leading to loss of market share and reducing profitabilty[2].
Perceived locus of control and self-efficacy	Perceptions of control and capacity are important precursors to pro-environmental behaviours. Prescriptive, top-down policy may reduce perceived control, whilst uncertainty around capacity for implementation may present an obstacle to adoption[4].
Information availability/quality	Research is not always produced and communicated with the aim of providing useable information to end users[6]. This effectively creates an information deficit, which may be exacerbated by low levels of trust towards researchers[4].
Policy coherence	Implementation of mitigation measures can be impeded when national laws and land use/agricultural policies are not aligned with international/national climate policy[3,6,7].
Incentivising mechanisms	There are currently few schemes formally incentivising reduced WTDs on agricultural peatlands for climate mitigation and provision of public goods[1]. Longer-term schemes will be required to ensure persistence of WTD changes and provide security[3].
Quantification of public goods	Lack of robust valuations for regulating, supporting and cultural ecosystem services delivered under mitigation measures leaves decision makers reliant upon incomplete information[6].
C market development	Peatland emissions are generally not eligible for compliance markets[1,8]. Commodity C prices required to offset opportunity costs are often higher than current scheme prices[1].
Indirect land-use change impacts	Productivity declines associated with adoption may lead to production being exported to or intensified in other areas[2]. Negative environmental effects elsewhere may therefore offset local benefits and generate resistance from relevant stakeholder groups[1,2,5].

Economic pressures will inevitably make short-term costs of wetland agriculture adoption highly salient for land managers (Ferré et al., 2019). Contrastingly, the longer-term economic costs of eventual peat loss are often unclear and quantification of the benefits of non-production ecosystem services is notoriously challenging (Rawlins and Morris, 2010; Ferré et al., 2019). Furthermore, the wider benefits of these ecosystem services often occur at a societal level (e.g. flood regulation, water supply, landscape value and climate regulation) and are external to the land manager (Reed et al., 2014). The information deficit around peat losses/ecosystem service benefits is clearly an impediment to the development and adoption of wetland agriculture systems. In this context, hesitancy by decision-makers is understandable, if not ideal given the urgency of response required.

The research community has an important role to play in overcoming this deficit. However, this will be impeded by low levels of trust towards researchers, who may not share land manager values (Reed et al., 2020). For example, substantial differences exist between solutions that researchers deem effective and those that land managers deem practical or economic (Taft, 2014). Strongly connected stakeholder networks, including both sources of scientific knowledge and local knowledge are associated with higher potential for adoption of wetland agriculture systems (Schaller et al., 2011). This suggests that researchers need to continue raising awareness about the unsustainability of drainage-based, conventional agriculture on peatlands and the importance of WTD management. However, the successful development of a robust wetland agriculture paradigm will also require participatory research approaches, and the co-creation of knowledge and workable solutions, alongside land managers.

The opportunity costs and initial capital investments required for wetland agriculture adoption mean that land managers will be reliant on changes in the economic landscape and an adequate C price to ensure economic viability (de Jong, 2020). Remuneration for delivery of public goods and pro-environmental outcomes will be essential to support continued production where agricultural profitability is reduced. Private sector funding has the potential to play an important role, through corporate social responsibility and voluntary C markets. Land managers who can demonstrate verifiable reductions in (or cessation of) GHG emissions from peatlands may be well placed to secure private investment to support mitigation measures. However, successful harnessing of this capital pool will depend on the development of attractive, robust schemes delivering quantifiable, secure and cost-effective benefits (Bonn et al., 2014). The main challenges to private sector finance inflows will be issues of cost-effectiveness, permanence, leakage, additionality, handling of co-benefits and the fact that many schemes focus on enhanced removals rather than avoided emissions. Viability will depend strongly upon the C commodity price (Ferré et al., 2019). C sequestration achieved through land-use change will always be open to future reversals, so payments may need to focus on avoided emissions as opposed to removals (Evans et al., 2020). Displacement of food production will also carry risks of displacing environmental impacts elsewhere (Evans et al., 2020). Legislatively compelled rewetting would not be suitable for private finance schemes and care will be needed to ensure that co-benefits (e.g. water quality or flood regulation) are adequately accounted for, through either service bundling or payment layering, so that land managers receive fair remuneration (Bonn et al., 2014). Development of regional C markets such as MoorFutures® in Germany and the Peatland Code in the UK show promise (Bonn et al., 2014). However, work is still needed to strengthen the evidence base (Evans et al., 2014) and ensure sufficient regulatory support (Bonn et al., 2014).

Governments may have a role to play in scenarios that are not cost-effective for private C markets (due to high implementation costs/limited benefits). They may also be required to regulate private schemes and provide longer-term security to land managers who will be bearing outsized personal risks in order to produce public goods (Buschmann et al., 2020; Reed et al., 2020). There will likely be a need for some pro-environmental regulation (e.g. on maximum baseline WTDs) to ensure adequate improvements in environmental outcomes. However, there is a balance to be struck, as excessive regulation or government financial support will impede private sector funding (Bonn et al., 2014). Regulation could be balanced by incentivising/support mechanisms; recognising that much drainage occurred before the environmental consequences were understood and supporting land managers to be active participants in the solution. Future emissions from drained peatlands could be seen as the responsibility of land managers who continue drainage-based management and the cost of these emissions could be recovered under the 'polluter pays principle'. This would be politically polarising but would fundamentally shift the economic landscape. A UK analysis balancing agricultural income against the cost of GHG emissions (and payments for net sequestration) indicated that peatland restoration and extensive grassland adoption could offer net benefits of ~£650 GBP ha^{-1} yr^{-1} on average over continued intensive arable production (2012 prices; Graves and Morris, 2013). These values would likely be lower on the most productive sites but could be enhanced by inclusion of payments for other ecosystem services and would favour wetland agriculture adoption and peatland restoration. Additionally, reduction or cessation of subsidy for drainage-based agriculture may incentivise adoption of pro-environmental alternatives (Ziegler et al. 2021). Decisions on how the costs of peatland GHG emissions are treated (penalties vs. payments for avoided emissions) and allocated (to land managers, governments and/or private investors) will be a major driver of eventual outcomes. Land use planning will require clear, robust decision-making frameworks (Figure 2.5), accurate spatial data and flexible responses to regional differences (Kekkonen et al., 2019). Developing a coherent policy framework (aligning goals between different policy areas and levels of governance) and ensuring the absence of legislative obstacles will be key to efficient implementation (Regina et al., 2016; Buschmann et al., 2020).

The socio-economic analysis presented draws strongly from European sources due to a geographical publication bias, which in turn reflects the relatively widespread and longer-term drainage and cultivation of European peatlands compared to most other mid-latitude regions. The relative profitability of agricultural peatlands, availability of subsidies/private funding flows and viability of different regulatory strategies will vary substantially across mid-latitude regions. Local (national or regional) solutions will need to be found that suit the circumstances, character and values of each region. Land managers will ultimately be responsible for implementing wetland agriculture strategies. However, it is clear that researchers, policymakers and private investors will all need to play key roles if wetland agriculture adoption is to be sufficiently successful and widespread to deliver a meaningful contribution to meeting climate goals. Given the complexity and urgency of the challenge, and the need to navigate risks and uncertainties, there is a real need for compromise, collaboration and cooperation between stakeholder groups, to ensure positive outcomes.

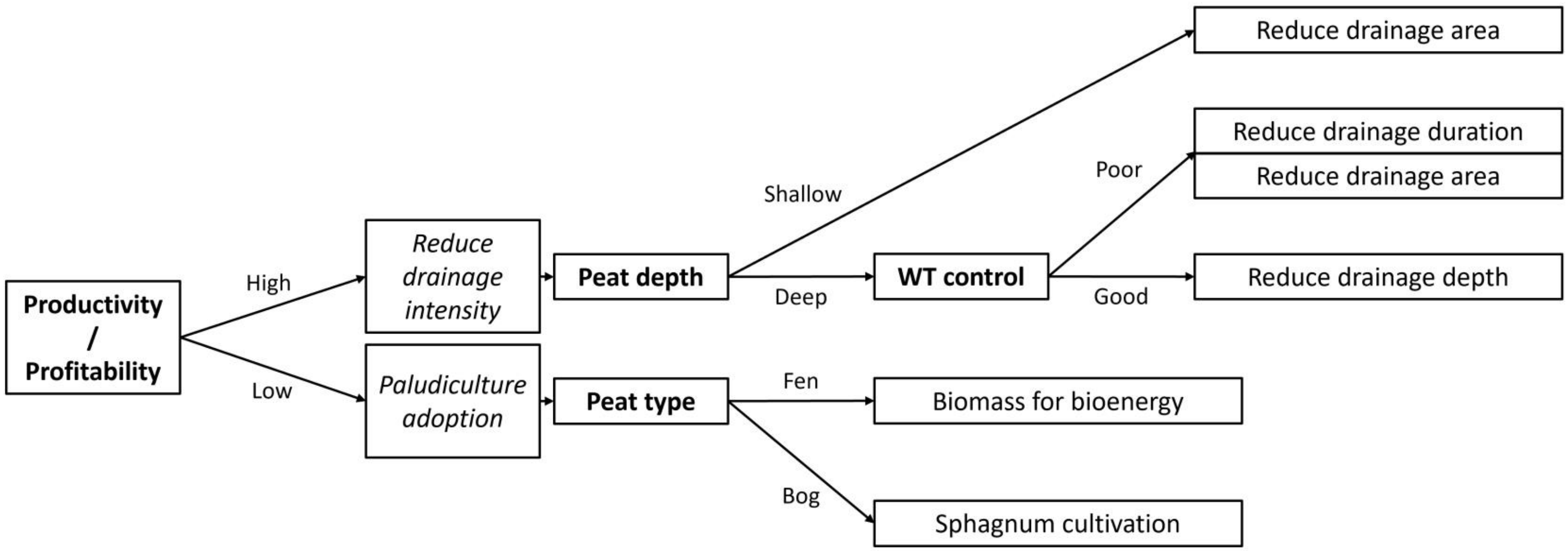

Figure 2.5. Example decision-making tree for wetland agriculture adoption. This is based on the approach of Kekkonen et al. (2019), who demonstrated that accurate spatial data combined with an appropriate decision-making tree could provide a practical tool for land use planning. In practice, the decision criteria selected and boundaries between classifications (e.g. deep/shallow peat) would need to be defined appropriately for the physical and socio-economic conditions of the nation/region in question. Those chosen here, along with the options presented in the right-hand column of boxes do not represent an exhaustive list and are presented for illustrative purposes only. Bold text indicates decision criteria. Italics indicate wetland agriculture sub-categories. Peatland restoration (not shown) would also be an essential component of wider responsible peatland management strategies. WT = Water table.

2.5. Conclusions

Application of conventional, drainage-based agricultural practices to mid-latitude lowland peatlands has produced highly productive agroecosystems. However, these systems are losing irrecoverable C and represent a disproportionately large source of GHG emissions relative to the area they occupy. There is widespread agreement that substantial reductions in GHG emissions will be required in order to mitigate potential harms stemming from global climate change. Emission reduction pressures will inevitably be focused on hotspot sectors and consequently, both international and national climate policies are driving efforts to balance productivity with improved environmental outcomes for agricultural lowland peatlands. Peatlands are naturally wet systems and as a result, hydrology is fundamental to their function and management. Maximising progress towards GHG emission targets will require management for peatlands with near-surface WTDs, suppressing rates of SOM mineralisation and CO_2 emissions, which account for the bulk of GHG emissions from agricultural peatlands. Socio-economic constraints mean that full rewetting/restoration will not always be immediately possible. Paludiculture appears to be a highly effective strategy for GHG mitigation but it is unclear at what scale it can be economically viable in the current economic/regulatory landscape. Consequently, development of reduced drainage intensity systems, involving compromise between food production and climate mitigation, will also be important to help minimise climate impacts until societal/economic conditions favour strategies that can completely halt emissions. Smaller reductions in agricultural profitability (relative to completely rewetted systems) will be needed in recognition of the constraints faced by land managers in the current economic landscape, in order to facilitate the delivery of admittedly more limited reductions in rates of GHG emissions per unit area over the large areas required to produce meaningful change.

Wetland agriculture adoption will require a range of agronomic, hydrological and socio-economic challenges to be overcome and inevitably presents risks to land managers. The viability of the different options identified is not yet clear. Therefore, assessments of viability and adoption risks represent important priorities for research, which must focus on the creation of workable solutions and avoid over simplistic idealism in the face of complex realities. The need for change creates uncertainty, which is exacerbated by the urgency of the response required by climate change and creates a challenging environment for land managers and policymakers alike. However, it is easy to focus on the risks and dismiss the opportunities. The development and implementation of well-designed wetland agriculture strategies could present an opportunity to create more resilient production systems, improve delivery of wider environmental benefits and protect valuable peatland ecosystems for everyone's future benefit. Combining wetland agriculture systems with restored semi-natural sites, constructed wetlands (e.g. for water treatment) and renewable energy systems (e.g. solar/wind), could allow us to create thriving, innovative and green wetland landscapes; delivering a vital contribution to global climate change mitigation efforts.

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Chapter 3

Farm management activities affect soil vulnerability to wind erosion in lowland agricultural peatlands.

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BWJF conducted the investigation and analysis, and wrote the manuscript. GFSW, CDE, DRC and DLJ reviewed the manuscript.

Abstract

Peatlands store >600 Gt of carbon (C) but following drainage for agriculture peatland C becomes vulnerable. Agriculturally managed peatlands are a known carbon dioxide emission hotspot but the contribution of wind erosion to C losses from agricultural peatlands is still poorly understood. Aeolian C fluxes from peat soils exhibit high spatiotemporal variability, which reflects complex interactions between environmental, physical and anthropogenic drivers. Mechanistic studies of erosion processes on peat soils to date have all used artificially prepared surfaces. Therefore, we undertook a field measurement campaign, in the East Anglian Fens, UK, using a Portable *In-Situ* Wind Erosion Laboratory (PI-SWERL), to examine how the interactions between management practices and soil physical properties affect the PM_{10}-C (particulate matter <10 μm) emission potential of bare peat surfaces. We sampled a range of management stages and soil organic matter (SOM) content soils, to evaluate the influences of tillage, crop establishment practices and vehicle traffic on erosion vulnerability, and the potential for SOM to mediate these effects. The proportion of wind erodible material adjusted for sample density was the best overall predictor of erosion vulnerability. Vulnerability to erosion generally increased throughout the crop management cycle in line with an increase in the proportion of wind erodible material. The vulnerability of tilled soils depended strongly on SOM content, with tilled high-SOM peat soils demonstrating high erosion vulnerability, whilst tilled lower-SOM content surfaces were not vulnerable to wind erosion. Crop establishment practices produced high wind erosion vulnerability across all SOM contents in the early cropped period. Areas subjected to high levels of vehicle traffic also represent potentially important erosion vulnerability hotspots. Mitigation efforts should focus on minimising periods of bare soil and maximising vegetation/residue cover of soil surfaces. Current farm management practices can result in high vulnerability of agricultural peatland C to wind erosion but the potential for mitigation is promising.

3.1. Introduction

An estimated 2000 Mt of dust are emitted to the atmosphere each year with 10-60% of this attributed to anthropogenic processes (Shao et al., 2011; Webb and Pierre, 2018). Dust emissions resulting from wind erosion of soil can have profound effects on climate regulation (Kok et al., 2017; Schepanski, 2018; Kok et al., 2023), nutrient cycling and ecology (Lawrence et al., 2013; Brahney et al., 2015; Dansie et al., 2022), food security (Lal, 2007; Amundson et al., 2015; Webb et al., 2017) and human health (Griffin et al., 2001; Prospero et al., 2014; Staffogia et al., 2016). The majority of global dust emissions arise from desert regions and are primarily natural (Kok et al., 2023) but dust emissions from agricultural soils also make an important contribution; significantly, one which can be influenced by management choices (Mulitza et al., 2010; Stanelle et al., 2014). Through reductions in vegetation cover and the application of tillage, agricultural management has the potential to substantially increase the vulnerability of soil to wind erosion (Houyou et al., 2016; Bartkowski et al., 2022). The most notorious example of this was the 1930's dustbowl on the Great Plains of North America, where in combination with drought and economic pressures, dryland agricultural practices led to the loss of an estimated 480 tons of topsoil per acre between 1930 and 1938 (Hansen and Libecap, 2003; Lee and Gill, 2015). Wind erosion of agricultural soils remains an issue today, with 64% of European cropland estimated to be eroding to some extent (Borelli et al., 2017).

Only 0.5% of global agricultural land is on peat soils but these areas are disproportionately vulnerable to carbon (C) loss (FAO, 2022, 2023). Peatlands as a whole, represent a globally important C store (>600 Gt C; Yu et al., 2010). However, 11% of the global peatland area has been drained for productive use (Joosten, 2010), which aerates the soil, leading to mineralisation of soil organic matter (SOM) and increased nitrogen cycling rates (Freeman et al., 2022). As a result, drained peatlands emit an estimated 1.91 Gt CO_2-e yr^{-1} and approximately 14% of the peatland C stock is estimated to be degrading (Leifeld and Menichetti, 2018). Whilst greenhouse gas (GHG) emissions from agriculturally managed peatlands are relatively well characterised, the contribution of aeolian processes to overall C losses from agricultural peatlands is still poorly understood (Freeman et al., 2022). Substantial aeolian C losses have been observed from agricultural peatlands but they appear to exhibit high spatiotemporal variability (Parent et al., 1982). Field monitoring studies using passive slot samplers have produced estimates of horizontal mass transport ranging from 230-1880 g m^{-2} yr^{-1} through a vertical plane up to 1 m above thick UK agricultural peat and

880-1400 g m^{-2} yr^{-1} up to 2 m above very thin, degraded ('wasted') peat (Cumming, 2018; Newman, 2022). These studies highlighted the correlation of peak horizontal mass fluxes with periods of high wind speeds and bare soil in the spring planting season. However, in the absence of estimates of vertical dust fluxes and deposition rates, net gains/losses from agricultural peatlands due to aeolian processes remain unclear. Warburton (2003) estimated an erosion rate of 0.46-0.48 t ha^{-1} for an isolated patch of bare upland peat in the UK. Wind-splash erosion by wind-driven rain was the primary erosive process but surface dust deflation was also important during dry periods with surface peat desiccation (Warburton, 2003; Foulds and Warburton, 2007). To the best of our knowledge, this remains the only published estimate of wind erosion rates for peat soil and this represents a significant knowledge gap.

Wind erosion results when air movement across a surface produces a shear stress sufficient to overcome the cohesive and gravitational forces acting on surface particles, inducing their detachment and transport before subsequent redeposition at another location. Erosion rates are therefore strongly influenced by factors affecting air movement across a surface and factors affecting the vulnerability of particles to detachment (Chepil, 1945). In the lowland agricultural peatland context, the distribution of vegetation within the landscape is the dominant controllable factor influencing air movement. The presence of trees (e.g. as shelter belts) can substantially reduce wind speed and erosion rates (Leenders et al., 2016; Řeháček et al., 2017; Chang et al., 2021), whilst low vegetation providing approximately 40% ground cover is typically sufficient to inhibit wind erosion (Funk and Engel, 2015). However, current agricultural practices often create large, exposed fields, and produce periods of bare soil or sparse vegetation cover, which coincide with the most erosive climatic conditions in the spring (Funk and Engel, 2015; Mendez and Buschiazzo, 2015; Newman, 2022). Consequently, during these windows of raised erosion vulnerability, soil physical properties including the aggregate size distribution (ASD), surface roughness, surface crusting, moisture content and particle density may play important roles in mediating wind erosion losses from lowland agricultural peat soils (Campbell et al., 2002; Kohake et al., 2010; Zobeck et al., 2013).

The mechanical action of farm machinery and associated abrasion of soil aggregates resulting from tillage practices can alter soil ASDs (Lyles and Woodruff, 1962; Zobeck and Popham, 1990). Conventional tillage has been shown to reduce mean aggregate size (MAS) and increase the proportion of wind erodible material (WEM) when compared to no-till practices (Yang and

Wander, 1998; Hevia et al., 2007). Tillage can increase surface roughness through the introduction of furrows (larger-scale roughness elements) and induction of clod (larger non-erodible aggregate) formation (Fryrear 1999; Masri, 2015; Wiggs and Holmes, 2011). However, the effects are dependent on tillage method/intensity, and secondary tillage can reduce surface roughness by destroying clods (Zobeck and Popham, 1990; Masri et al., 2015; Riegler-Nurscher et al., 2020). Vehicle traffic can cause compaction, and can also generate substantial erosive forces through the action of tyres/tracks and by creating aerodynamic turbulence (Pinnick et al., 1985; Kuhns et al., 2010; Destain et al., 2016). Goosens et al. (2001) found dust emissions from agricultural fields associated with periods of tillage and vehicle activity were 6.6 times higher than emissions associated with periods of wind erosion alone. Similarly, Cui et al. (2019a) found that PM_{10} emissions from unpaved roads in China were approximately double those from arable land, suggesting bare soil subject to heavy traffic may be highly vulnerable to erosion. Both tillage and vehicle traffic represent anthropogenic influences with the potential to increase the vulnerability of agricultural peat soils to erosion.

Irrigation is also an important component of salad crop establishment and management practices. The hydraulic action of irrigation can cause compaction and surface crusting of agricultural soils, whilst influencing the ASD and soil moisture content (Pilatti et al., 2006; Hondebrink et al., 2017; Laker et al., 2019). It is well established that increasing soil moisture content increases the threshold friction velocity of entrainment ($u*_t$) and decreases wind erosion fluxes by increasing the mass of soil particles and the cohesive forces between them (Chepil, 1956; Selah and Fryrear, 1995; Chen et al., 1996; Wiggs et al., 2004; Yuge and Anan, 2019). However, sufficient increases in soil moisture to prevent erosion of the active surface layer (top 2 mm of soil) following wetting can be short-lived (Bergametti et al., 2016). The surface of sandy soils can air dry within 1-2 hours, whilst heavier soils can air dry within a few days (Ravi et al., 2006). Therefore, apart from when the timing of irrigation is coincident with erosive climatic conditions, the main impact of irrigation on erodibility of agricultural soils is likely to be via effects on soil structure. Kohake et al. (2010) exposed agricultural peat soils to simulated rainfall and observed that surface roughness and MAS decreased, whilst the proportion of WEM increased in all cases. Abrasion coefficients of the crusts formed after simulated rainfall were also determined to be higher than aggregate abrasion coefficients, indicating that crust formation increased the vulnerability of peat soils to erosion. Contrastingly, Campbell et al. (2002) found very low erosion rates from crusted milled peat soil following simulated rainfall. However, the simulated rainfall methods differed markedly between these two studies. Campbell et al. (2002, p. 86) applied 2.8 mm of water as a "fine mist", whilst Kohake et al. (2010) used a 10 m high rainfall tower to simulate 32 mm of rainfall with average drop size of 1.69 mm and kinetic energy similar to natural rainfall. The effects of irrigation are likely to depend on specific practices but the results of Kohake et al. (2010) suggest that irrigation and natural rainfall have the potential to substantially increase the erosion vulnerability of agricultural peat soils.

Across lowland agricultural peatland areas SOM content can exhibit substantial spatial variation because of differences in drainage history and associated SOM mineralisation. The SOM content of peat soil is inversely related to particle density (Zobeck et al., 2013), so the equivalent diameter of WEM (0.85 µm for mineral material) is larger for soils with higher SOM content (Chepil and Woodruff, 1963). Consequently, for a given ASD, a higher-SOM peat soil will have a higher proportion of WEM; a higher proportion of the soil aggregates will be erodible. The density-adjusted proportion of WEM has previously been shown to be an excellent predictor of wind erosion losses for milled peat soils (Campbell et al., 2002). Kohake et al. (2010) found a negative relationship between MAS and soil loss, whilst Zobeck et al. (2013) found PM_{10} emissions to be predicted by SOM content for agricultural peat soils. Whilst these three studies all found slightly different relationships and measured different variables, the general conclusion is that SOM content, through its influence on particle density, may be an important factor influencing the erodibility of peat soils. It is also worth noting that there may be indirect, interactive effects of SOM on erodibility to the extent that it modulates the effects of mechanical/hydraulic action on soil structure. The effects of tillage on soil ASDs have been found to depend on soil texture in mineral soils, suggesting there is potential for SOM content to mediate tillage effects in peat soils (Fryrear, 1995). Similarly, in mineral soils, increasing SOM is generally associated with maintenance of larger MAS and greater resistance to surface sealing following irrigation (Zhang et al., 1997; Ozlu et al., 2019; Grandinetti et al., 2022). Extrapolating these effects to the very high SOM content of some peat soils would be unwise. However, the presence of these effects in mineral soils and the high proportion of SOM in peat soils suggests that the potential for such effects warrants further investigation.

Spatiotemporal variability in erosion rates from agricultural peat soils likely reflects complex interactions between numerous environmental, physical and anthropogenic factors which combine to produce observed erosion rates. Erosion fluxes from peatlands remain poorly characterised, and mechanistic studies of erosion processes on peat soils to date have all used artificially prepared surfaces. Therefore, we undertook a field measurement campaign to examine how the interactions between management practices and soil physical properties affect the dust emission potential of lowland agricultural peat soils, under field conditions during periods of minimal vegetation cover. We aimed to capture a range of management stages to evaluate the potential influences of tillage, crop establishment practices and vehicle traffic on erosion vulnerability. We also sampled across a wide range of SOM contents to evaluate the potential for SOM to mediate management effects on erodibility. Based on our results we seek to identify the key factors leading to spatiotemporal peaks of erosion vulnerability, and to guide the development of mitigation strategies to protect these valuable soil resources.

3.2. Study area

The East Anglian Fens represent the largest area of lowland peatland in the United Kingdom (UK). The Fens experience ~600 mm of mean annual rainfall, a lowest mean monthly minimum temperature of 1.5 °C and highest mean monthly maximum of 23.0 °C (1990-2020 averages for Mepal; UK Met Office 2023). The average monthly wind speed for East Anglia at a height of 10 m is 4.2 m s^{-1} (1990-2020 average, UK Met Office 2023). The prevailing wind direction is from the SW, and wind speeds are highest and most variable during the winter and spring (Newman, 2022). The Fens were formed as the result of raised and fluctuating sea levels following the last glacial period. Repeated marine flooding formed silt and clay deposits, which impeded drainage of the freshwater rivers of the Fen Basin, producing alluvial and lacustrine deposits of sand, silt, gravel and marl. The resulting wet, anoxic conditions were also highly favourable for peat formation and it is likely that there were once ~1,500 km^{-2} of peat soil in the Fens, with large areas of peat >5 m in depth (Hutchinson, 1980; Waller and Kirby, 2021). However, since the 17th century there has been a progressively intensifying program of drainage in the region, predominantly for agricultural purposes (Sly, 2010). Consequently, the area has experienced significant subsidence and wastage, and the remaining area of peat soil >40 cm has been estimated at 165 km^{-2} (Hutchinson, 1980; Holman, 2009), with much of the former peat soil area now ‘wasted’ peat (<40 cm in depth, typically intermixed with mineral soil) or entirely lost. The Fens are a highly productive agricultural landscape, accounting for ~7% of England’s total agricultural production and supporting a combined agricultural and food chain industry worth £2.3 billion (NFU, 2019). However, agricultural soils in the Fens can be highly vulnerable to wind erosion and severe episodic erosion events known as ‘Fen blows’ are well documented (Thompson, 1957; Borelli et al., 2017). Chappell and Warren (2003) observed average and maximum erosion rates of 0.6 t ha^{-1} yr^{-1} and 32.6 t ha^{-1} yr^{-1} respectively for mineral soils in the region, finding that most erosion was spatially concentrated in hotspots and balanced by equivalent areas of deposition.

Within this regional context, our study sites comprised three large arable farms with varying SOM content peat soils (Figure 3.1). Rosedene Farm (52°32’06’’N 0°27’27’’E) is situated on thick peat (1-2 m) with high topsoil SOM content ~65%. Redmere Farm (52°26’39’’N 0°24’33’’E) represents thin peat, predominantly <1 m, with intermediate topsoil SOM content ~45%. Plantation Farm (52°28’11’’N 0°21’48’’E) represents predominantly wasted peat soils <0.4 m deep with low topsoil SOM content ~30%. All farms grow a range of crops including lettuce, celery, potatoes and wheat. Rosedene Farm also produces several other vegetable crops including onions, leeks, beets and radishes, whilst Redmere and Plantation Farms have land allocated to maize production. At Redmere and Plantation Farms, the fields are generally large and mostly divided by drainage ditches (Figure 3.2e), with few hedges or tree shelter belts. Erosion mitigation is implemented through the use of cover crops, retention of crop residues on the field surface after harvest (Figure 3.2i) and minimisation of the time interval between tillage and planting. At Rosedene Farm, due to the lighter soil, field sizes are kept substantially smaller and fields are often divided by hedgerows/shelterbelts (Figure 3.2l). The use of companion crops alongside juvenile cash crops and fleecing of prepared planting beds are implemented as additional measures to mitigate erosion losses (Figure 3.2d).

Fields under salad vegetable crops are especially vulnerable to wind erosion in the early stages of crop growth when ground cover is limited due to the small size of juvenile crop plants (Figure 3.2a). Crop establishment first involves primary tillage operations to integrate the previous cash/cover crop into the topsoil (Figure 3.2c). The soil is then harrowed (secondary tillage; Figure 3.2d) to create finer-textured planting beds before nursery-grown plug-plants are planted. Crops are sprayed and irrigated frequently during the growing period, to provide water, nutrients and protection from pests (Figure 3.2g; Figure 3.2h). There is a period of potential vulnerability to wind erosion extending from tillage through to the time when the established crop provides sufficient ground cover to limit erosion (Figure 3.2b). Salad crop planting times are staggered across the farm to ensure stable production rates and minimise waste. However, this period of potential vulnerability generally occurs during the spring, which coincides with higher wind speeds (Newman, 2022). A second crop is often possible on higher-SOM peat soils which may create a second period of potential vulnerability in summer, though wind speeds at this time tend to be lower. Harvest can be mechanised or manual depending on the crop, with many salad crops packed in the field. Following harvest, crop residue remains on the field surface (Figure 3.2i; Figure 3.2j). However, areas of bare soil result from heavy vehicle activity in the field. Vehicle access routes (farm tracks and field headlands) and harvest working areas experience high levels of traffic and may consequently represent within-farm/field erosion hotspots (Figure 3.2f). The wide range of SOM contents and relatively consistent management practices across our study sites provided an excellent context in which to evaluate the influence of these factors on the vulnerability of agricultural peat soils to wind erosion.

Figure 3.1. Location of study sites. Main image shows the study region with the study sites (Redmere, Rosedene and Plantation Farms) indicated by markers. The main watercourses and settlements in the region are also indicated using dark blue and grey highlighting respectively. The inset indicates the location of the study area within the United Kingdom. Source: Google Earth v9.191.0.0 WebAssembly, Accessed: July 2023, https://earth.google.com/web/

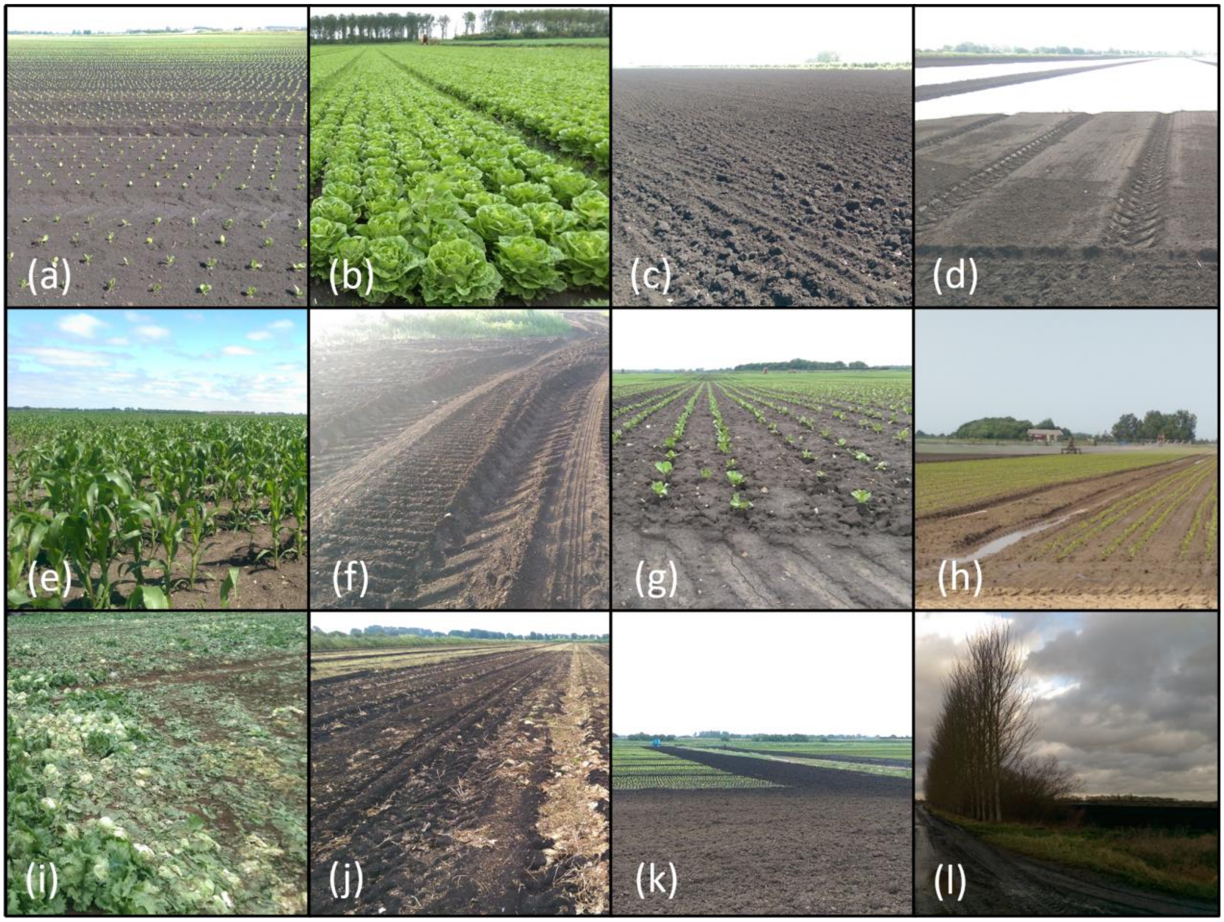

Figure 3.2. Study context. (a) Exposed bare soil between juvenile lettuce crop plants. (b) Mature lettuce crop plants provide substantial ground cover. (c) Primary tillage produces furrowed bare soil. (d) Secondary tillage produces smooth fine textured planting beds – note fleece covering to mitigate erosion. (e) Large fields on lower organic matter peat soils in the region have few trees in the landscape. (f) Vehicle traffic on fields can significantly modify soil surface properties. (g) Irrigation creates a surface crust. (h) Irrigation inputs can be substantial and cause strong wetting of surface soil – note surface ponding in tramlines. (i) Crop residue can provide substantial surface cover immediately after harvest. (j) Decomposition of crop residue with time can leave bare soil exposed. (k) Soil organic matter content can be highly spatially variable – note the colour change from light foreground soil with lower organic matter content to dark background thicker peat soil. (l) Tree shelterbelts are used by some farmers to provide protection from wind erosion.

3.3. Methods

3.3.1. Selection of study surfaces

Study surfaces (Figure 3.3) were selected from across the three farms in order to fulfill our objectives, and capture both a range of management stages and SOM contents. We considered four stages of the farm management cycle which result in bare soil or very sparse vegetation cover: Primary tillage (PT), Secondary tillage (ST), Early cropped (EC) and Heavy traffic (HT). The study surfaces included a range of SOM contents from 13-73%. Surface descriptors indicate the management stage and mean SOM content (subscript). Five primary tillage surfaces were sampled, including two from Plantation (PT_{23}, PT_{27}), two from Redmere (PT_{13}, PT_{49}) and one from Rosedene (PT_{70}). Surface PT_{13} was from a rodden, where peat subsidence has exposed the underlying mineral material from a former river channel. This surface was included to provide a mineral soil baseline, to contextualise peat soil measurements. Two secondary tillage surfaces were sampled, including one from Plantation (ST_{28}) and one from Rosedene (ST_{73}). Our selection of surfaces was constrained by availability, which was dependent on the management schedule of the farms. Due to the short window between secondary tillage and planting, we were unable to sample the secondary tillage condition at Redmere Farm. Four early cropped surfaces were sampled, including two from Plantation (EC_{27}, EC_{47}), one from Redmere (EC_{46}) and one from Rosedene (EC_{61}). Finally, one heavy traffic surface was sampled from a harvest working area at Rosedene Farm, to scope the potential influence of on-field vehicle traffic on erodibility and evaluate a potential ‘worst-case’ scenario of erosion vulnerability. Based on the above criteria, twelve surfaces in total were selected for measurement.

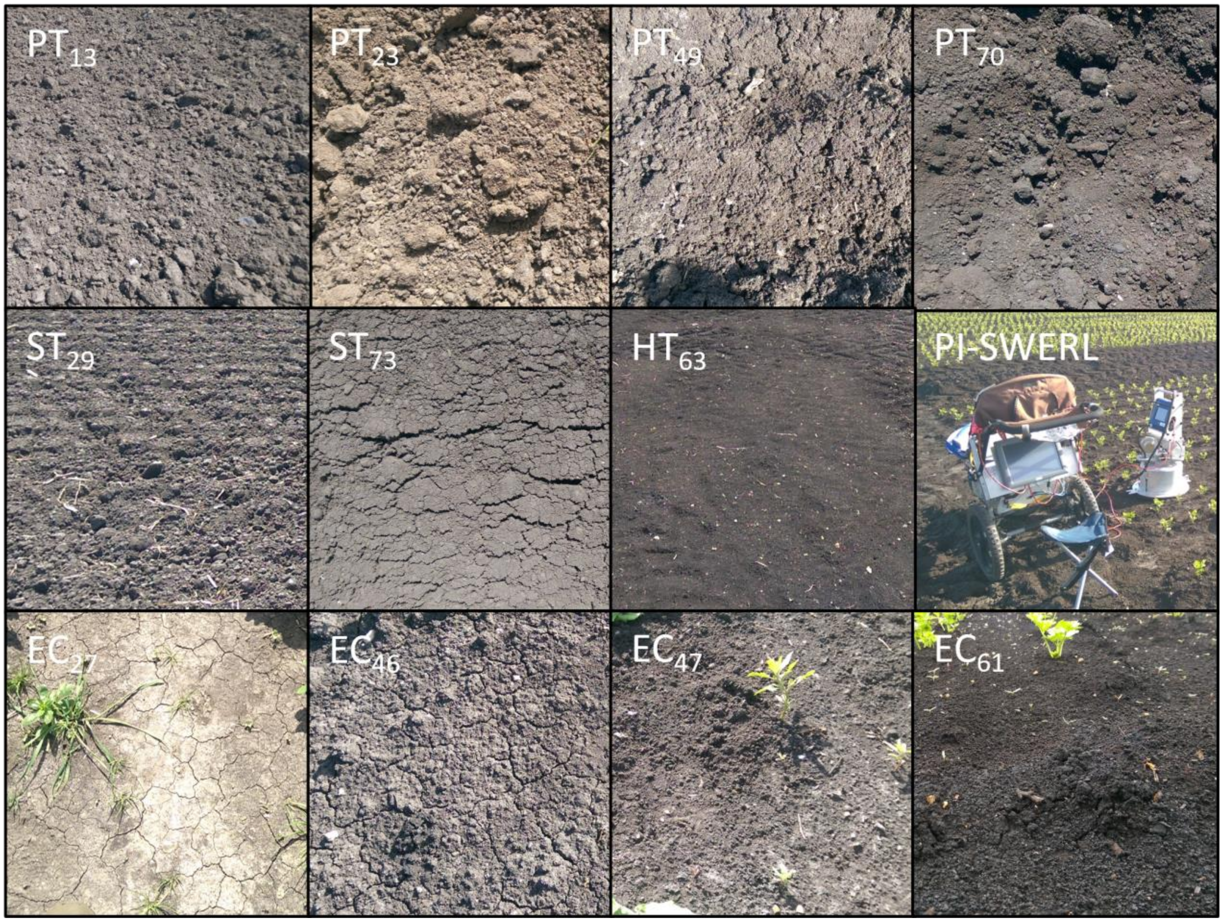

Figure 3.3. Study surfaces. Panels are labelled with surface IDs. No image was available for surface PT_{27}. The PI-SWERL setup is also shown for demonstration purposes. The angle and distance from the surfaces vary between images. Images were selected to give an impression of the surface roughness, aggregation and crusting of study surfaces at the scale of PI-SWERL measurements.

3.3.2. Wind erosion measurements

We used a Miniature Portable *In-Situ* Wind Erosion Laboratory (PI-SWERL; Figure 3.3) to estimate erosion rates under field conditions (Etyemezian et al., 2007). The PI-SWERL has been tested and calibrated across a range of environments and produces dust emission estimates comparable to field wind tunnels (Sweeney et al., 2008; Sweeney et al., 2011; Sweeney and Mason, 2013; von Holdt et al., 2021; Vos et al., 2021; Sweeney et al., 2022). The PI-SWERL's small size and relatively short operation period mean it is well suited to our study objectives, where replicable measurements are required across a range of conditions (e.g., King et al., 2011; Sweeney et al., 2011, 2016; Cui et al., 2019a, 2019b; Vos et al., 2021). The operating principles of the PI-SWERL have been described comprehensively elsewhere (Etyemezian et al., 2007; Sweeney et al., 2008; Sweeney and Mason, 2013). Saltation rate was estimated as a count of saltating particles using optical gate sensors mounted within the chamber. The concentration (D; mg m^{-3}) of particulate matter <10μm in diameter (PM_{10}) in the exhaust flow from the chamber was measured using an attached nephelometer style dust monitor (DustTrak II model 8530). The introduction of clean air into the chamber at a constant rate (F; m^3 s^{-1}) and the known effective area for the PI-SWERL (A_{eff} = 0.026 m^2) allow calculation of a mean PM_{10} emission flux (E; mg m^{-2} s^{-1}; Equation 3.1) over the test duration ($t_{end,i} - t_{begin,i}$) by cumulative summation of fluxes across 1s (t_0) intervals.

Equation 3.1:

$$E_i = \frac{\sum_{begin,i}^{end,i}(D_i \times F \times t_0)}{(t_{end,i} - t_{begin,i}) \cdot A_{eff}}$$

We operated the PI-SWERL using a pre-defined test program including four different phases (Figure 3.4). Phase 1: Stable operating speed of 0 RPM (revolutions per minute) for 90 s. Phase 2: Gradual increase from 0 to 5000 RPM over 120 s. Phase 3: Stable operating speed of 5000 RPM for 120 s. Phase 4: Stable operating speed of 0 RPM for 120 s. The same test program was used on all surfaces. Phase 1 was a flush phase to allow dust concentrations to return to baseline levels before measurements, following any disturbance from chamber placement. Phase 2 allowed identification of u^*_t for both saltating (sand-sized) and PM_{10} particles. We defined u^*_t for saltation as a sustained increase in the 20 s moving average of saltation rate above background noise (7 particles counted per second; pc s^{-1}; Sweeney and Mason, 2013). We applied a similar approach to identifying u^*_t for PM_{10} but used the mean measured dust concentration over the 30 s period preceding each measurement as a baseline. Phase 3 was used to estimate the mean PM_{10} flux and saltation rate. The 5000 RPM value for Phase 3 was chosen because the friction velocity (u^*) of 0.82 m s^{-1}, calculated using Equation 3.2 (Dust-Quant LLC, 2011), was similar to that from the wind tunnel measurements of Zobeck et al. (2013; 0.88 m s^{-1}) on agricultural peat soils. This value was also comparable to the higher end of the wind speed range recorded for Rosedene Farm by Cumming (2018; 9-12 m s^{-1}).

Equation 3.2:

$$u^* = -1.49E-12 \cdot RPM^3 + 9.20E-09 \cdot RPM^2 + 1.42E-04 \cdot RPM + 0.0872$$

Phase 4 was a final flush phase that allowed dust concentrations in the chamber and DustTrak to drop before the end of the measurement run. The PI-SWERL chamber was then carefully brushed clean between runs to minimise dust contamination effects between measurements. The DustTrak is calibrated to Arizona Road Dust standard (ISO 12103-1). Therefore, the PM_{10} mass flux was adjusted using particle density (ρ_P) and C content, to give a PM_{10}-C flux and facilitate comparison of relative soil C loss rates across the wide range of SOM content soils observed at the study sites. All measurements were made in a single four-day, summer period of stable weather to minimise potentially confounding effects of humidity/rainfall on comparisons (Ravi and D'Odorico, 2005; Ravi et al., 2006). Between three and five replicate measurements were made consecutively on each surface (Table 3.1). Replicates were spaced at least 50 m apart but within the same crop management unit to minimise spatial dependence whilst ensuring consistent management within study surfaces. Replicates were reduced at times on surfaces with high erosion potential to avoid passing excessive dust loads through the DustTrak. Where any small weeds were present these were trimmed level with the ground surface.

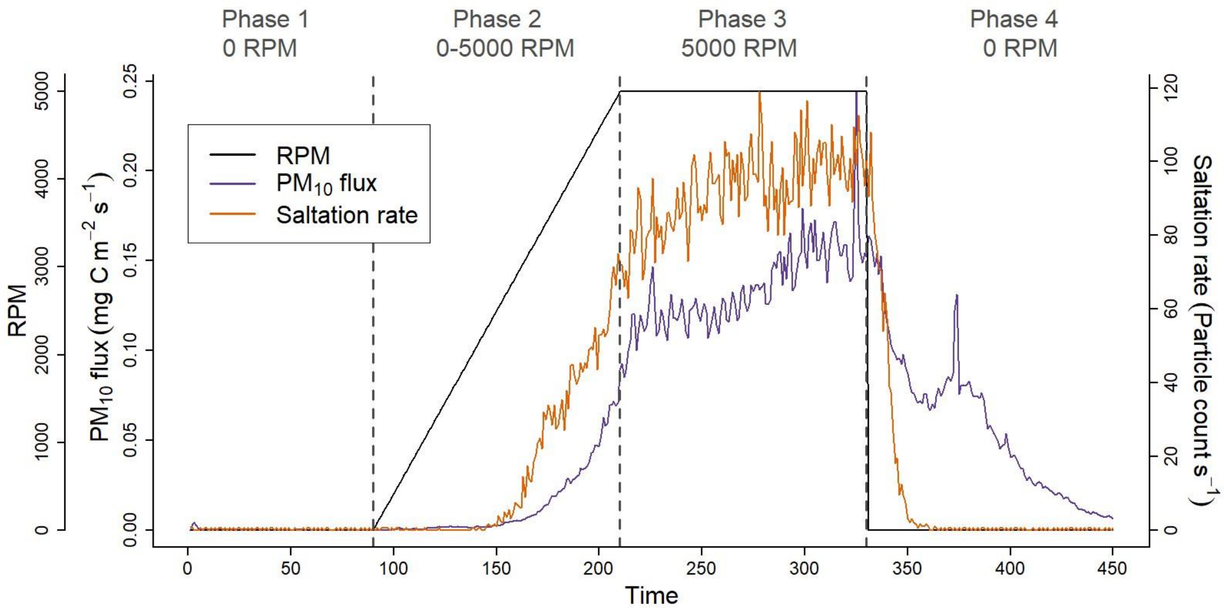

Figure 3.4. Example PI-SWERL measurement. Timeline of a test program from surface EC_{27} with RPM, PM_{10} fluxes and saltation rate shown at 1 s intervals. The test program was broken into several phases indicated by vertical dashed lines and labelled above the figure. These phases were chosen to allow (Phase 1) dust concentrations to return to the baseline before measurements commenced, (Phase 2) estimation of the threshold friction velocity of entrainment, (Phase 3) estimation of mean PM_{10} emissions/saltation rate at a stable friction velocity (0.82 m s^{-1}) and (Phase 4) dust concentrations to drop to facilitate PI-SWERL cleaning between measurements.

3.3.3. Soil properties analysis

Alongside each wind erosion measurement, a 700 ml bulk sample of surface soil (~0-2 cm depth) was collected immediately adjacent to the PI-SWERL footprint. These were stored in rigid plastic containers during transport to minimise disturbance. These samples were air-dried and the ASD was estimated by flat sieving (Mesh diameters = 0.063 mm, 0.21 mm, 0.35 mm, 0.85 mm, 2 mm, 8 mm, 16 mm) using a vibratory device with a fixed vibration intensity (Endecotts Minor 200; 1.6 mm amplitude; 50 Hz) and fitting a Weibull distribution (Zobeck et al., 2003; Figure 3.5). We used two methods to summarise the ASD. Firstly, the MAS was calculated as the diameter at which 50% of particles were undersized. Secondly, we calculated the proportion of WEM, which can be estimated as the percentage of soil mass passing through U.S.A. Standard test sieve No. 20 (<0.85 mm in diameter; Chepil, 1950; Kohake et al., 2010; Zobeck et al., 2013; Webb et al., 2016; Motaghi et al., 2020). The equivalent diameter (D_E; mm) to mineral soil particles of 0.85 mm for each peat sample was calculated using Equation 3.3, where 2.65 g cm^{-3} is the density of quartz sand and ρ is the sample bulk density (g cm^{-3}; Chepil and Woodruff, 1963). This allows density adjustment of the WEM estimate for each sample using the ASD (Campbell et al., 2002).

Equation 3.3:

$$D_E = \left(\frac{2.65}{\rho}\right) 0.85$$

Dry aggregate stability (DAS) was estimated by re-sieving the fraction >0.85 mm (Zobeck et al., 2013). Determination of ρ was undertaken by inserting a metal ring (H = 5 cm, V = 100 cm^3) into the soil adjacent to the PI-SWERL footprint and oven drying the sample (105 °C; 24 h). Further samples were stored at 4°C, before estimation of volumetric water content (VWC) by oven-drying (105°C; 24h) and determination of SOM content by loss-on-ignition (450°C; 16h). We calculated ρ_P (g cm^{-3}) using Equation 3.4 (Rühlmann et al., 2006; where SOM_R is SOM content expressed as kg kg^{-1}) and we estimated C content (g kg^{-1}) using Equation 3.5, where SOM is expressed in g kg^{-1} (Wright et al., 2008). This C content adjustment method uses C content estimates for the entire ASD and is based on the simplifying assumption that C content does not vary significantly by aggregate size. Cumming (2018) found little variation in SOM content by sampling height when using passive slot samplers on a high-SOM peat soil at Rosedene Farm, suggesting a relatively uniform distribution of SOM content by aggregate size in this context.

Equation 3.4:

$$\rho_P = \left[\frac{SOM_R}{1.127 + 0.373 \cdot SOM_R} + \frac{1 + SOM_R}{2.684}\right]^{-1}$$

Equation 3.5:

$$C = 0.516 \cdot SOM - 18.1$$

3.3.4. Statistical analysis

All analyses were performed using R v4.2.2 (R Core Team, 2022). Erosion measurements can exhibit high levels of variation. Therefore, replicate measurements were undertaken on each surface to mitigate the potential influence of outliers. However, this produced a trade-off, reducing the number of surfaces captured (replicates at the level of management cycle stages). Therefore, to account for variation associated with inter-surface differences in our analysis, we either (i) included a random effect in models or (ii) calculated geometric means for each surface and performed analyses using this aggregated data. Pairwise comparisons were performed only on primary tillage and early cropped surfaces to ensure balanced comparison groups. Surface PT_{13} was excluded from the primary tillage category for these purposes as these measurements were from a mineral soil and there was no equivalent mineral soil surface in the early cropped category. Where necessary, dependent variables were log-transformed for linear regression, or non-linear regression was used due to the skewed distributions of the erosion variables.

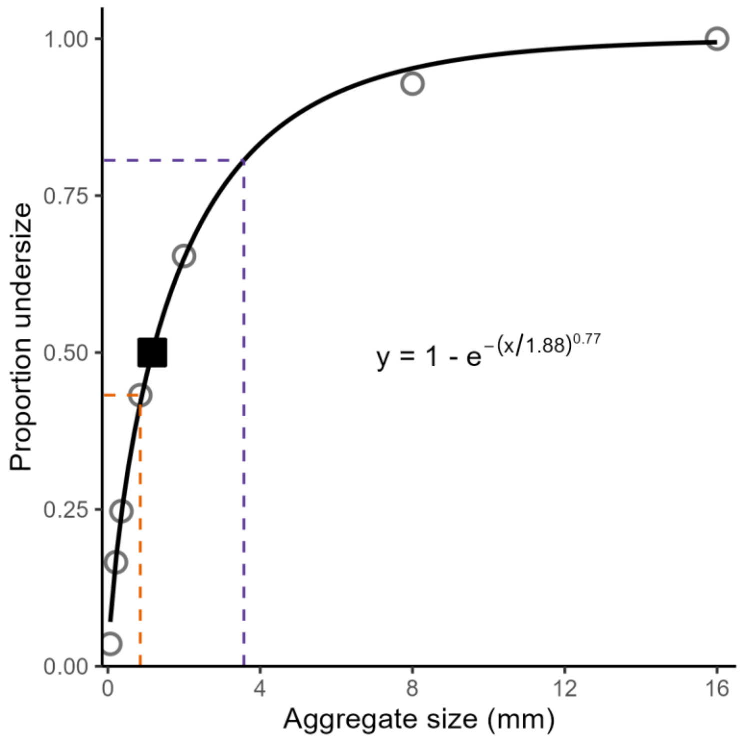

Figure 3.5. Example aggregate size distribution showing summary statistics. Weibull distribution for a replicate from surface EC_{47} (p<0.0001; R_2 = 0.997). Hollow grey circles show measured values. The solid black square shows the mean aggregate size (1.17 mm; proportion undersize = 50%). The orange dashed line indicates the proportion smaller than 0.85 mm diameter (43%). The purple dashed line indicates the proportion smaller than the effective diameter of wind erodible material (3.6 mm), after adjusting for particle density relative to mineral material (wind erodible material = 81%).

3.4. Results

3.4.1. Soil physical properties

Soil physical properties for each surface are summarised in Table 3.1 and correlations between soil physical properties are reported in Table 3.2. The mineral soil from the rodden (PT_{13}) was by far the heaviest textured soil sampled. It exhibited the highest density (both ρ and ρ_P) and MAS, and the lowest moisture content and proportion of WEM for any surface. This clearly highlights the two factors predisposing peat soils to relatively higher erosion vulnerability when compared to mineral soil in our sample: lower particle density and smaller aggregate sizes. MAS was generally higher before planting in the primary/secondary tillage conditions than after planting in the early cropped condition. However, for both tillage categories, the higher-SOM surfaces from Rosedene Farm had a lower MAS than the lower-SOM surfaces in the same category. As we would expect, the density-adjusted proportion of WEM was correlated both with differences in the ASD and SOM content between surfaces. For both high-SOM, tilled surfaces (PT_{70}, ST_{73}), and for all of the early cropped and heavy traffic surfaces, the mean proportion of WEM was >70%, whilst the surface ASDs formed a steep curve with the majority of the soil mass contained within smaller aggregates (Figure 3.6). Contrastingly, for the remaining low-/intermediate-SOM tilled surfaces, the mean proportion of WEM was ≤41% and the ASDs formed flatter curves, with a greater proportion of soil mass contained in larger aggregates (Figure 3.6). This indicates a significant disparity in the potential supply for erosion fluxes between these two groups of surfaces. Soil moisture content was generally higher for tilled surfaces, which may reflect both moisture shielded from evaporative forces within larger aggregates and/or moisture brought to the surface by tillage operations. There was a strong negative correlation between ρ and SOM content. However, the ρ of the surface subject to heavy traffic (HT_{63}) was the highest observed for any management stage at Rosedene Farm, suggesting that vehicle traffic had resulted in compaction of this surface.

Table 3.1. Soil physical properties of study surfaces. Surface ID's reflect management cycle stage and soil organic matter content (%) as subscript. PT = Primary tillage, ST = Secondary tillage, EC = Early cropped, HT = Heavy traffic. Numbers in square brackets next to study IDs indicate the number of replicate measurements made on that surface. Values reported as arithmetic means ± standard errors.

Surface ID [n]	Bulk density (g cm^{-3})	Soil organic matter (%)	Particle density (g cm^{-3})	Mean aggregate size (mm)	Wind erodible material (%)	Dry aggregate stability (%)	Volumetric moisture content (%)
PT_{13} [5]	1.11 ± 0.07	12.6 ± 0.8	1.90 ± 0.07	13.74 ± 2.58	15.9 ± 2.1	94.4 ± 1.3	4.5 ± 0.4
PT_{23} [5]	0.81 ± 0.05	23.2 ± 1.0	1.54 ± 0.06	5.19 ± 0.80	35.7 ± 3.2	96.0 ± 0.6	14.8 ± 0.8
PT_{27} [5]	0.83 ± 0.05	27.7 ± 0.3	1.43 ± 0.02	5.47 ± 0.84	36.8 ± 3.6	90.8 ± 1.3	15.1 ± 2.0
PT_{49} [5]	0.50 ± 0.00	48.5 ± 1.4	1.08 ± 0.04	6.76 ± 0.83	41.1 ± 2.5	93.4 ± 1.3	14.3 ± 0.9
PT_{70} [5]	0.33 ± 0.03	70.5 ± 1.9	0.88 ± 0.03	3.04 ± 0.65	73.2 ± 5.1	91.9 ± 1.3	23.4 ± 2.7
ST_{29} [5]	0.75 ± 0.04	28.5 ± 0.4	1.41 ± 0.02	7.09 ± 0.44	29.9 ± 1.2	98.2 ± 0.3	21.2 ± 2.3
ST_{73} [5]	0.40 ± 0.01	73.2 ± 1.5	0.86 ± 0.02	2.34 ± 0.22	76.6 ± 3.6	91.9 ± 0.3	18.4 ± 1.3
EC_{27} [5]	0.90 ± 0.02	26.6 ± 0.5	1.45 ± 0.02	1.06 ± 0.10	71.9 ± 3.9	81.0 ± 2.5	10.6 ± 0.7
EC_{46} [5]	0.61 ± 0.03	46.7 ± 2.4	1.11 ± 0.07	1.34 ± 0.12	79.2 ± 2.7	89.9 ± 0.9	10.4 ± 1.4
EC_{47} [5]	0.58 ± 0.04	47.2 ± 0.7	1.10 ± 0.02	1.15 ± 0.11	81.4 ± 2.9	89.6 ± 2.3	15.2 ± 2.1
EC_{61} [4]	0.46 ± 0.03	61.4 ± 5.7	0.96 ± 0.12	1.88 ± 0.12	75.5 ± 1.7	94.5 ± 1.4	10.8 ± 1.8
HT_{63} [3]	0.53 ± 0.03	63.1 ± 1.0	0.93 ± 0.01	0.83 ± 0.10	95.1 ± 2.0	91.6 ± 0.7	11.0 ± 1.3

Table 3.2. Correlations between soil property variables. Performed on the full dataset. Spearman's Rho with Holm correction for multiple testing. Significance indicated as: * = p<0.05, ** = p<0.001, *** = p<0.0001.

Variable	Soil organic matter	Mean aggregate size	Wind erodible material	Dry aggregate stability	Volumetric moisture content
Bulk Density (g cm^{-3})	-0.93 ***	0.27	-0.58 ***	0.07	-0.37
Soil organic matter (%)		-0.39 *	0.69 ***	-0.12	0.43 *
Mean aggregate size (mm)			-0.89 ***	0.50 ***	0.10
Wind erodible material (%)				-0.39 *	0.04
Dry aggregate stability (%)					0.28

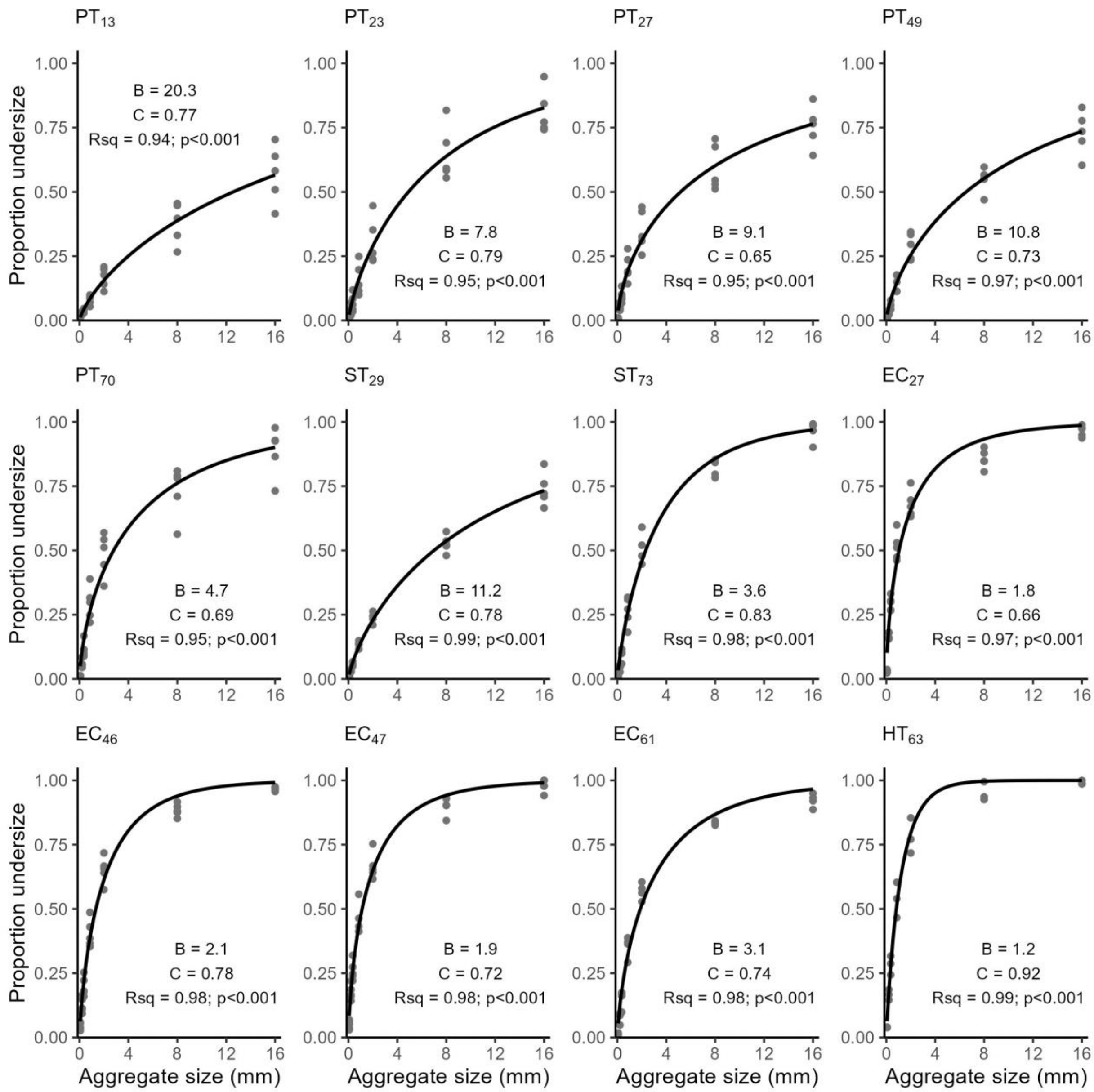

Figure 3.6. Mean aggregate size distributions for study surfaces. Surface ID's reflect management cycle stage and soil organic matter content (%) as subscript. PT = Primary tillage, ST = Secondary tillage, EC = Early cropped, HT = Heavy traffic. Points show measured values. Lines show fitted Weibull distributions of the form y = 1 – exp(-((x/B)^C)). Parameter values for B and C are shown on each panel along with the model R^2 (Rsq) and p value.

3.4.2. Threshold friction velocities of entrainment

The mean u^*_t for PM_{10} emissions observed for all surfaces in this study was 0.39 ± 0.02 m s^{-1}. Management had a significant overall effect on u^*_t for PM_{10} particles across the study surfaces ($p = 0.0006$; Figure 3.7a). The mean u^*_t of PM_{10} particles was significantly lower for early cropped surfaces than primary tillage surfaces ($p = 0.029$; Figure 3.7b) and the highest mean u^*_t estimates for PM_{10} emissions in our sample were those from surfaces subject to primary tillage. However, whilst the sample range was fairly wide (0.22-0.72 m s^{-1}), a substantial portion of this is attributable to within-surface variation, and gross differences between management category means were substantially smaller (Figure 3.7b). In comparison, it is notable that the u^*_t for saltation exceeded the parameters of our test program on several surfaces (Figure 3.7c). This produced a clear divide between surfaces on which saltation occurred and those on which it was not observed at $u^* \leq 0.82$ m s^{-1}. These groups align with the surfaces previously highlighted as displaying substantial differences in the proportion of WEM. We estimated a mean u^*_t of 0.34 ± 0.02 m s^{-1} for PM_{10} emissions and a u^*_t of 0.41 ± 0.01 m s^{-1} for saltation from surfaces where saltation was observed.

Our u^*_t estimates are of a similar order of magnitude to the mean of the u^*_t estimates of Kohake et al. (2010; 0.56 ± 0.09 m s^{-1}), who conducted a laboratory wind tunnel test on prepared trays of agricultural peat soil. The higher estimate of Kohake et al. (2010) is driven by one site with 61% SOM content, 11% WEM (not adjusted for density) and MAS of 8.82 mm, which is not comparable to any of the surfaces in our sample (Table 3.1). It is possible that either the tray preparation process resulted in a surface dissimilar to those observed under field conditions or that this simply reflects a limitation in the range of our sample. Excluding this site, the mean of the u^*_t estimates from Kohake et al. (2010) is 0.40 ± 0.03 m s^{-1}, which is very similar to our mean estimate for comparable surfaces. The u^*_t of saltation was higher than the u^*_t of PM_{10} emissions for 82% of our measurements, though it should be noted that in 56% of measurements the u^*_t for saltation was not reached. For measurements where saltation was recorded, the mean difference between u^*_t for saltation and PM_{10} dust was 0.06 ± 0.01 m s^{-1}. This was significantly greater than zero ($t = 4.6$, $p<0.0001$) but negligible in practical terms, given the precision of u^*_t estimation.

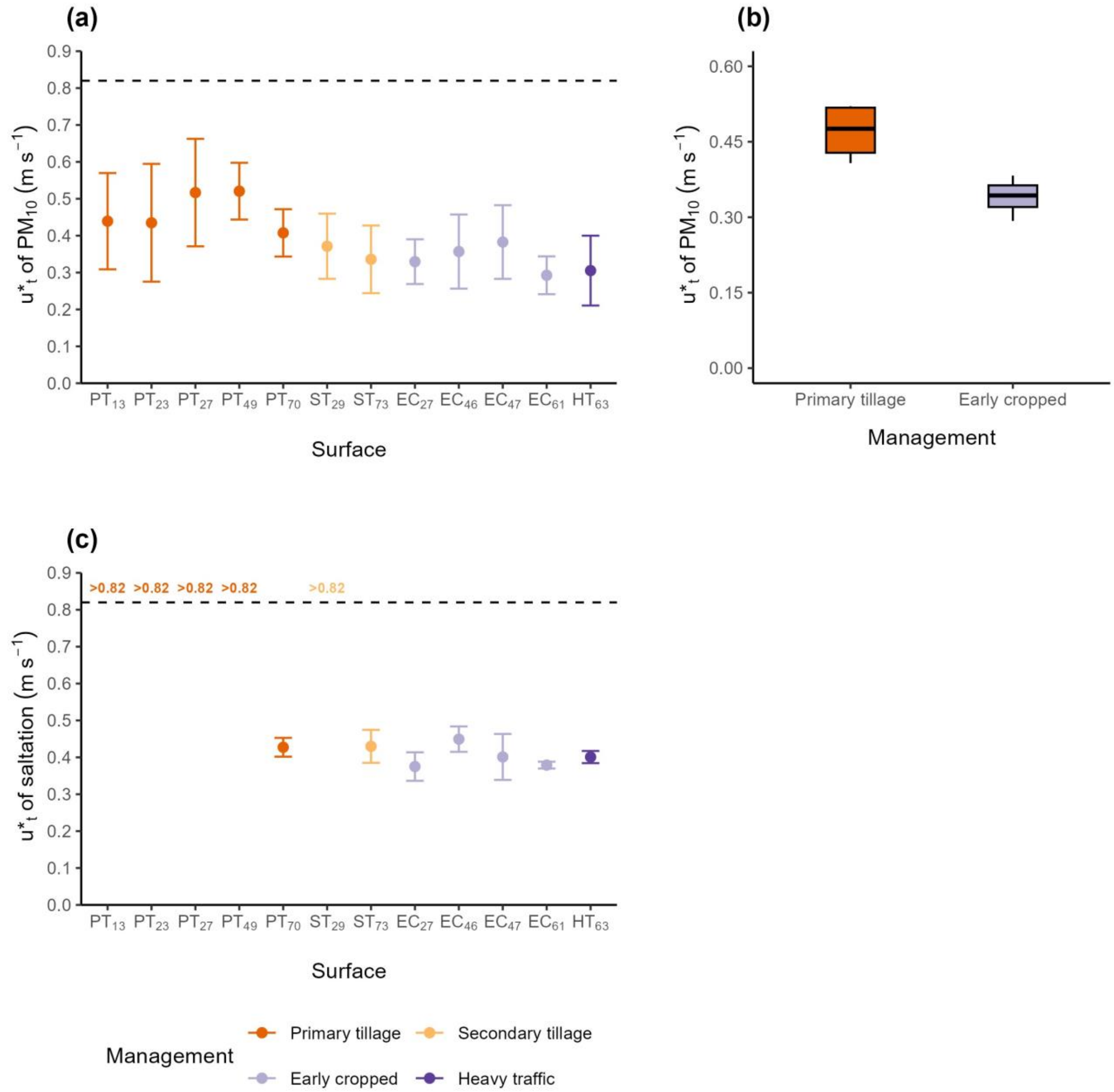

Figure 3.7. Threshold friction velocity of entrainment (u^*_t) of study surfaces. (a) u^*_t for PM_{10}. (b) Comparison of u^*_t for PM_{10} between primary tillage (excluding PT_{13}) and early cropped surfaces. (c) u^*_t for saltation. Data in (a) and (c) presented as arithmetic means (points) and standard deviations (error bars). Uppercase letters in surface identification labels align with management, whilst subscript numbers indicate soil organic matter content (%). The horizontal dotted line indicates the maximum friction velocity applied during the test (0.82 m s^{-1}). In (c) surfaces for which saltation was not observed at $u^* \leq 0.82$ m s^{-1} are indicated above the maximum friction velocity line. Data in (b) presented as a boxplot of surface arithmetic means.

3.4.3. Erosion rates

The mean of the PM_{10}-C fluxes observed for saltating surfaces in this study was 0.066 ± 0.017 mg C m^{-2} s^{-1}. This is a similar order of magnitude to the mean of the PM_{10} fluxes from Zobeck et al. (2013) when adjusted using mean site C content, for agricultural peat soils; 0.029 ± 0.011 mg C m^{-2} s^{-1}. However, the values obtained in this study are approximately double those of Zobeck et al. (2013) and this can likely be attributed to differences in management/surface preparation. Zobeck et al. (2013) used specifically prepared test surfaces, which were rolled and measured shortly after preparation. In contrast, we sampled surfaces under a range of field conditions to address the absence of such measurements in previously reported studies. Our higher PM_{10} flux estimates could be due to (i) weathering processes – both natural (e.g. wind/rainfall) and anthropogenic (e.g. irrigation) – further increasing the erodibility of our early cropped and heavy traffic surfaces in the period since tillage and (ii) differences in moisture content and humidity (Wiggs et al., 2004; Ravi and D'Odorico, 2005; Ravi et al., 2006).

The highest PM_{10}-C fluxes were observed on surface HT_{63} (Figure 3.8a). In both the tillage categories, the high-SOM content surfaces from Rosedene Farm exhibited higher PM_{10}-C fluxes than the other lower-SOM content surfaces. The overall trend was for surfaces to exhibit higher PM_{10}-C fluxes (greater erosion vulnerability) with progression through the crop management cycle. Management had a significant effect on mean PM_{10}-C fluxes ($p = 0.015$). Mean PM_{10}-C fluxes were significantly higher from early cropped surfaces than primary tillage surfaces ($p = 0.003$; Figure 3.8b). Mean saltation rate was not significantly affected by management overall ($p = 0.051$; Figure 3.8c). However, mean saltation rate was significantly higher on early cropped surfaces than primary tillage surfaces ($p = 0.002$; Figure 3.8d). Mean saltation rates were fairly consistent across surfaces which exhibited saltation (Figure 3.7c, Figure 3.8c), especially if saltation rates are standardised using ρ_P. This is in line with the observations of Zobeck et al. (2013) for surfaces with similar SOM contents to our sample (28-67%).

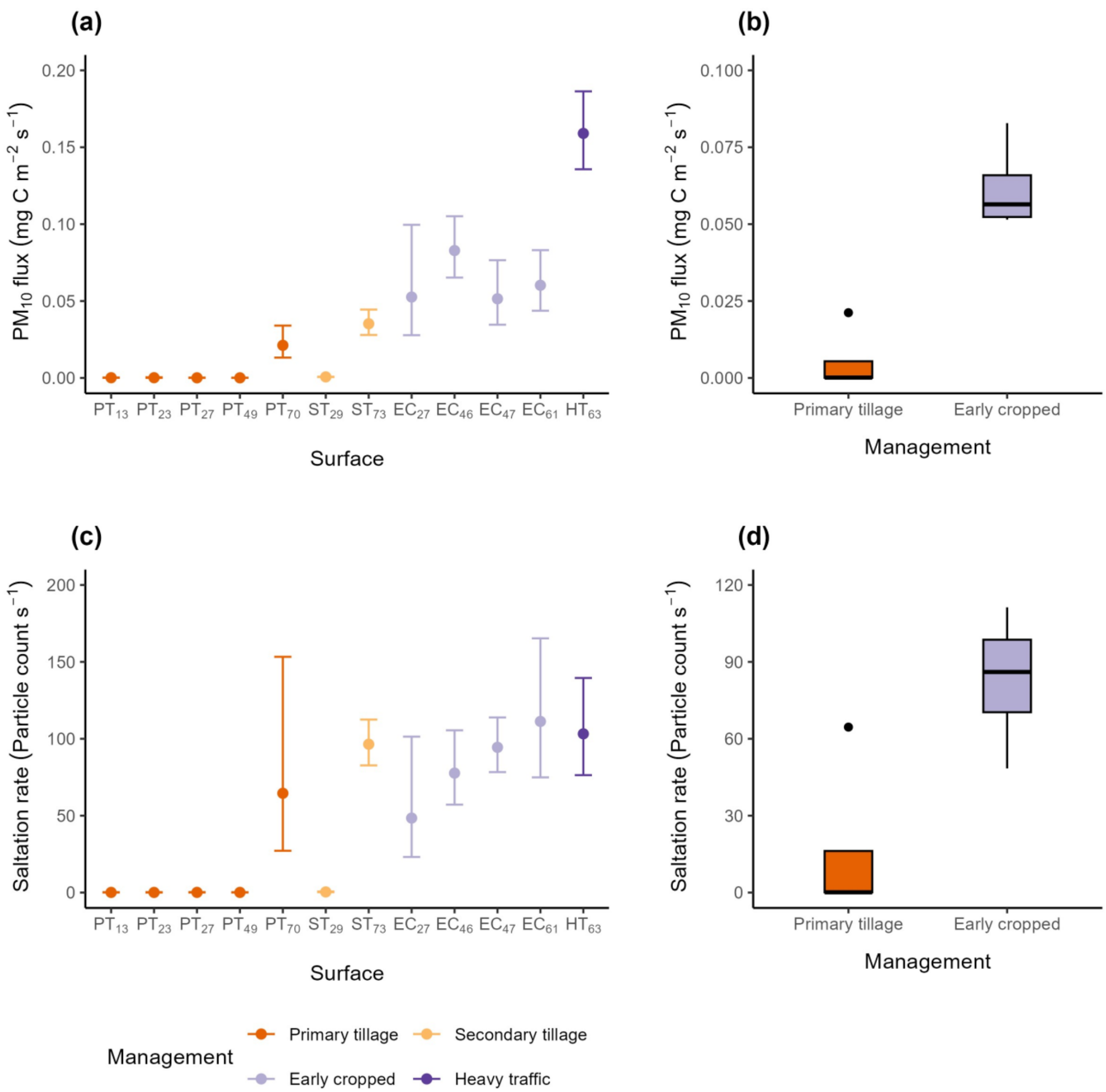

Figure 3.8. Effects of management on the vulnerability of study surfaces to erosion. (a) PM$_{10}$ carbon flux by surface. (b) Comparison of mean PM$_{10}$ carbon flux between primary tillage (excluding PT$_{13}$) and early cropped surfaces. (c) Mean saltation rate by surface. (d) Comparison of mean saltation rate between primary tillage (excluding PT$_{13}$) and early cropped surfaces. Data in (a) and (c) presented as geometric means and geometric standard deviations. Uppercase letters in surface identification labels align with management categories, whilst subscript numbers indicate soil organic matter content (%). Data in (b) and (d) presented as boxplots of surface geometric means.

PM_{10}-C fluxes were significantly higher from surfaces on which saltation occurred than from those on which saltation was not observed ($p<0.0001$; Figure 3.9a). Even though the u^*_t of PM_{10} emissions was exceeded on the surfaces where the u^*_t of saltation was not, PM_{10}-C fluxes from these surfaces were still negligible (Figure 3.7a; Figure 3.8a). The low PM_{10} emissions from these surfaces likely reflect the disturbance of a small reservoir of loosely held dust on the outer surfaces of larger non-erodible aggregates. Across our whole sample, the geometric means of PM_{10}-C flux and saltation rate showed a strong positive correlation (Spearman's $\rho = 0.83$, $p = 0.001$). However, there was no significant correlation within the subsample of surfaces from which saltation was observed ($r = 0.40$, $p = 0.38$).

Surfaces on which saltation occurred had significantly lower MAS ($p = 0.0001$; Figure 3.9b), higher SOM content ($p = 0.01$; Figure 3.9c) and consequently, higher proportions of WEM ($p<0.0001$; Figure 3.9d). Overall, a positive exponential relationship was observed between PM_{10}-C flux and WEM ($p<0.0001$; $R^2 = 0.91$; Figure 3.9e). When the difference between the MAS and the D_E was calculated (Figure 3.10), we observed that this value was less than zero for surfaces which exhibited saltation, and greater than zero for surfaces which did not. Essentially, in our sample, saltation and substantial PM_{10}-C fluxes only occurred if >50% of a soil's mass was contained in aggregates with diameter < D_E (This is equivalent to WEM >50%). Contrastingly, if >50% of a soil's mass were contained in aggregates with diameter > D_E (WEM < 50%), this was strongly protective against erosion losses. When examining only the surfaces on which saltation was observed, the exponential relationship between PM_{10}-C flux and WEM was further mediated by the moisture content of surface soil (~0-2 cm depth; $p = 0.004$; Marginal $R^2 = 0.70$; Figure 3.11). This suggests that WEM has a stronger effect on PM_{10} fluxes in drier soils, whilst moisture content has a stronger effect in soils with a greater proportion of WEM.

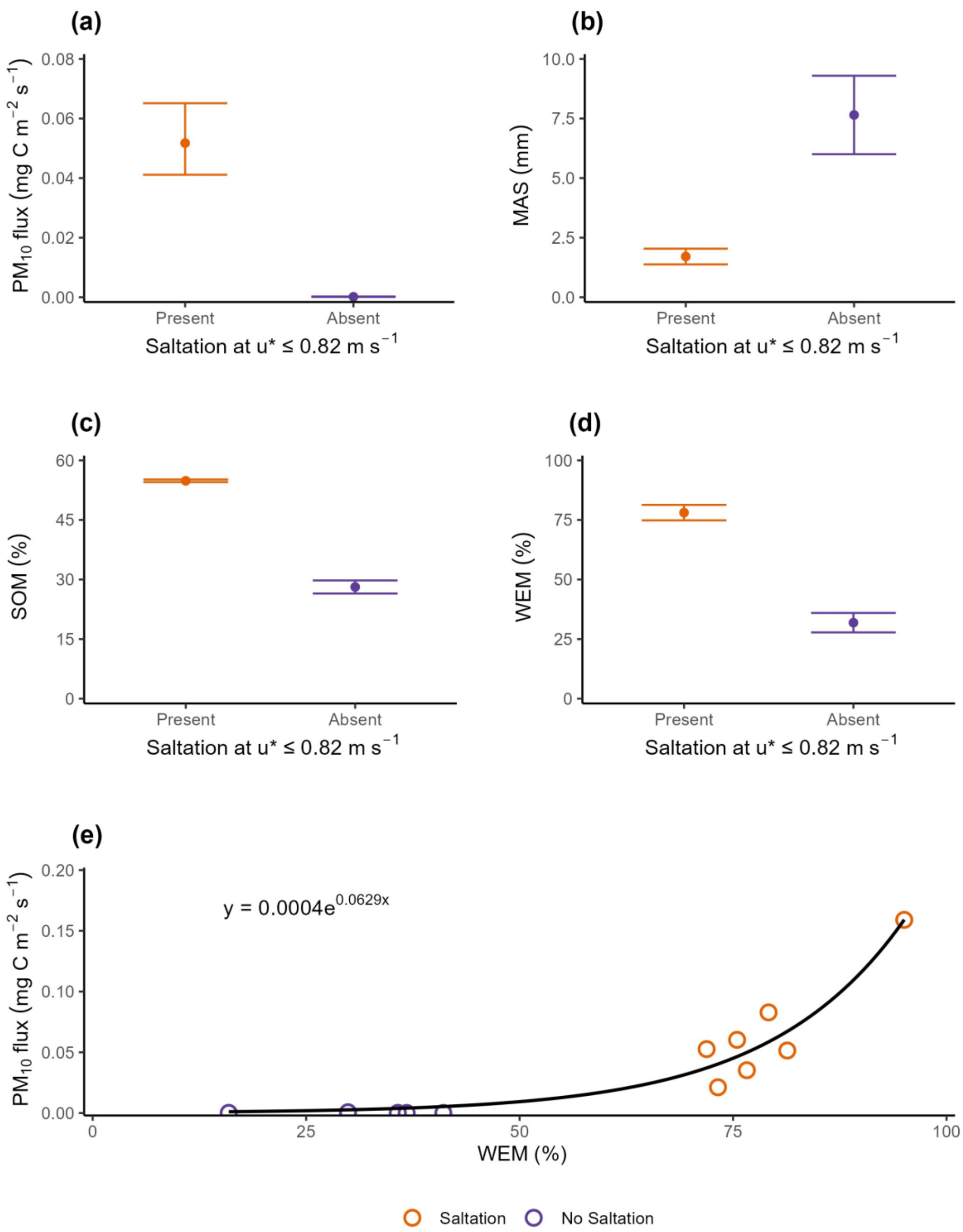

Figure 3.9. Relationship between erodibility and soil physical properties. Comparisons of (a) PM_{10} carbon fluxes, (b) mean aggregate size, (c) soil organic matter content and (d) wind erodible material between surfaces where saltation was present/absent. Values presented as geometric means and 95% confidence intervals in (a) and arithmetic mean and 95% confidence intervals in (b), (c) and (d). (e) Shows the exponential relationship between surface geometric mean PM_{10} carbon fluxes and surface mean WEM ($p<0.0001$; $R^2 = 0.91$).

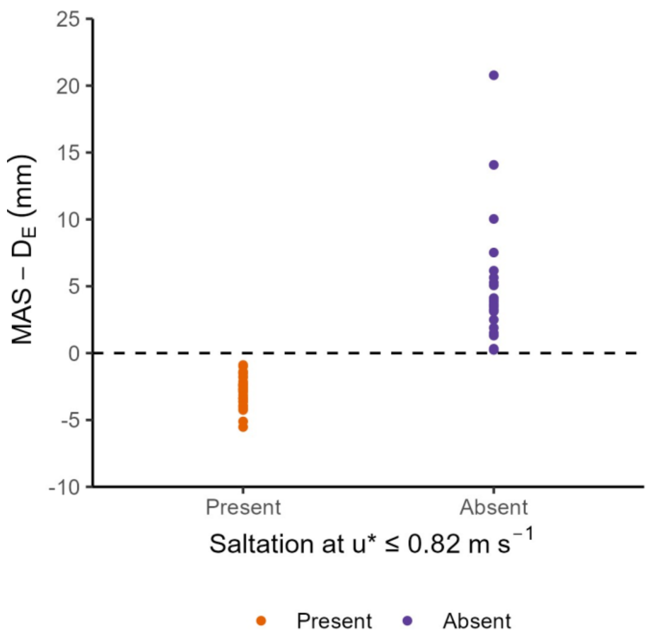

Figure 3.10. Comparison of the difference between mean aggregate size (MAS) and effective diameter of erodible material (D_E) between surfaces for which saltation was present/absent at $u^* \leq 0.82$ m s^{-1}. This value was always less than zero for surfaces which exhibited saltation and greater than zero for surfaces which did not. MAS is the diameter for which 50% of soil mass is undersized. D_E is the diameter equivalent to mineral material with a diameter of 0.85 mm after adjusting for sample density. Therefore, diameters less than D_E indicate erodible material and a y-axis value of zero is equivalent to a density-adjusted wind erodible material value of 50%. Points indicate individual measurements. The dashed horizontal line indicates a value of zero.

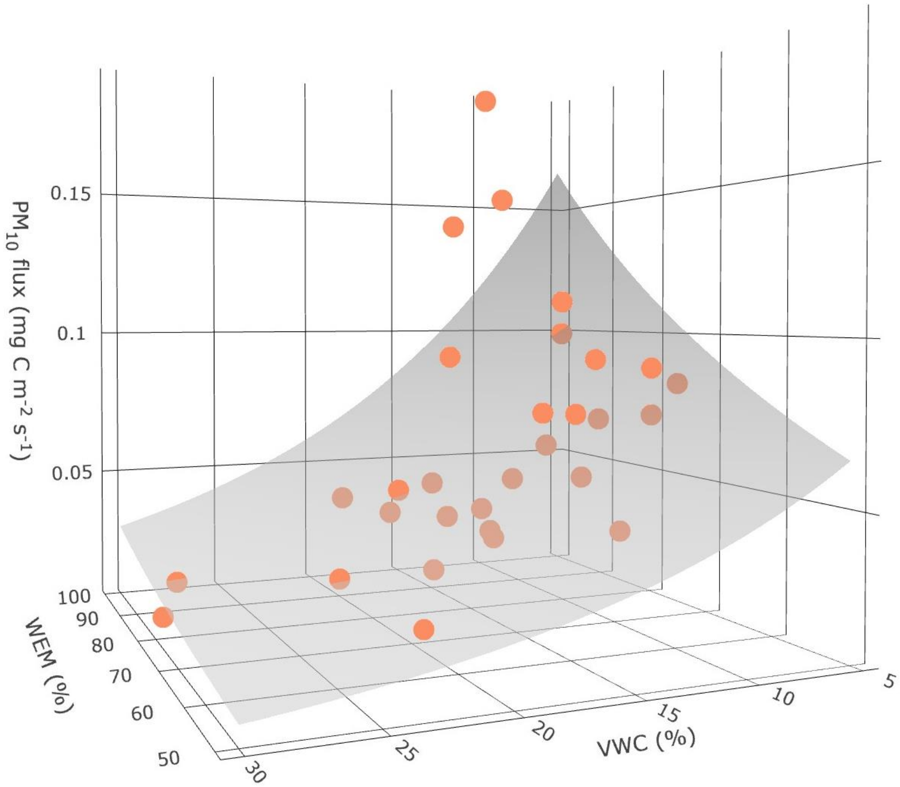

Figure 3.11. Relationship between PM_{10} flux, density-adjusted proportion of wind erodible material (WEM) and volumetric moisture content (VWC). This analysis only includes measurements from surfaces that exhibited saltation at a $u^{*} \leq 0.82$ m s^{-1}. The plane shows the mean model predictions ($p = 0.004$; Marginal $R^2 = 0.70$).

3.4.4. Principal component analysis

Principal component analysis (PCA) results also highlighted the role of the proportion of WEM adjusted for sample density as a major driver of whether saltation would or would not be observed. In the first PCA (all surfaces; Figure 3.12a), principal component (PC) 1 was negatively related to SOM and WEM, whilst being positively related to ρ_P and MAS, and explained 60.1% of the variance in the independent variables. PC2 explained a further 22.4% of the variance and was positively related to VWC and DAS. In total, PC1 and PC2 accounted for 82.5% of the variance in the independent variables. When only surfaces that exhibited saltation were considered, it was notable that early cropped surfaces were on average, more compacted, with smaller, drier, less stable aggregates than surfaces from the two tillage conditions. The heavy traffic surface HT_{63} was uniquely characterised by having the highest proportion of WEM as well as the highest PM_{10}-C fluxes (Figure 3.8a). In the second PCA (surfaces PT_{70}, ST_{73}, EC_{27}, EC_{46}, EC_{47}, EC_{61}, HT_{63}; Figure 3.12b), PC1 was negatively related to SOM, MAS, DAS and VWC, whilst being positively related to ρ and ρ_P. and explained 59.3% of the variance in the independent variables. PC2 explained a further 24.2% of the variance, was strongly positively related to WEM and negatively related to MAS. In total, PC1 and PC2 accounted for 83.4% of the variance in the independent variables.

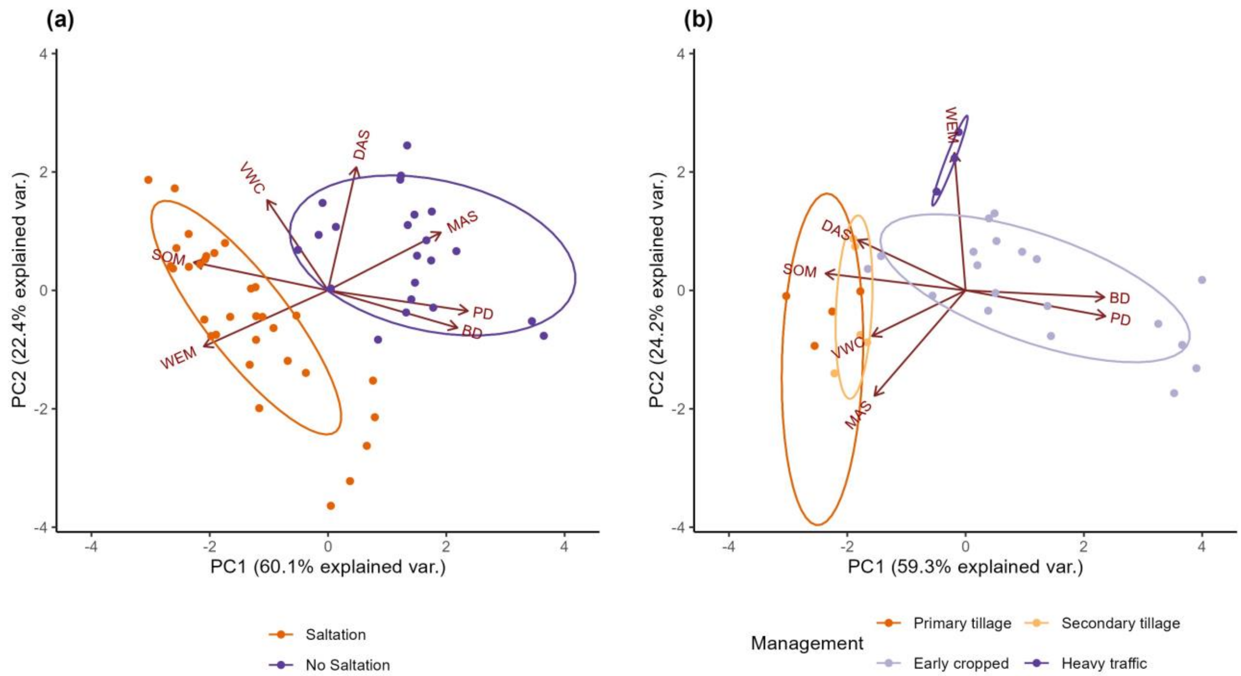

Figure 3.12. Principal component analysis biplots of independent variables. (a) All data grouped by whether saltation was or was not observed at friction velocity ≤ 0.82 ms^{-1}. (b) Data from surfaces which exhibited saltation at friction velocity ≤ 0.82 ms^{-1}, grouped by management category. DAS = Dry aggregate stability, MAS = Mean aggregate size, PD = Particle density, SOM = Soil organic matter content, VWC = Volumetric water content, WEM = Proportion of wind erodible material adjusted for sample density.

3.5. Discussion

In this study, the density-adjusted proportion of WEM was the single best predictor of the erosion vulnerability of bare agricultural peat soil surfaces. Density-adjusted WEM was predictive both of whether saltation would occur on a surface at $u^* \leq 0.82$ m s^{-1} (Figure 3.9d; Figure 3.10) and of mean PM_{10}-C fluxes at a u^* of 0.82 m s $^{-1}$ (Figure 3.9e; Figure 3.11). Density-adjusted WEM has previously been shown to be an excellent predictor of the available erodible material for peat soils managed for extraction in wind tunnel tests by Campbell et al. (2002). A limited further analysis presented in Appendix 3.1 suggests that the observations of Zobeck et al. (2013) would broadly agree with our conclusions on a C-adjusted basis using an approximation of the density-adjusted proportion of WEM as a predictor. Given the wide range of soil conditions captured by our sample and the apparent alignment of our results with previous findings, there is reasonable evidence that the density-adjusted proportion of WEM is an important indicator of vulnerability to wind erosion for agricultural peat soils, as it is in other soils. It may therefore also represent a useful practical indicator for land managers. Estimation of the full ASD is labour intensive but our analysis (Appendix 3.1) suggests a reasonably robust approximation could be produced from low cost, easily obtainable measurements. In our sample, WEM could be predicted quite robustly using a combination of (i) the proportion of sample mass passing through a 0.85 mm sieve and (ii) the SOM content of the sample. Values >50% would then indicate soils which may be vulnerable to erosion if left bare (Figure 3.10), allowing improved spatiotemporal targeting of mitigation strategies. Production of a more robust reference database covering a wider range of scenarios would be the next step necessary in developing this as a practical solution for assessment of erosion vulnerability of bare soil on lowland agricultural peatlands.

Management was found to influence the erosion vulnerability of agricultural peat soils primarily by modifying soil physical properties. The general pattern observed in this study was of increasing vulnerability to erosion with progress through the farm management cycle (Figure 3.8), which was paralleled by an increasing proportion of WEM (Table 3.1; Figure 3.9e). In our sample, the mechanical action of both primary and secondary tillage produced notably higher erosion vulnerability on high-SOM peat soils than lower-/intermediate-SOM surfaces (Figure 3.7c; Figure 3.8). Tillage can alter soil MAS and the ASD through mechanical abrasion of soil aggregates (Lyles and Woodruff, 1962; Zobeck and Popham, 1990) and in the tillage conditions, high-SOM peat soils displayed both the lowest MAS and highest proportion of WEM (Table 3.1; Figure 3.6).

From our observations, it appears that in peat soils, SOM content plays a role which is equivalent to that of soil texture in mediating the effects of tillage on the ASD of mineral soils (Fryrear, 1995). Whilst erosion rates observed on tilled surfaces were generally low (Figure 3.8), high-SOM peat soils appear prone to forming small aggregates following tillage and are therefore potentially vulnerable to erosion unless mitigation strategies are employed. The WEM values <50% for low-/intermediate-SOM tilled surfaces (Table 3.1) suggest that sufficient non-erodible surface aggregates were present to shield erodible material and prevent erosion of these surfaces at u^* $\leq$0.82 m s^{-1}. At a larger scale (beyond the scope of PI-SWERL measurements), primary tillage may also result in the introduction of furrows to the field surface. This may further increase surface roughness and offer additional protection if oriented appropriately in relation to the prevailing wind direction (Fryrear, 1995; Wiggs and Holmes, 2011). Therefore, in practice, erosion rates from primary tillage surfaces may be slightly lower than the potential erosion rates identified by our measurements. Overall, our evidence implies that tillage is likely to result in non-erodible bare soil surfaces for lower-SOM content peat soils, whereas high-SOM peat soils may be vulnerable to erosion following tillage.

We observed very high PM_{10}-C fluxes on the post-harvest working area surface, which had been subjected to heavy vehicle traffic (HT_{63}; Figure 3.8a). This was associated with an extremely high density-adjusted proportion of WEM (Table 3.1; Figure 3.9e), even when compared to other erodible surfaces (Figure 3.12b). When compared to surfaces of similar SOM content, the ρ of this surface was also high (Table 3.1). This strongly suggests that the mechanical action from repeated vehicle passes disintegrates soil aggregates, resulting in a relatively smooth, compacted soil surface with a very large supply of erodible material and very few larger non-erodible aggregates to provide protection from erosive forces. We only examined one high-SOM surface that had experienced heavy traffic. Therefore, whilst our data suggest that surfaces subject to heavy traffic may represent major PM_{10} emission hotspots (especially under dry conditions; Figure 3.11), we must use some caution in extrapolating this conclusion to lower-SOM surfaces. However, Zobeck et al. (2013) measured their highest PM_{10} fluxes and a MAS of 0.20 mm (Compared to a MAS for HT_{63} of 0.83 mm) for a low-SOM (16.7%) peat soil surface that had been flattened by turf rolling, which suggests that lower-SOM peat soils could also be vulnerable to high intensity mechanical action. Given the generally high erosion rates from unpaved roads (Cui et al., 2019a) and the active role vehicles themselves play in producing erosive forces (Pinnick et al., 1985; Goosens et al., 2001; Kuhns et al., 2010; Destain et al., 2016), a precautionary approach dictates that we highlight areas subjected to heavy vehicle traffic (e.g. harvest working areas, farm tracks, tramlines and field headlands) as potentially important erosion hotspots in agricultural peatlands systems. Nonetheless, given the limited nature of our sampling with regard to this issue, firm conclusions would require further observations examining the erodibility of high traffic farm areas and these represent a research priority.

Whilst management practices are relatively consistent across the study farms, this was as an observational study. Crop establishment practices (including irrigation rate/frequency) and time since planting were not controlled between surfaces (the same is true for tillage intensity). The observational nature of the study design does not prevent us achieving our goal of identifying potential management influences on erodibility and spatiotemporal peaks of erosion vulnerability. However, controlled experimental manipulations would be required to elucidate and quantify any causal relationships between specific management practices and erosion vulnerability. As a result of these design limitations, we cannot say with certainty which factors resulted in high erosion vulnerability for early cropped surfaces in our sample. However, irrigation (~15 mm) is often undertaken shortly after planting to support the establishment of juvenile plants and minimise mortality rates. Due to warm, dry conditions during the measurement period, all early cropped surfaces in our sample had been irrigated in the period since planting and prior to measurement and the influence of irrigation on study surfaces was visible as surface crusting (see Figure 3.3). Natural weathering processes may also have contributed to differences between secondary tillage and early cropped surfaces. However, irrigation was the main anthropogenic (management) difference between these groups in our sample and there did appear to be some association between irrigation and increased erosion vulnerability.

In contrast to tilled surfaces, where low-/intermediate-SOM soils displayed lower erosion rates, erosion rates were quite consistent across all surfaces following planting (early cropped condition), regardless of SOM content (Figure 3.8a; Figure 3.8c). Our results indicate that crop establishment practices (including irrigation) may substantially increase the erosion vulnerability of low-/intermediate-SOM peat soils by modifying the ASD, lowering MAS and increasing the proportion of WEM (Table 3.1; Figure 3.6; Figure 3.8). Crop establishment practices appeared to have a smaller effect on the ASD and erodibility of high-SOM peat soil (Table 3.1; Figure 3.6; Figure 3.8) but did appear to slightly lower the u^*_t, which may reflect a decrease in surface roughness due to crust formation following irrigation (Figure 3.2g; Figure 3.3; Kohake et al., 2010). The difference in the effect size from crop establishment practices between high-SOM and lower-SOM peat soils may reflect that following tillage the MAS of high-SOM peat soils was already very low leaving less scope for further disaggregation (Table 3.1). It is also possible that smaller, higher SOM content soil aggregates have greater wet aggregate stability due to the hydrophobicity and binding action of organic matter (Le Bissonnais, 1996). The cumulative kinetic energy of water droplets applied to a surface is the main driver of aggregate disintegration in response to rainfall/irrigation (Darboux et al., 2016). As seen in the comparison of the conclusions of Kohake et al. (2010; higher cumulative kinetic energy) and Campbell et al. (2002; lower cumulative kinetic energy), this can have substantial effects on the consequent erodibility of peat soils. Our observations suggest that the hydraulic action of irrigation following planting may be an important factor increasing the vulnerability of agricultural peat soils to wind erosion. However, controlled experiments would be required to confirm the role of irrigation and rule out other crop establishment/natural weathering effects. The results of Kohake et al. (2010) also suggest that heavy rainfall may produce similar effects.

Our observations highlight three major spatiotemporal peaks in the erosion vulnerability of bare peat soil during the management cycle of salad crop production. The first follows tillage and is specific to high SOM content peat soils. The second follows planting, is not dependent on SOM content, and lasts until crop growth produces sufficient vegetative cover to shield the surface from erosive winds. In the absence of mitigation efforts, the combination of these two peaks produces a prolonged period of erosion vulnerability on high SOM peat soils, extending from primary tillage through to crop maturation. Thirdly, areas of bare soil subject to heavy vehicle traffic likely represent spatial hotspots. Some of these (e.g., farm tracks/headlands) will represent persistent hotspots of vulnerability as they remain bare throughout the year. Others will be more short-lived and episodic vulnerability hotspots (e.g., bare areas subject to heavy on-field vehicle activity during harvest) but may represent larger areas for their duration.

As vegetation providing ~40% ground cover is sufficient to inhibit wind erosion (Funk and Engel, 2015), the primary option for erosion mitigation from bare soil should be minimisation of the area and time for which soil is left bare and exposed to erosive forces (Hagen and Armbrust, 1994; Touré et al., 2019). This is especially the case in spring when regional wind speeds are highest (Newman, 2022). Vegetation can shield particles from wind, increase surface roughness and trap entrained sediment (inducing rapid redeposition and minimising sediment losses from the system; Mayaud and Webb, 2017). There are numerous mitigation options with the potential to reduce erosion rates from lowland agricultural peat soils. Options to increase vegetative cover include cover cropping and reduced tillage (Hoepting et al., 2008), companion cropping of interplanted cereals between vegetable crop rows (Schultz and Carlton, 1959, Schultz et al., 1963), retention of crop residues on field surfaces following harvest (Mendez and Buschiazzo, 2010; Pi et al., 2020; Lin et al., 2021) and the installation of shelterbelts/hedgerows between fields (Leenders et al., 2016; Řeháček et al., 2017; Chang et al., 2021). Where bare soil is unavoidable, artificial cover from fleecing, chemical soil stabilisers (e.g., Kavouras et al., 2009) and mulches (Cong et al., 2016; Robichaud et al., 2017) may also provide options to reduce erosion vulnerability. Chemical soil stabilisers may be particularly valuable for erosion control on areas subject to heavy vehicle traffic where vegetation cover is not possible (e.g., farm tracks/headlands). However, chemical soil stabilisers have generally been considered unsuitable for peat soils (Riksen et al., 2003) and further research would be required to identify/develop suitable options. Given the potential influence of irrigation on surface erodibility, methods with lower cumulative kinetic energy (e.g. drip irrigation) should be investigated to see whether they could reduce erosion vulnerability during the early cropped period. However, given the potential for heavy rainfall to induce similar effects, it is unlikely this would be sufficient as an isolated erosion mitigation measure. Due to the high GHG emissions and the economic importance of agricultural peat soils systems (Freeman et al., 2022) it is imperative that indirect effects of wind erosion mitigation strategies be considered before they are recommended to land-managers, so that cost/benefit trade-offs between erosion mitigation benefits and any GHG emission/economic consequences can be fully assessed.

3.6. Conclusions

Our data indicate that the proportion of WEM, once adjusted for sample density, is the single best predictor of the erosion vulnerability of lowland agricultural peat soils. Surfaces with WEM values >50% appear vulnerable to wind erosion. PM_{10}-C fluxes from erodible surfaces were then further mediated by moisture content, with drier soils displaying higher PM_{10}-C fluxes. Tillage, crop establishment practices and vehicle traffic can strongly influence the erosion vulnerability of agriculturally managed peat soil surfaces. The effects of tillage appear to be strongly mediated by SOM content, with tilled high SOM soils demonstrating high erosion vulnerability, whilst tilled lower SOM content soils were not erodible. Crop establishment induced changes in soil physical properties resulted in similar, high erosion rates across all SOM contents in the early cropped period, where soil is especially vulnerable due to low vegetation cover and surface roughness. Consequently, high SOM content peat soils are vulnerable to erosion (depending on environmental conditions) from first tillage until crop growth provides sufficient cover to buffer erosive forces and shield erodible surface aggregates. All agricultural peat soils are potentially vulnerable to erosion from the time of planting until vegetation cover develops. Measurements on a single surface subject to high levels of vehicle traffic suggest that these may represent important vulnerability hotspots. This is likely to be exacerbated by the long duration of bare soil periods on some such surfaces (e.g., farm tracks/headlands) and the erosive forces generated by vehicles themselves. Mitigation efforts should focus on minimising periods of bare soil and maximising vegetation/residue cover of soil surfaces. Min-/No-till and cover cropping practices should be evaluated for use on agricultural peat soils to minimise bare soil exposure during the crop cycle. Chemical soil stabilisers should also be evaluated for use on peat soils as a possible mitigation option where bare soil is unavoidable. Farm management operations increase the vulnerability of agricultural peatland C to wind erosion during bare soil periods by negatively influencing soil physical properties. However, development and deployment of mitigation strategies should offer tangible improvements.

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Chapter 4

Evaluating the wind erosion mitigation potential of commercially available chemical soil stabilisers for use on lowland agricultural peatlands.

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BWJF conducted the investigation and analysis, and wrote the manuscript. GFSW, DRC, DLJ and CDE reviewed the manuscript.

Abstract

Cropland accounts for ~20% of anthropogenic dust emissions and 64% of European cropland is vulnerable to wind erosion. Peatlands store a third of global soil carbon (C) but have been widely drained. As a result, ~11% of global peatlands are degrading and rates of wind erosion from agriculturally managed peat soils can be substantial. Periods of bare soil resulting from management activities create peaks of erosion vulnerability and this is particularly the case on high soil organic matter (SOM) content peat soils. Chemical soil stabilisers have been widely used for erosion control in industrial/construction environments. They have therefore been proposed as an erosion mitigation option where bare soil is unavoidable but have generally been considered unsuitable for use on peat soils due to their unique physical properties. We undertook a laboratory experiment, using a Portable *In-Situ* Wind Erosion Laboratory (PI-SWERL), to evaluate the potential of several commercially available chemical soil stabilisers for erosion control on high-SOM peat soil. Products tested include acrylic polymers, a Hypromellose emulsion and a tall oil pitch emulsion. The only product to show significant erosion mitigation potential was anionic polyacrylamide (PAM) at a high application rate of 45 kg ha^{-1}. PAM produced a resistant surface crust, which prevented erosion even though the soil contained a large reservoir of erodible aggregates. Lower application rates of PAM were not effective, suggesting high-SOM peat soils require stronger applications than mineral soils. Consequently, PAM may be an expensive option for general erosion control on high-SOM agricultural peat soils. However, it may have utility for managing small areas of acute erosion risk or for one-off treatments to support revegetation of bare areas. Hypromellose emulsion produced mixed results and remains a candidate for further testing but tall oil pitch emulsion performed poorly. Field studies are required to evaluate the persistence of erosion mitigation in the presence of imported abraders, environmental stresses and vehicle traffic. This study represents a first step towards developing chemical soil stabilisation for erosion control on valuable lowland agricultural peat soils.

4.1. Introduction

Global dust emissions are estimated at approximately 2000 Mt yr^{-1} and have approximately doubled since the start of the Industrial Revolution (Shao et al., 2011; Hooper and Marx, 2018). Dust emissions can have disruptive effects on climate regulation (Kok et al., 2017; Schepanski, 2018; Kok et al., 2023), nutrient cycling and ecology (Lawrence et al., 2013; Brahney et al., 2015; Dansie et al., 2022), food security (Lal, 2007; Amundson et al., 2015; Webb et al., 2017) and human health (Griffin et al., 2001; Prospero et al., 2014; Staffogia et al., 2016). Wind erosion occurs when air movement across soil surfaces produces a shear stress sufficient to overcome the cohesive and gravitational forces acting on surface particles, inducing their detachment and transport before subsequent redeposition at another location. Agricultural management increases erosion vulnerability through reductions in vegetation cover and the application of tillage, resulting in soils which are more exposed to erosive forces and are more erodible (Chapter 3; Houyou et al., 2016; Bartkowski et al., 2022). Cropland has been estimated to account for ~20% of anthropogenic dust emissions globally (Chen et al., 2019). Wind erosion of agricultural soils is a widespread issue; for example, 64% of European cropland has been estimated to be eroding to some extent (Borelli et al., 2017). This makes agriculture an important target for wind erosion mitigation measures but currently these remain relatively poorly developed (Maffia et al., 2020).

Peatlands occupy less than 3% of the global land area but store around a third of global soil C (>600 Gt C; Yu et al., 2010; Scharlemann et al., 2014; Xu et al., 2018). However, ~11% of the global peatland area has been drained for productive use (half of this for agriculture) and consequently ~14% of the global peatland C stock occurs in degrading sites (Joosten and Clarke, 2002; Joosten, 2010; Leifeld and Menichetti, 2018). Drainage increases soil organic matter (SOM) mineralisation and nutrient cycling rates, leading to substantial greenhouse gas (GHG) emissions from agricultural peatlands (Freeman et al., 2022). Wind erosion losses from agricultural peatlands are known to be substantial under some circumstances (Parent et al., 1982) but remain poorly quantified (Freeman et al., 2022). Field monitoring studies have estimated horizontal mass transport rates above agricultural peat soils, with values ranging from 230-1880 g m^{-2} yr^{-1} up to 1 m above deep UK agricultural peat and 880-1401 g m^{-2} yr^{-1} up to 2 m above wasted peat (Cumming, 2018; Newman, 2022). However, the only estimate of the net wind erosion rate that we know of for a peatland remains that of Warburton (2003); 0.46-0.48 t ha^{-1} for an isolated patch of bare upland peat in the UK. This is similar to the erosion rate estimated empirically for UK mineral soils under agriculture by Chappell and Warren (2003; 0.6 t ha^{-1} yr^{-1}) and model estimates for European mineral soils under agriculture (0.53 t ha^{-1} yr^{-1}; Borelli et al., 2017). However, it is lower than the empirical estimates of Chappell and Baldock (2016; 4.4 t ha^{-1} yr^{-1}) and Van Pelt et al. (2017; 3.7-6.6 t ha^{-1} yr^{-1}) for erosion rates from dryland agricultural soils in Australia and the USA respectively. Only limited data are available to make an assessment but it seems reasonable to assume that erosion rates from agricultural peat soils may be comparable to those from mineral soils in a similar climate. However, aeolian C losses from peat soils may be higher due to the high C content of SOM and C enrichment of eroded material (Webb et al., 2013).

Management operations can increase the vulnerability of peat soils to wind erosion by modifying dynamic soil physical properties (e.g., aggregate size distribution, moisture content, surface roughness, bulk density, surface crusting; Chapter 3; Campbell et al., 2002; Tissari et al., 2006; Kohake et al., 2010). Additionally, conventional agricultural practices often produce periods of bare soil or sparse vegetation cover associated with tillage, vehicle traffic and the early growth period of crop plants, which increase the exposure of the soil surface to erosive forces and render it more vulnerable to wind erosion (Chapter 3; Funk and Engel, 2015; Maffia et al., 2020). Cultivated high-SOM peat soils can experience a prolonged period of vulnerability, as potential erosion rates from these soils are high from primary tillage throughout the entire management cycle until such time as crop growth provides sufficient cover to prevent erosion (Chapter 3; Funk and Engel, 2015). Whilst increasing vegetation cover and minimising periods of bare soil offer potential for mitigation of erosion losses from agricultural soils (Hagen and Armbrust, 1994; Touré et al., 2019; Lin et al., 2021), some areas/periods of bare soil are unavoidable and as such, alternative mitigation options are needed.

Chemical soil stabilisers are generally high molecular weight organic polymers and are used widely for erosion control on unpaved roads, storage piles, mine tailings and other open surfaces in industrial/construction environments (USEPA, 1992). A wide variety of chemicals have been used for these purposes including among others, acrylic polymers such as polyacrylamide (PAM), polyvinyl acetate and various hydrogels, petroleum resins, wood pulp byproducts and asphalt emulsions. The precise mechanisms of action vary between different chemical soil stabilisers but treatments generally result in increased cohesion of particles in the surface layer and/or creation of a resistant surface crust (Genis et al., 2013; Wei et al., 2021). The potential to reduce wind erosion losses from mineral soils has been demonstrated for PAM (Yang and Tang, 2012; Genis et al., 2013), polyvinyl acetate (Feizi et al., 2019), a range of hydrogels (Yang et al., 2008; Erci et al., 2021; Wei et al., 2021), petroleum resin and asphalt emulsions (Lyles et al. 1974) and wood pulp byproducts (Kavouras et al., 2009; Robichaud et al., 2017; Lee et al., 2020; Preston et al., 2020). However, chemical soil stabilisers have generally been considered unsuitable for erosion control on peat soils (Riksen et al., 2003), though we know of no published studies demonstrating this explicitly. As a result of their high SOM content, peat soils display high total porosity, hydrophobicity, shrinkage/swelling capacity, compressibility and low particle/bulk density relative to mineral soils (Berglund and Persson, 1996; Price and Schlotzhauer, 1999; Schwärzel et al., 2002; Kechavarzi et al., 2010; Kohake et al., 2010; Rezanezhad et al., 2016). These unique features of peat soils potentially pose challenges for chemical soil stabilisation. However, evidence from research in geotechnical engineering shows that chemical soil stabilisers can increase the compressive strength of peat (e.g., Norazam et al., 2017). This suggests that surface chemical stabilisation for erosion control may be possible if an appropriate chemical can be identified.

Near-term costs associated with crop damage and long-term costs associated with SOM losses, alongside a recent policy focus on responsible management of UK agricultural peat soils (Defra, 2021, 2023) provide a strong incentive for farmers to adopt soil loss mitigation measures. Chemical soil stabilisers may provide an option for wind erosion mitigation where protection through increased vegetation cover is not possible but their efficacy has not been demonstrated for peat soils. Our aim with this study was to take a first step on the path towards potentially developing chemical soil stabilisers as an erosion control measure for agricultural peat soils. We evaluated several commercially available products with different chemical compositions, under laboratory conditions. We aimed to identify products with the capacity to stabilise agricultural peat soils (increasing threshold friction velocities of entrainment and reducing wind erosion rates), in order to create a candidate set of chemicals for future field testing. It is hoped that this work will assist land managers working to protect valuable lowland agricultural peat soils.

4.2. Study area

A detailed description of the study area can be found in Chapter 3. Briefly, the East Anglian Fens are the largest area of lowland peatland in the United Kingdom and have a long history of agricultural drainage (Sly, 2010). Having once covered ~150,000 ha, the area of deep peat soil in the Fens has been greatly reduced by drainage induced subsidence, so that only an estimated 16,500 ha of peat deeper than 40 cm remain (Holman, 2009; Evans et al., 2016). Drainage has also produced highly productive soils and, consequently, the Fens account for ~7% of England's total agricultural production and support a regional food industry worth £2.3 billion (NFU, 2019). East Anglia is a region with high vulnerability to wind erosion and severe erosion events known as 'Fen blows' have long been reported (Thompson, 1957; Borelli et al., 2016). Erosion rates for mineral soils in East Anglia have been estimated to average 0.6 t ha^{-1} yr^{-1} with maximum rates of 32.6 t ha^{-1} yr^{-1} by Chappell and Warren (2003). High maximal rates were balanced by high rates of deposition in other locations, which resulted in the relatively low average rate for their study area but demonstrated that some locations can experience severe erosion losses.

Within this regional context, our study site was Rosedene Farm (52°32'06''N 0°27'27''E; Figure 4.1a), which produces high-value salad vegetables on deep, high SOM content (~65%) peat soil. Salad vegetable production is especially vulnerable to wind erosion as ground cover is limited during the early stages of crop growth in spring/summer (Figure 4.1f). Erosion at this time can also negatively affect crop yield and quality due to soil damage and contamination of plants (Figure 4.1g; Genis et al., 2013). Tillage and first crop planting occur predominately during the spring, which coincides with high wind speeds in the region, further exacerbating erosion risks (Newman, 2022). Erosion mitigation measures used at Rosedene Farm include minimisation of the time interval between tillage and planting, fleecing of prepared planting beds, companion cropping, retention of crop residues on the field surface after harvest, management for small field sizes and planting of hedgerows/tree shelterbelts (Schultz and Carlton, 1959; Funk and Engel, 2015; Chang et al., 2021). However, exposed/bare soil is unavoidable at times, for example following primary tillage, during the early period of crop growth, and on areas of high vehicle traffic and farm tracks (Figure 4.1). Rosedene Farm therefore provides an excellent example of the context in which chemical soil stabilisers could have high utility as a wind erosion mitigation strategy if effective options can be developed.

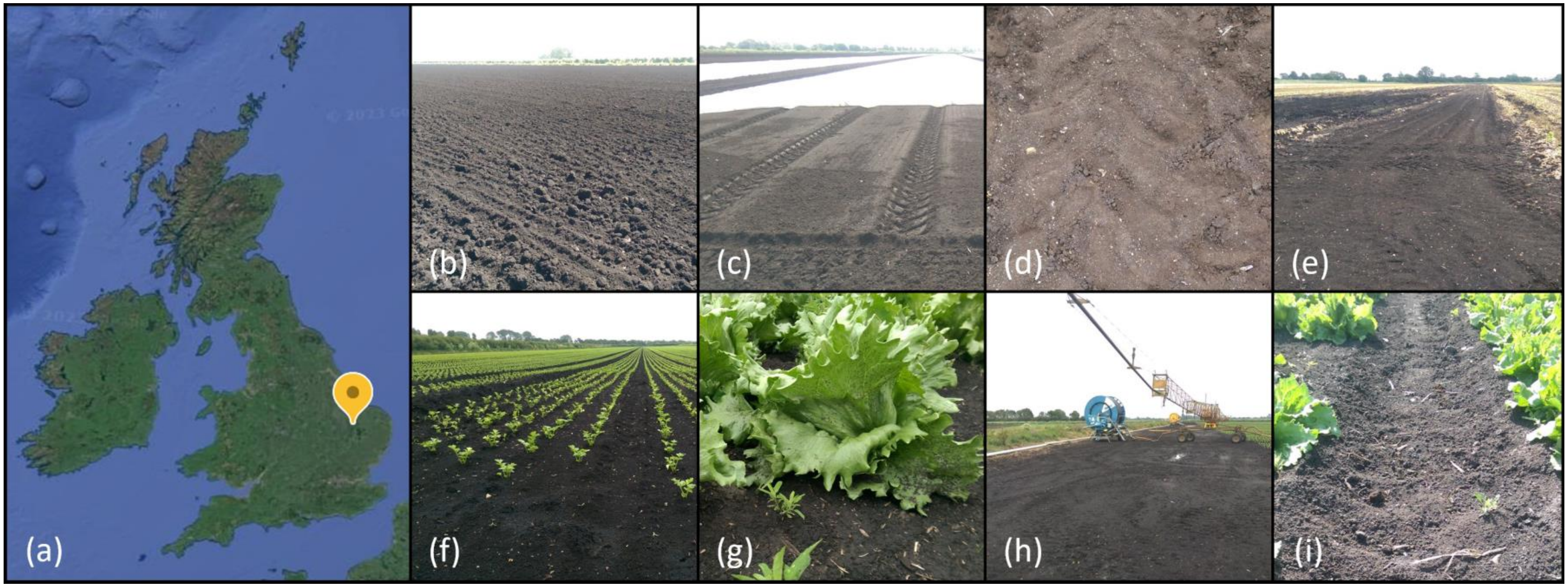

Figure 4.1. Study context. (a) Location of Rosedene Farm within United Kingdom indicated by marker [Source: Google Earth v9.191.0.0 WebAssembly, Accessed: July 2023, https://earth.google.com/web/]. (b) Bare soil after primary tillage. (c) Secondary tillage produces smooth fine textured planting beds – note fleece covering to mitigate erosion. (d) Vehicle tracks produce compaction and disaggregation to fine erodible particles. (e) Vehicle traffic following harvest can result in bare soil areas even when crop residue is left in the field. (f) Bare soil exposed between juvenile crop plants. (g) Soil contamination of lettuce crop plant. (h) Headlands experience heavy traffic and can become compacted and vulnerable to erosion – note the irrigation boom; irrigation also contributes to disaggregation and surface crusting of agricultural peat soils. (i) Tramlines where vehicle wheels pass through crops remain bare and may become within-field dust reservoirs.

4.3. Methods

4.3.1. Experimental design

Topsoil (0-30 cm depth) was collected from four fields on Rosedene Farm, transported to Henfaes Research Centre, Bangor, UK and stored in covered bulk bags, before being manually placed into plastic trays (52 x 42 x 9 cm). Trays were stored indoors until treatment application to minimise confounding environmental effects on surfaces. Due to disturbance/mixing during collection, the physical properties of prepared control surfaces closely resembled those of the field surface subject to primary tillage from Rosedene Farm reported in Chapter 3 (Surface PT_{70}; Table 4.1).

Six commercially available chemical soil stabilisers with potential utility for agricultural peatlands were procured. Products P1, P41a, P4b and P5 were acrylic polymers (Table 4.2). Acrylic polymers are long hydrocarbon chains which interlink and form electrostatic linkages with soil aggregates to form a matrix. Cohesion between molecules and soil particles produces a resistant surface crust. Product P2 was a tall oil pitch emulsion (Table 4.2). Tall oil pitch is a sticky, viscous substance produced as a byproduct of the paper pulping process. Emulsification reduces viscosity, allowing products to penetrate soil pore spaces, before curing to bind aggregates together. Product P3 was a Hypromellose emulsion (Table 4.2), which again can be produced as a wood pulp byproduct and penetrates soil pore spaces before curing to bind aggregates. Applications followed manufacturers' recommendations.

All products except P4a (granular solid) were sourced as liquid preparations. P4a was prepared by mixing with water prior to application. As all treatments were applied as aqueous dilutions, we included a comparator treatment applying the same volume of water for each product, to differentiate between the effects of the product and the effects of the water used in the application. All products and water volumes except P5/W5 were applied using handheld spray bottles (Table 4.2). Due to the higher viscosity preparation of product P5 a watering can with a rose was used for the application of P5 and W5. A control treatment included trays which were prepared in the same manner but had no treatment applied. There were twelve treatment conditions in total: One untreated control, six chemical stabiliser treatments and five equivalent volume water applications (as P4a and P4b used the same water application volume). Product P2 required a 48-hour dry curing period in warm conditions. Therefore, all trays were held in a large grain-drying oven for 48 hours at 30°C before measurement. Positioning in the oven was randomised to mitigate potential systematic effects arising from any temperature gradients present within the oven.

Table 4.1. Summary of soil physical properties for prepared control surfaces. Presented as a comparison with a field condition surface from the same farm following primary tillage (Surface PT_{70}; Chapter 3). Data presented as arithmetic means and standard errors.

Product	**Prepared (Control)**	**Field (Ploughed)**
Bulk density (g cm^{-3})	0.28 ± 0.01	0.33 ± 0.03
Soil organic matter content (%)	71.90 ± 5.94	70.45 ± 1.91
Win erodible material (%)	31.75 ± 4.51	28.91 ± 2.93
Dry aggregate stability (%)	91.33 ± 1.40	91.92 ± 1.31
Volumetric water content (%)	25.39 ± 2.94	23.36 ± 2.68

Table 4.2. Summary of experimental conditions. All chemical soil stabiliser products were sourced as liquid preparations except for P4a, which was sourced as a granular solid.

Product	Product description	Product application rate (kg DM ha^{-1})	Water application rate (mm)
C	Untreated	-	-
P1	Anionic acrylic polymer	27	1.0
P2	Tall oil pitch emulsion	187	1.4
P3	Hypromellose emulsion	584	2.7
P4a	Anionic polyacrylamide (Granular)	1	3.5
P4b	Anionic polyacrylamide	7	3.5
P5	Anionic polyacrylamide	45	10.0
W1	Water only	-	1.0
W2	Water only	-	1.4
W3	Water only	-	2.7
W4	Water only	-	3.5
W5	Water only	-	10.0

4.3.2. Wind erosion measurements

We used a Miniature Portable *In-Situ* Wind Erosion Laboratory (PI-SWERL) to assess the vulnerability to wind erosion of peat soil in the prepared trays (Figure 4.2). The PI-SWERL has been tested and calibrated across a range of environments and produces dust emission estimates comparable to field wind tunnels (Sweeney et al., 2008; Sweeney et al., 2011; Sweeney and Mason, 2013; von Holdt et al., 2021; Vos et al., 2021; Sweeney et al., 2022). The PI-SWERL's relatively short operation period means it is well-suited to our study objectives, where replicable measurements were required across a large number of treatment conditions (e.g., Kavouras et al., 2009). The operating principles of the PI-SWERL have been described comprehensively in (Etyemezian et al., 2007; Sweeney et al., 2008; Sweeney and Mason, 2013). Saltation rate was estimated as a count of saltating particles using optical gate sensors mounted within the chamber. The concentration (D; mg m^{-3}) of particulate matter <10μm in diameter (PM_{10}) in the exhaust flow from the chamber was measured using an attached nephelometer style dust monitor (DustTrak II model 8530). The introduction of clean air into the chamber at a constant rate (F; m^3 s^{-1}) and the known effective area for the PI-SWERL (A_{eff} = 0.026 m^2) allow calculation of a mean PM_{10} emission flux (E; mg m^{-2} s^{-1}; Equation 4.1) over the test duration ($t_{end,i} - t_{begin,i}$) by cumulative summation of fluxes across 1s (t_0) intervals.

Equation 4.1:

$$E_i = \frac{\sum_{begin,i}^{end,i}(D_i \times F \times t_0)}{\left(t_{end,i} - t_{begin,i}\right) \cdot A_{eff}}$$

We operated the PI-SWERL using a pre-defined, hybrid test program including eight distinct phases (Figure 4.3). Phase F1: Stable operating speed of 0 RPM (revolutions per minute) for 210 s. Phase R1: Increase from 0 to 3000 RPM over 180 s. Phase S1: Stable operating speed of 3000 RPM for 90 s. Phase R2: Increase from 3000 to 4000 RPM over 60 s. Phase S2: Stable operating speed of 4000 RPM for 90 s. Phase R3: Increase from 4000 to 5000 RPM over 60 s. Phase S3: Stable operating speed of 5000 RPM for 90 s. Phase F2: Stable operating speed of 0 RPM for 60 s. Phase F1 was a flush phase to allow dust concentrations to return to baseline levels before measurements began, following any disturbance from chamber placement. Phases R1, R2 and R3 were ramp phases with gradually increasing RPM to allow determination of the threshold friction velocity of entrainment of soil particles (u^*_t; m s^{-1}). We defined u^*_t for saltation as a sustained increase in the 20 s moving average of saltation rate above background noise (7 particles counted per second; pc s^{-1}; Sweeney and Mason, 2013; Cui et al., 2019). We applied a similar approach to identifying u^*_t for PM_{10} but used the mean measured dust concentration over the 30 s period preceding each measurement as a baseline. Phases S1, S2 and S3 were step phases used to estimate mean PM_{10} fluxes and saltation rates at a range of friction velocity (u^*) values (0.55, 0.69 and 0.82 m s^{-1}). The 5000 RPM value used for the final step phase (S3) was chosen because the u^* of 0.82 m s^{-1} at 5000 RPM, calculated using Equation 4.2 (Dust-Quant LLC, 2011) was comparable to the u^* used by Zobeck et al. (2013; 0.88 m s^{-1}) for wind tunnel measurements on agricultural peat soils. This value was also comparable to the higher end of the wind speed range recorded for Rosedene Farm by Cumming (2018; 9-12 m s^{-1}). The cumulative total PM_{10} emissions for the entire test period were also calculated, as material eroded during earlier phases would not be available to contribute to mean erosion rate estimates for later steps.

Equation 4.2:

$$u^* = -1.49E-12 \cdot RPM^3 + 9.20E-09 \cdot RPM^2 + 1.42E-04 \cdot RPM + 0.0872$$

Phase F2 was a final flush phase that allowed dust concentrations in the chamber and DustTrak to drop before the end of the measurement run. Between experimental runs the PI-SWERL chamber was carefully brushed clean before being placed on a clean surface whilst we ran a cleaning program (0 RPM for 30 s, ramp from 0-5000 RPM over 30 s, 5000 RPM for 60 s, 0 RPM for 30 s) to minimise dust contamination effects between measurements. The DustTrak is calibrated to Arizona Road Dust standard (ISO 12103-1). Therefore, the PM_{10} mass flux was adjusted using particle density (ρ_P) and C content, to give the PM_{10}-C flux and facilitate comparison with previous field measurements (Chapter 3). We made measurements on four separate days, with one replicate from each treatment measured on each day. Measurements on each day were made in the order of anticipated increasing PM_{10} emissions, to minimise the potential for contamination of low emission runs. The same test program was used on all surfaces. One replicate of condition P4b was discontinued to avoid passing an excessive dust load through the DustTrak.

Figure 4.2. Demonstration of the experimental setup. Shows the PI-SWERL chamber in position on a prepared tray.

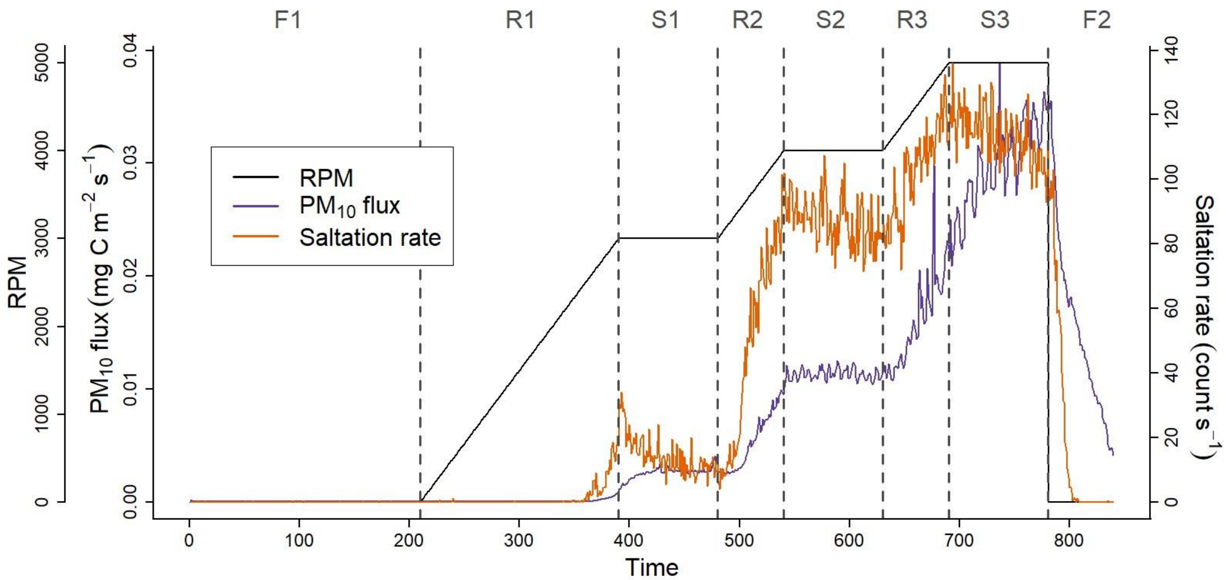

Figure 4.3. Example PI-SWERL measurement. Timeline of a test program from an untreated control treatment replicate with RPM, PM_{10} fluxes and saltation rate shown at 1 s intervals. The test program was broken into several phases indicated by vertical dashed lines and labelled above the figure. These phases were chosen to allow (F1) dust concentrations to return to the baseline before measurements commenced, (R1, R2, R3) estimation of the threshold friction velocity of entrainment, (S1, S2, S3) estimation of mean PM_{10} emissions/saltation rate at stable friction velocities (0.55, 0.69, 0.82 m s^{-1} respectively) and (F2) dust concentrations to drop to facilitate PI-SWERL cleaning between measurements.

4.3.3. Soil physical properties

Following each wind erosion measurement, surface shear strength was estimated using a handheld shear vane on the remaining surface area of the tray which had not been disturbed by the PI-SWERL footprint. Samples of undisturbed surface soil were also collected and stored in rigid plastic containers, before being air-dried and flat-sieved using a vibratory device with a fixed vibration intensity (Endecotts Minor 200; 1.6 mm amplitude; 50 Hz). The wind erodible material (WEM) was estimated as the percentage of soil mass <0.85 mm in diameter (U.S.A. Standard test sieve No. 20; Chepil, 1950; Kohake et al., 2010; Zobeck et al., 2013). Dry aggregate stability (DAS) was estimated by re-sieving the fraction >0.85 mm (Zobeck et al., 2013). Determination of ρ was undertaken by inserting a metal ring (H = 5 cm, V = 100 cm^3) into the soil adjacent to the PI-SWERL footprint and oven drying the sample (105 °C; 24 h). Volumetric water content (VWC) was estimated by oven-drying (105°C; 24h) and SOM content by loss-on-ignition (450°C; 16h). We calculated ρ_P (g cm^{-3}) using Equation 4.3 (Rühlmann et al., 2006; where SOM_R is SOM content expressed as kg kg^{-1}) and we estimated C content (g kg^{-1}) using Equation 4.4, where SOM is expressed in g kg^{-1} (Wright et al., 2008).

Equation 4.3:

$$\rho_P = \left[\frac{SOM_R}{1.127 + 0.373 \cdot SOM_R} + \frac{1 + SOM_R}{2.684}\right]^{-1}$$

Equation 4.4:

$$C = 0.516 \cdot SOM - 18.1$$

4.3.4. Statistical analysis

All analyses were performed using R v4.2.2 (R Core Team, 2022). Where necessary, dependent variables were log/square root transformed for linear models, or non-linear/non-parametric approaches were used to account for skewed distributions. Pre-planned contrasts were used to maximise statistical power available for pairwise comparisons. Each treatment was compared only to the control and to the equivalent water volume for that treatment, using t-tests with the Sidak correction for multiple testing. Mann-Whitney U tests were used when the means of all water treatments (pooled) were compared with controls. Correlations were examined using Spearman's Rho with the Holm correction for multiple tests.

4.4. Results

4.4.1. Soil physical properties

There was little overall effect of treatments on soil physical properties with the exception of surface shear strength ($p<0.0001$). Surface shear strength was significantly higher than untreated controls for treatments P3 ($p=0.0008$), P5 ($p<0.0001$), W3 ($p=0.006$), W4 ($p=0.027$) and W5 ($p<0.0001$; Figure 4.4). However, only product P5 exhibited a significantly higher surface shear strength than the equivalent volume of water-only treatment (W5; $p<0.0001$). There were no statistically significant effects of chemical soil stabiliser treatments or equivalent volume water treatments relative to untreated controls for ρ ($p = 0.17$), DAS ($p = 0.80$), VWC ($p = 0.57$) or WEM ($p = 0.65$). However, three of four replicates for treatment P5 did show low WEM values compared to both W5 and controls. The remaining replicate displayed consistently low WEM values across all treatments and the lowest WEM values of any replicate, which could possibly be an artefact of surface preparation. WEM was negatively correlated with DAS (Appendix 4.1).

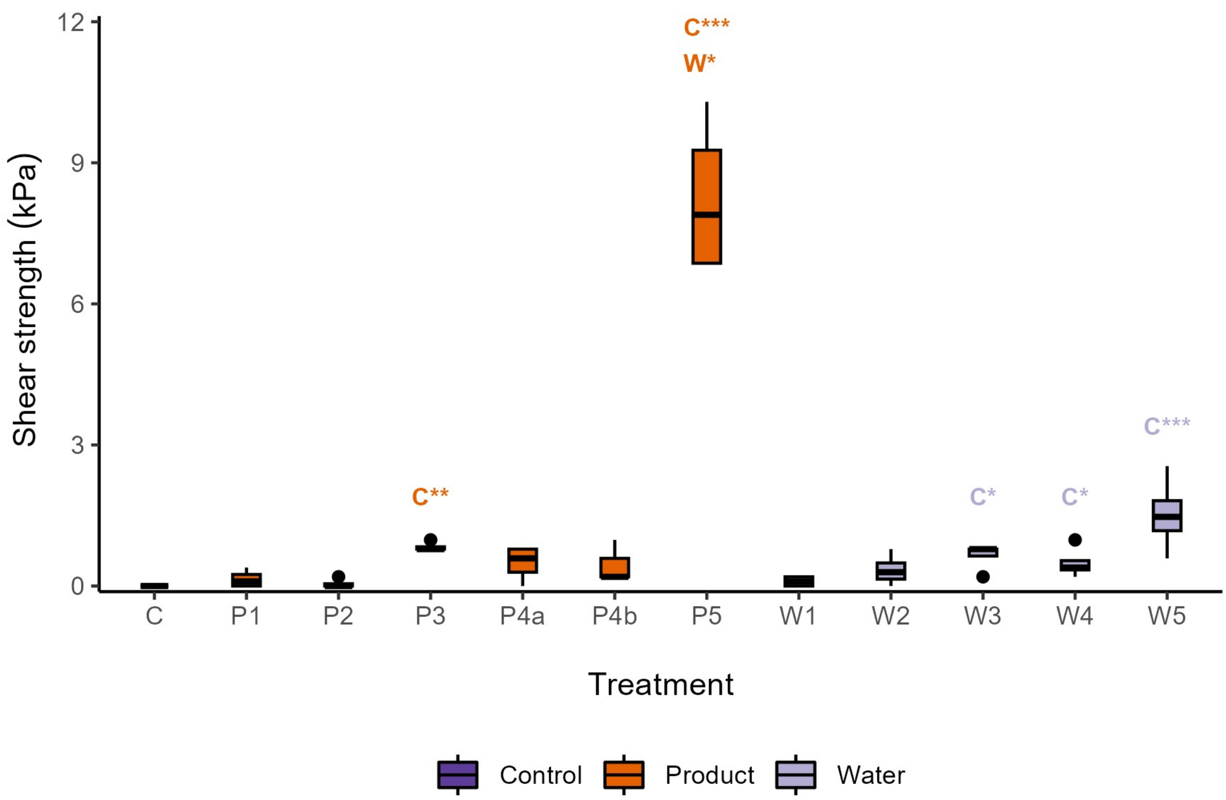

Figure 4.4. Surface shear strength by treatment. Annotations above boxes indicate the results of pre-planned contrasts: C indicates significant difference from untreated control, W indicates significant difference from the equivalent volume water treatment for chemical soil stabiliser treatments, * = $p<0.05$, ** = $p<0.001$, *** = $p<0.0001$. Data presented as a boxplot. Points indicate values more than 1.5 times the interquartile range from the median.

4.4.2. Threshold friction velocity of entrainment

The mean u^*_t observed for PM_{10} emissions (0.52 ± 0.01 m s^{-1}) and saltation (0.52 ± 0.01m s^{-1}) from control surfaces in this study are very similar to the mean of the u^*_t estimates of Kohake et al. (2010; 0.56 ± 0.09 m s^{-1}) for wind tunnel tests on trays prepared using agricultural peat soils. However, our estimates are slightly higher than values we previously obtained for erodible field surfaces for PM_{10} emissions (0.34 ± 0.02 m s^{-1}) and saltation (0.41 ± 0.01 m s^{-1}; Chapter 3). This difference likely reflects the substantially weaker forces applied to aggregates during manual tray preparation processes in comparison to those applied by farm machinery/irrigation under field conditions. The field sample in Chapter 3 included surfaces subject to tillage, irrigation and heavy traffic and the difference in u^*_t persists even for the field surface with very similar soil physical properties (Table 4.1; Table 4.3).

Treatment with water (all application volumes pooled) significantly increased u^*_t relative to untreated controls for both PM_{10} emissions (38% increase; $p = 0.001$; Figure 4.5b) and saltation (40% increase; $p = 0.001$). Generally, the effect on u^*_t of chemical stabiliser treatments was indistinguishable from equivalent water volumes (Figure 4.5a, 4.5c). However, treatment P5 (high application rate of anionic PAM) noticeably increased the u^*_t of both PM_{10} emissions and saltation as neither threshold was reached for P5 in any replicate. The u^*_t for PM_{10} emissions and saltation were not reached for two of the four replicates in treatment P3 (Hypromellose). However, the mean u^*_t of the remaining replicates (0.72 m s^{-1}) was similar to that for the equivalent water volume (0.75 m s^{-1}). Whilst treatment P2 (Tall oil pitch emulsion) significantly increased the u^*_t of both PM_{10} emissions ($p = 0.008$) and saltation ($p = 0.006$) relative to controls, it significantly underperformed water alone (W2; PM_{10}: $p = 0.005$; Saltation: $p = 0.006$; Figure 4.5d).

Overall, the u^*_t for saltation and PM_{10} emissions were strongly correlated (Rho = 0.98; $p<0.0001$), and the mean difference between the u^*_t for saltation and PM_{10} emissions was very small (0.008 ± 0.003 m s^{-1}), suggesting that initiation of saltation and dust suspension were approximately simultaneous. For surfaces where u^*_t could be determined it was negatively related to WEM and SOM content and positively related to shear strength for both PM_{10} emissions ($p = 0.001$) and saltation ($p = 0.0004$; Appendix 4.2).

Table 4.3. Comparison of erosion variables between prepared control surfaces and a field condition surface from the same farm following primary tillage. Comparator is Surface PT_{70}, reported in Chapter 3. Threshold friction velocities (u^*_t) of saltation and PM_{10} dust emissions presented as arithmetic means and standard errors. Mean PM_{10}-C flux and saltation rate at $u^* = 0.82$ m s^{-1} presented as geometric means and standard error ranges.

Product	**Prepared (Control)**	**Field (Ploughed)**
Mean PM_{10}-C flux (mg C m^{-2} s^{-1})	0.007 (-0.002, +0.002)	0.021 (-0.004, +0.005)
Mean saltation rate (count s^{-1})	132.12 (-7.94, +8.45)	64.55 (-20.71, +30.49)
u^*_t PM_{10} (m s^{-1})	0.52 ± 0.01	0.41 ± 0.03
u^*_t saltation (m s^{-1})	0.52 ± 0.01	0.43 ± 0.01

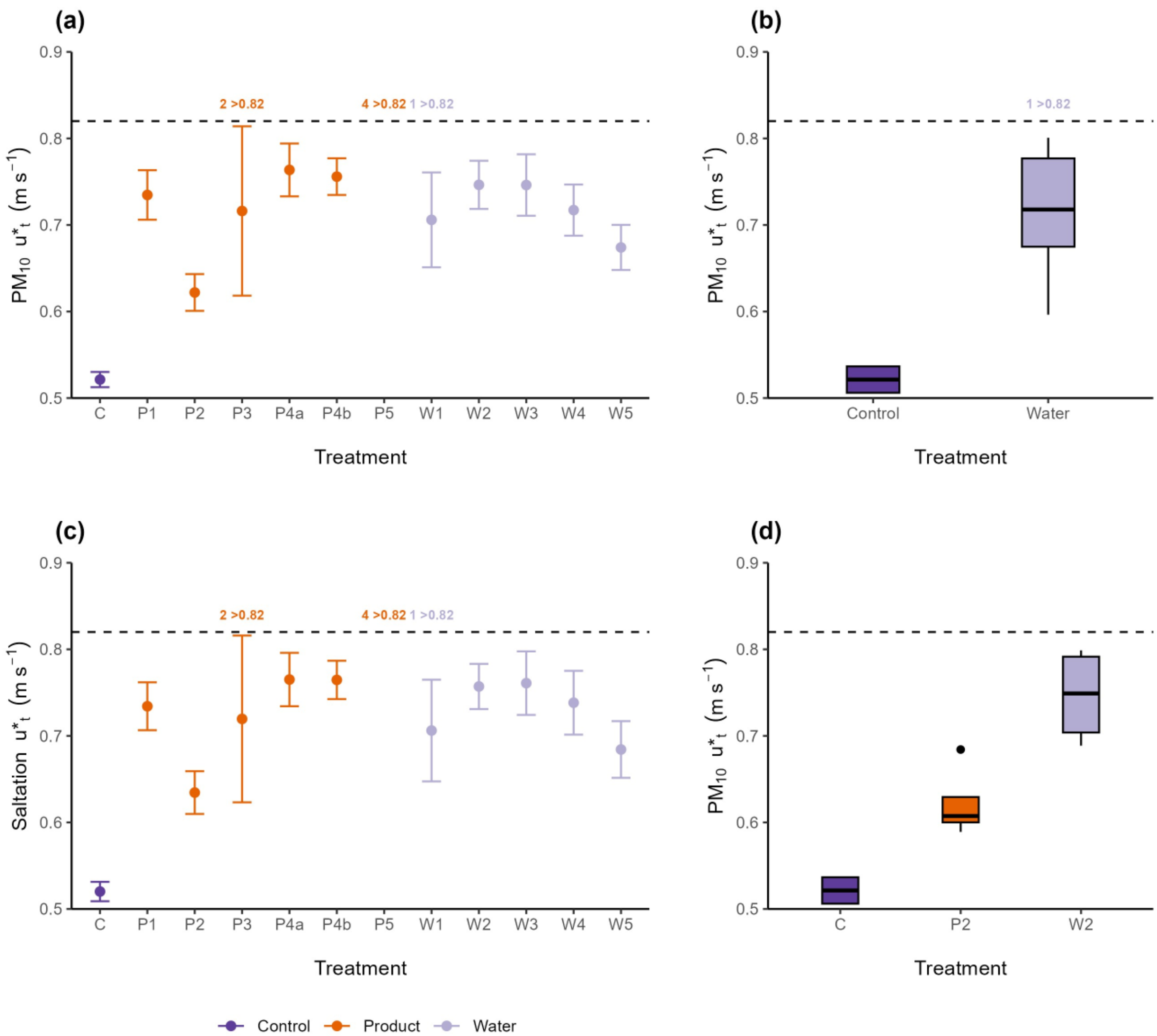

Figure 4.5. Threshold friction velocity (u^*_t) estimates. (a) u^*_t of PM_{10} emissions by treatment. (b) u^*_t of PM_{10} emissions for all water treatments (pooled) compared to control. (c) u^*_t of saltation by treatment. (d) u^*_t of PM_{10} emissions for treatment P2 compared to controls and equivalent volume water application. Data in (a) and (b) presented as arithmetic means (points) and standard errors (bars). Data in (b) and (c) presented as boxplots. Points on boxplots indicate values more than 1.5 times the interquartile range from the median. The horizontal dotted lines indicate the maximum friction velocity applied during the test (0.82 ms^{-1}). For some treatments erosion was not observed at $u^* \leq 0.82$ ms^{-1} for some or all replicates. In these cases, the number of replicates for which erosion was not observed at $u^* \leq 0.82$ ms^{-1} is indicated above the maximum friction velocity line.

4.4.3. Erosion rates

The mean PM_{10}-C flux at $u^* = 0.82$ m s^{-1} from control surfaces (0.007 ± 0.002 mg C m^{-2} s^{-1}) was lower than the mean of the PM_{10} fluxes recorded by Zobeck et al. (2013; adjusted using mean site C content; $u^* = 0.88$ m s^{-1}) for agricultural peat soils prepared with a turf roller (0.029 ± 0.011 mg C m^{-2} s^{-1}) and also lower than the field surface with similar soil physical properties from Chapter 3 (Table 4.1; Table 4.3). Given the higher saltation rate of control surfaces in this study relative to the similar field surface from Chapter 3 (Table 4.3), it would appear that manual tray preparation applies weaker mechanical action than tillage, resulting in a positively shifted aggregate size distribution, where a larger proportion of erodible material in our prepared trays is sand-sized rather than PM_{10} sized.

Overall, only product P5 (high application rate of anionic PAM) showed robust erosion mitigation potential. At the lowest u^* step (Phase S1; 0.55 m s^{-1}), PM_{10}-C fluxes were significantly lower for all treatments when compared to untreated controls ($p<0.0001$) but there were no differences between products and equivalent water volume treatments (Figure 4.6a). At the intermediate u^* step (Phase S2; 0.69 m s^{-1}), treatment geometric mean PM_{10}-C fluxes were generally low apart from the control treatment and P2 (Figure 4.6b). However, there was within treatment variability, with some individual replicates demonstrating higher erosion rates. There was a significant overall effect of treatment on mean PM_{10}-C flux ($p = 0.04$) for step S2, reflecting that treatment P5 had significantly lower mean PM_{10}-C flux than the control ($p = 0.04$). However, at this u^*, the mean PM_{10}-C flux from P5 was not significantly different from that for the equivalent volume of water (W5; $p = 0.51$). At the highest u^* step (Phase S3; 0.82 m s^{-1}), all treatments except P5 showed evidence of substantial erosion in at least some replicates (Figure 4.6c). There was a significant overall effect of treatment on mean PM_{10}-C flux ($p<0.0001$). Treatment P5 PM_{10}-C fluxes were not significantly greater than zero ($p=0.059$) and were significantly lower than both the control treatment ($p<0.0001$) and the equivalent volume of water (W5; $p<0.0001$). There were no significant differences between any other treatments at this high u^* value.

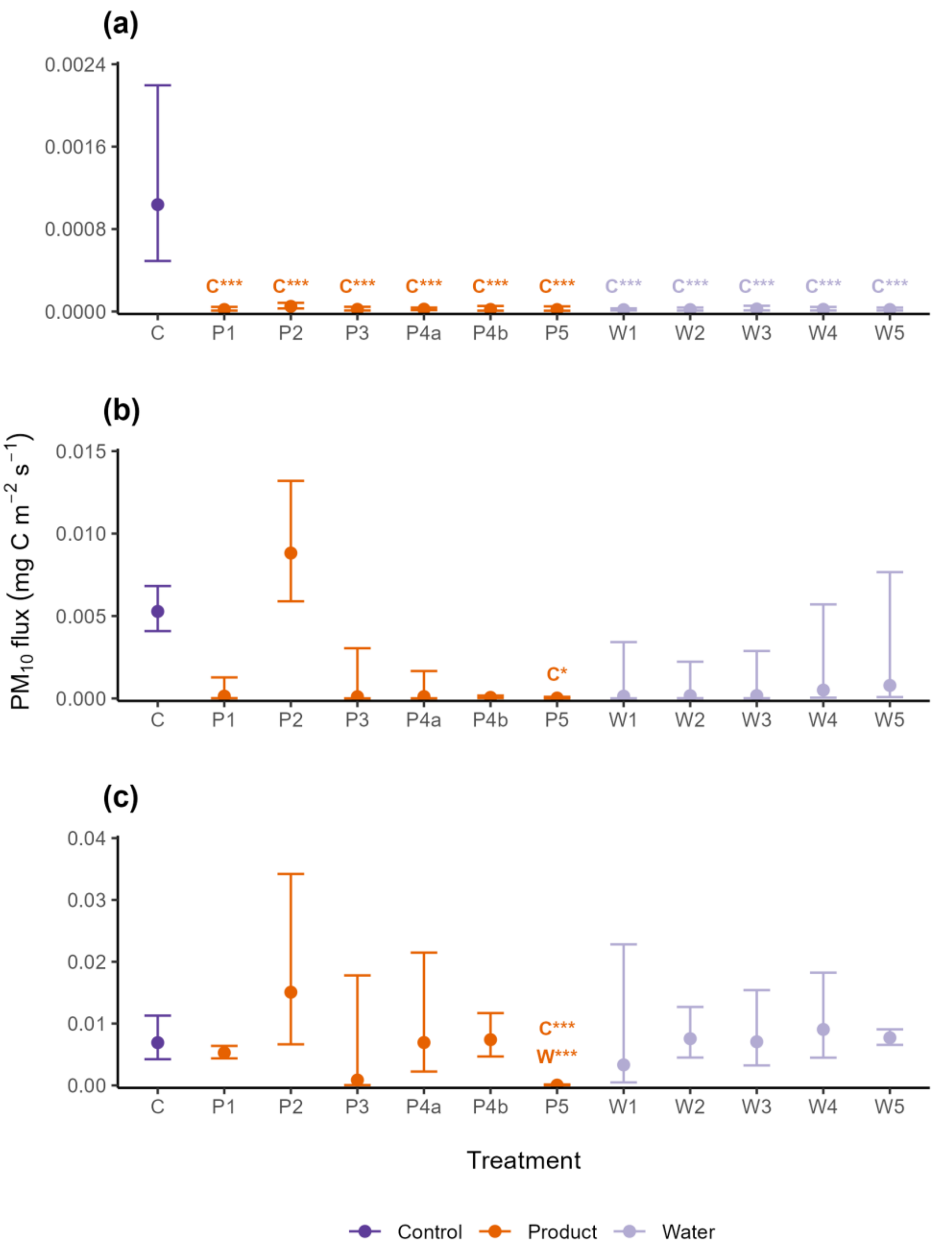

Figure 4.6. Mean PM_{10} carbon fluxes during step periods. (a) S1, $u^* = 0.55$ ms^{-1}. (b) S2, $u^* = 0.69$ ms^{-1}. (c) S3, $u^* = 0.82$ ms^{-1}. Data presented as geometric means (points) and geometric standard deviations (error bars). Annotations indicate results of pre-planned contrasts: C indicates significant difference from untreated control, W indicates significant difference from the equivalent volume water treatment for chemical soil stabiliser treatments, * = p<0.05, *** = p<0.0001. Note the differences in y-axis scale between panels.

Cumulative PM_{10}-C emissions were also evaluated to account for potential suppression of the mean flux for the 0.82 m s^{-1} step (Phase S3) on eroding surfaces, due to the potential for supply limitations due to emission of erodible material in earlier phases (R1, S1, R2, S2, R3). The results followed a similar pattern to those from the 0.82 m s^{-1} step (S3; Figure 4.7a). There was a significant overall effect of treatment on cumulative PM_{10}-C emissions over the measurement duration ($p = 0.0002$). Treatment P5 had significantly lower cumulative PM_{10}-C fluxes than both the untreated control treatment ($p = 0.0001$; Figure 4.7a) and equivalent water volume (W5; $p = 0.0006$). There were no significant differences in cumulative PM_{10}-C emissions between any other treatments. Cumulative PM_{10}-C emissions were not significantly lower from water-only treatments (all application volumes pooled) than untreated control trays ($p = 0.06$; Figure 4.7b). However, the cumulative saltation count was significantly lower from water-treated trays than control trays ($p = 0.0006$; Figure 4.7c), indicating some erosion mitigation effect.

There was generally strong agreement between the inter-treatment patterns of PM_{10}-C fluxes and saltation rates at all u^* values (Appendix 4.3). Cumulative PM_{10}-C emissions and cumulative saltation particle count displayed a strong positive relationship ($p<0.0001$; Figure 4.8). Saltation and PM_{10}-C related dependent variables were generally strongly intercorrelated, except for the 0.55 m s^{-1} step (Phase S1) mean PM_{10}-C flux and saltation rate, which results from the fact that most treatments exhibited u^*_t values >0.55 m s^{-1} (Appendix 4.1). Taken alongside the u^*_t data, these results suggest that for highly friable high-SOM peat soils, recorded saltation predominately represents transport of organic microaggregates (and not sand-sized mineral grains), which experience abrasion/disintegration during collisions with other soil aggregates and PI-SWERL surfaces, resulting in dust production.

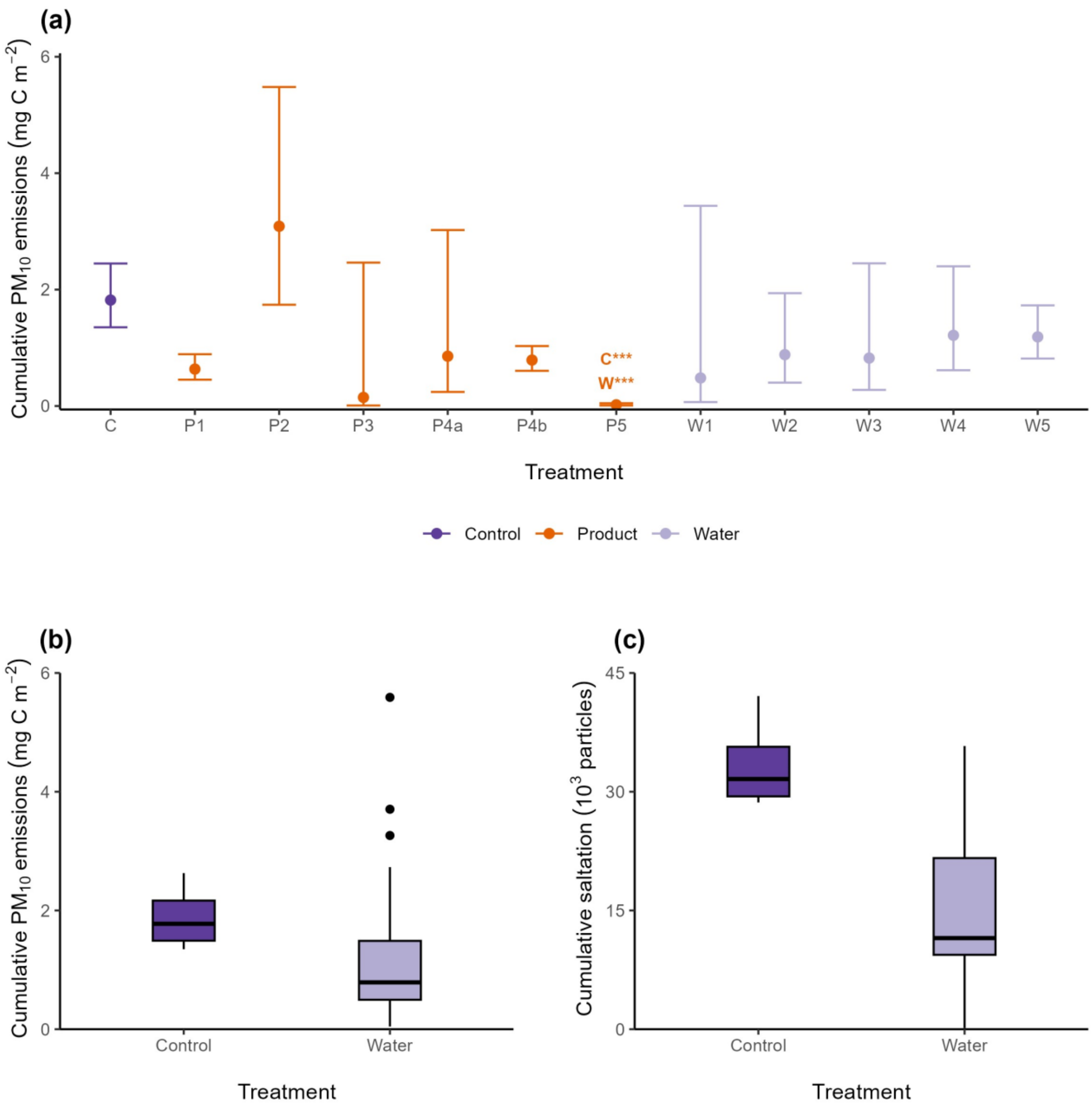

Figure 4.7. Cumulative PM_{10} carbon emissions and saltation counts. (a) Cumulative PM_{10}-C emissions by treatment. (b) Comparison of cumulative PM_{10}-C emissions between water treatments (all volumes pooled) and untreated controls. (c) Comparison of cumulative saltation particle count between water treatments (all volumes pooled) and untreated controls. Data in (a) presented as geometric means (points) and geometric standard deviations (error bars). Annotations indicate results of pre-planned contrasts: C indicates significant difference from untreated control, W indicates significant difference from the equivalent volume water treatment for chemical soil stabiliser treatments, *** = p<0.0001. Data in (b) and (c) presented as boxplots. Points in boxplots indicate values more than 1.5 times the interquartile range from the median.

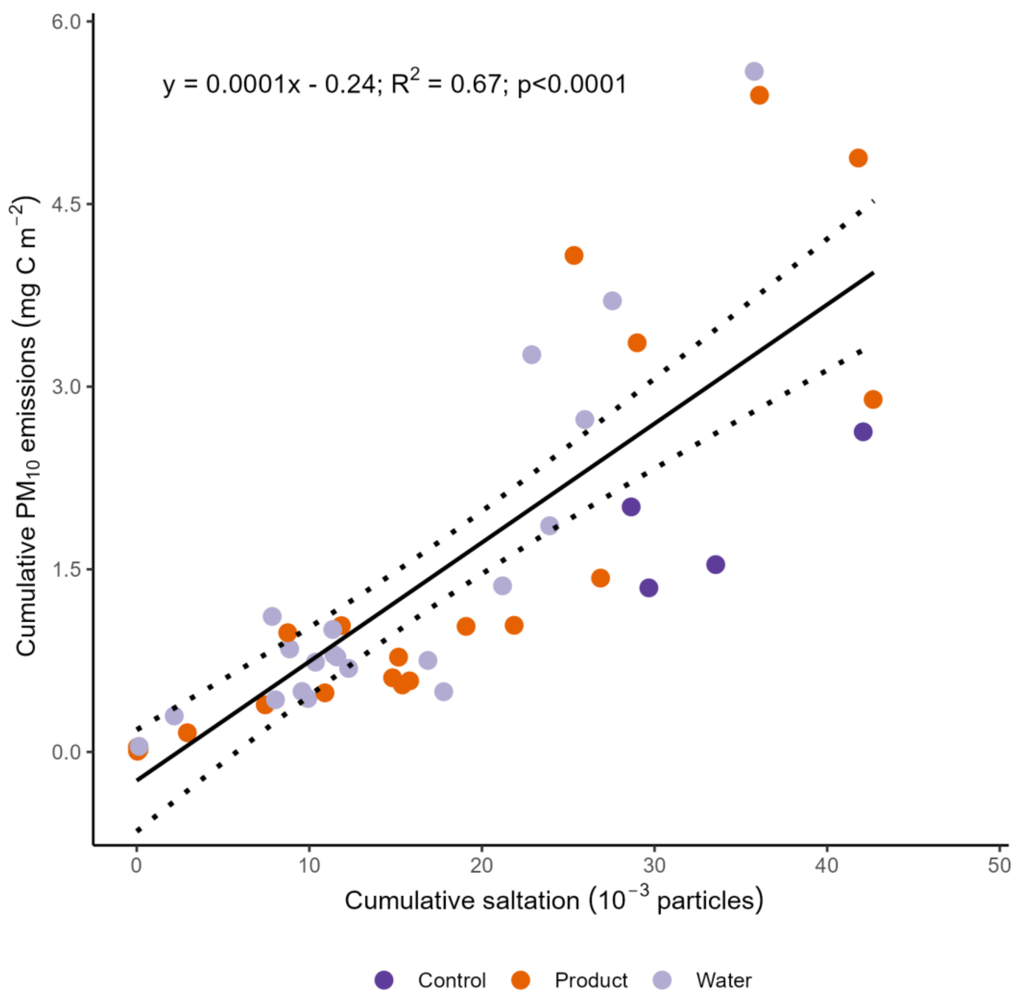

Figure 4.8. Relationship between cumulative PM_{10} carbon emissions and cumulative saltation count. Solid line shows mean linear regression predictions and dotted lines show 95% confidence intervals.

The relatively strong surface crust formed following treatment application was clearly responsible for the low erosion rates observed on surfaces from treatment P5. After removing these points from the dataset, the cumulative saltation count observed from the remaining surfaces was relatively well explained by a linear model including the proportion of WEM, SOM content and surface shear strength as predictors ($p<0.0001$; $R^2 = 0.52$; Table 4.4; Figure 4.9). Cumulative saltation count was positively predicted by SOM content and the proportion of WEM, indicating that erosion rates were higher when a larger supply of erodible material was available. The shear strength of the surface was negatively related to cumulative saltation count suggesting that in our sample even weak crusts (<2.55 kPa) offered some resistance to erosive forces. For surfaces where u^*_t could be determined, cumulative PM_{10}-C emissions were strongly negatively correlated with the u^*_t for PM_{10} (Rho = -0.77; $p<0.0001$) and the cumulative saltation count was even more strongly correlated with the u^*_t for saltation (Rho = -0.91; $p<0.0001$).

Table 4.4. Parameters of linear model describing cumulative saltation rate. For all measurements except those for treatment P5 ($n = 43$; $p<0.0001$; $R^2 = 0.52$).

Term	Fitted range	Estimate	Standard error	t-value	p-value
Intercept		-40988	10419	-3.93	0.0003
SOM (%)	50.5 - 79.5	619	123	5.04	<0.0001
WEM (%)	9.7 – 46.7	700	136	5.16	<0.0001
Shear strength (kPa)	0 – 2.55	-7841	2415	-3.25	0.0024

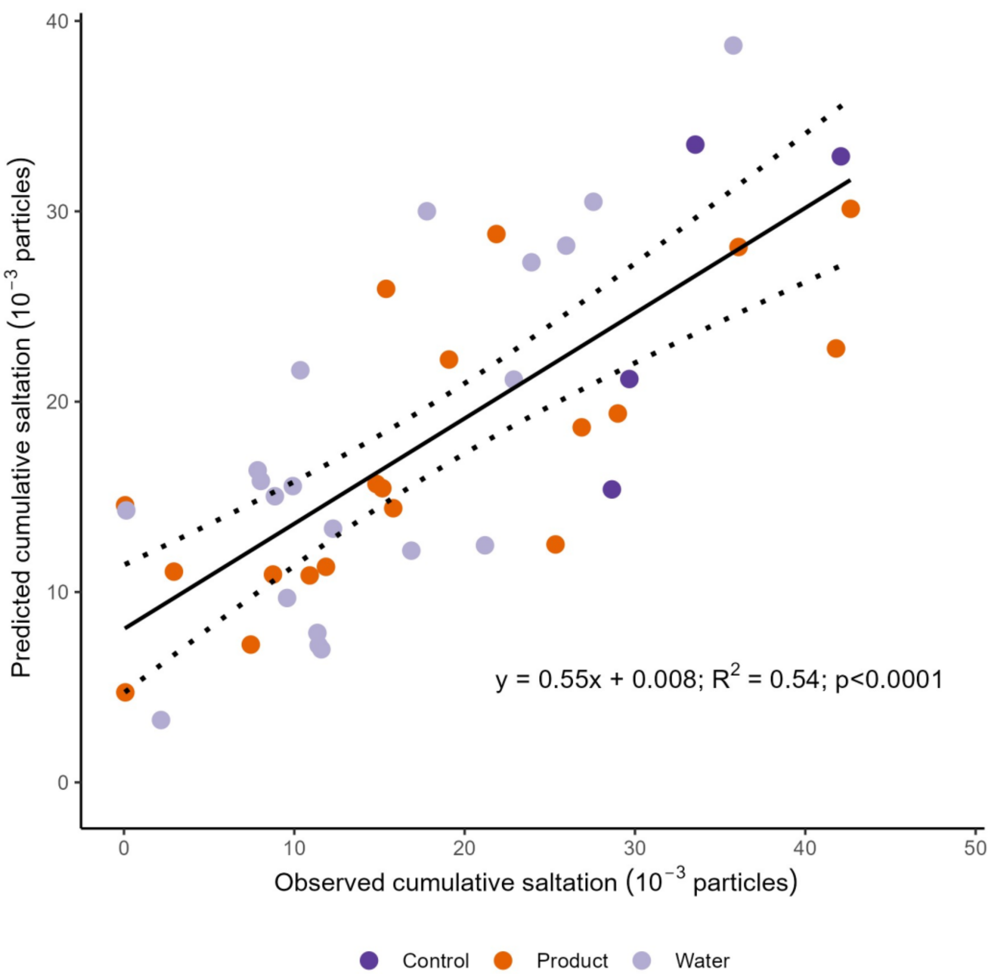

Figure 4.9. Predicted vs. observed values for model describing cumulative saltation count. Solid line shows mean linear regression predictions and dotted lines show 95% confidence intervals.

4.5. Discussion

We observed that the use of anionic PAM as a chemical soil stabiliser at a high application rate (treatment P5; 45 kg ha^{-1}) prevented wind erosion losses from prepared trays of high-SOM peat soil at $u^* \leq 0.82 m\ s^{-1}$ (Figure 4.5; Figure 4.6; Figure 4.7), which is comparable to the higher end of the wind speed range observed at Rosedene Farm by Cumming (2018; 9-12 m s^{-1}). The effectiveness of this application as a wind erosion mitigation measure appeared to result from the formation of a resistant surface crust (Shear strength = 8.24 ± 0.84 kPa; Figure 4.4), which increased the u^*_t of both PM_{10} emissions and saltation above the maximum u^* of 0.82 m s^{-1} applied during our test program (Figure 4.5a, 4.5c). Generally, the factors driving erosion rates from eroding surfaces in our sample were similar to those observed for previous field measurements (WEM/SOM content; Table 4.4; Chapter 3; Zobeck et al., 2013). Given the SOM content and WEM proportion of the surfaces in treatment P5, erosion would be expected to occur, as the density adjusted proportion of WEM was greater than 50% (Estimated at 67 ± 4% using regression from Chapter 3, Appendix 3.1). This suggests that the cohesive properties of anionic PAM and resulting crust were sufficient, at a high application rate, to override soil physical properties predisposing the surface to be vulnerable to wind erosion.

The soil physical properties of our prepared trays broadly resembled field surfaces which had been subject to primary tillage (Table 4.1). Such surfaces represented the lower end of the range of potential erosion rates for high SOM content peat soils in the field (Chapter 3) and our prepared trays were slightly less erodible again (Table 4.3). The ability of the high application rate PAM treatment (P5) to override soil physical properties predisposing erosion vulnerability, suggests that there is potential for erosion mitigation even on more erodible surfaces. However, it is possible that even higher application rates than we tested may be required to achieve this. Other PAM treatments applied at lower rates did not mitigate wind erosion losses beyond the benefits obtainable from the application of the equivalent volume of water alone (P4a/P4b; 1 and 7 kg ha^{-1} respectively; Figure 4.6c; Figure 4.7a). This contrasts noticeably to the results of Genis et al. (2013), who found that PAM application rates as low as 0.5 kg ha^{-1} produced significant erosion mitigation benefits on sandy soils in prepared trays in wind tunnel tests. The disparity in findings suggests that the PAM application rates required for erosion mitigation on high SOM peat soils may be substantially higher than those required for equivalent benefits on mineral soils.

The shear strength estimate for treatment P5 was similar to estimates obtained for rainfall-induced crusts on sandy soils in several other studies (Zimbone et al., 1996; Goosens, 2004; Vos et al., 2020). Sandy soil crusts of similar shear strength were associated with erosion resistance at $u^* \leq 0.59$ m s^{-1} under laboratory conditions (Vos et al., 2020) but were found to be erodible under windstorm conditions in the field (10 min average $u^* = 0.6$ m s^{-1} but max u^* would have been higher; Goosens, 2004). Contrastingly in this study we found no evidence of erodibility for treatment P5 even at high friction velocities ($u^* = 0.82$ m s^{-1}). The likely explanation for this discrepancy in erodibility is that shear vane testing underestimates the strength of PAM-induced crusts. PAM increases cohesive forces between soil particles and forms a network of interconnected molecules within the soil matrix. Insertion of the shear vane into the soil surface will mechanically damage this matrix and disrupt the cohesion of the crust prior to vane rotation, rendering the crust weaker and confounding precise estimation of shear strength. The shear vane measurements in this study were useful to highlight the presence of a relatively strong crust in response to treatment P5. However, caution is advised with the quantitative estimate presented, as the true shear strength of the crust formed following high application rate anionic PAM treatment may well be higher. Penetration resistance may be a superior measure of crust strength for PAM-induced crusts and this has been observed to be substantial; Genis et al. (2013) estimated values between 150 and 300 kPa for applications ranging from 0.5-3.9 kg ha^{-1} on sandy soil, compared to ~130 kPa for water alone, suggesting the potential for PAM to produce very strong surface crusts.

Another possible influence of on the erodibility of crusted peat soils is the presence/absence of loose erodible material on the crust surface (Kohake et al., 2010). Rainfall-induced crusting has been shown to increase the erodibility of agricultural peat soils in wind tunnel tests, with a key driver being the presence of loose erodible material on the crust surface to act as abraders under erosive conditions (Kohake et al., 2010). Armbrust (1999) has previously shown that PAM application (5.6 kg ha^{-1}) on mineral soil can reduce loose erodible material by up to 98%. We did not directly determine the amount of loose erodible material present on crust surfaces in this study. However, the u^*_t of saltation was not reached for any replicate of P5, suggesting either (i) the absence of sand-sized surface microaggregates following high application rate PAM treatment, or (ii) that any such particles were strongly enough attached to the crust that they were not erodible (Figure 4.5c). There was also some evidence that treatment P5 may have reduced the WEM proportion of the surface layer by binding aggregates together but this was inconclusive. In addition to forming a resistant crust, it appears likely that the cohesive action of PAM bound surface particles into the crust and reduced the reservoir of loose erodible material available to act as abraders. However, it should be noted that both PAM-induced crusts on mineral soils (Armbrust, 1999) and rainfall-induced crusts on peat soils (Kohake et al., 2010) show increased vulnerability to erosion in the presence of abraders. Preston et al. (2020) also found that erosion rates from mine tailings treated with several different chemical soil stabilisers for erosion mitigation increased substantially following the introduction of abraders in wind tunnel tests. Consequently, performance under field conditions where abraders may be imported from untreated areas could be substantially poorer than demonstrated in laboratory testing. This makes field testing an essential next step in developing PAM as a potential erosion control measure for agricultural peat soils.

The water application rate associated with treatment P5 (10 mm) is similar to the volume of water used for single irrigation events on salad crops in the study region and would therefore be reasonable in practice. However, the resulting PAM preparation was quite viscous, which would almost certainly present practical challenges for application in the field. In practice, PAM could be applied as a dry powder before being activated by subsequent irrigation. Dry applications were not tested in this study but they have previously been shown to produce erosion control superior to liquid preparations on mineral soils (Armbrust, 1999). The USDA (2020) standards for application of PAM to soil for erosion control under sprinkler irrigation indicate a maximum rate per application event of 4.5 kg ha^{-1}. The application rate at which we observed erosion mitigation on high SOM content peat soils was an order of magnitude higher at 45 kg ha^{-1}. We cannot rule out the possibility that slightly lower application rates may have been effective but an application rate of 7 kg ha^{-1} in this study did not mitigate wind erosion, suggesting that PAM application rates for high-SOM peat soils might exceed current USDA guidelines. For critical areas, USDA (2020) guidelines state that the total application should not exceed 224 kg ha^{-1} yr^{-1}, which would allow five applications at the rate of treatment P5 per year.

This study did not evaluate the persistence over time of erosion resistance provided by the crust formed. However, Kavouras et al. (2009), found that the erosion mitigation effect of tall oil pitch decreased with time under field conditions and a similar effect would be expected with PAM. Several factors might be expected to reduce the performance of the crust for erosion mitigation over time under field conditions, as PAM is subject to mechanical, chemical, photolytic and biological degradation (Xiong et al., 2018). Abrasion by saltating particles imported from other locations may result in dust emission from otherwise erosion resistant crusts, whilst physical disturbance (e.g., by vehicle traffic) has been shown to disrupt surface crusts formed by chemical soil stabilisers and significantly increase erosion rates (Preston et al., 2020). Physical disturbance may be particularly problematic on peat soils due to their high compressibility and low shear strength (Rezanezhad et al., 2016), which may allow deeper layers to deform under load, leading to severe fracturing of surface crusts. These environmental factors may explain why the performance of soil stabilisers under field conditions is far more mixed than in laboratory studies; sometimes offering little advantage over rainfall or irrigation (e.g. Armbrust, 1999; Lyles et al., 1974; Van Pelt and Zobeck, 2004; Preston et al., 2020). This suggests that chemical soil stabilisation may be best suited to short periods of acute erosion vulnerability (e.g. following planting of salad crops). There may also be a role for one-off applications of PAM to stabilise bare peat surfaces during peatland restoration until vegetation cover is reestablished. Repeated applications might be limited by the relatively high price of PAM (Xiong et al., 2018). At an estimated cost of $13 USD per kg (USEPA, 2021), a 45 kg ha^{-1} application would cost just under $600 USD ha^{-1}. This would be likely to impact financial margins and may limit PAM use on agricultural peat soils, to periods when an economic benefit is gained by its application (e.g., by avoiding crop damage/contamination).

There was some evidence that the Hypromellose emulsion (P3) may have potential to mitigate erosion losses from agricultural peat soils but our results were mixed and inconclusive. Hypromellose has previously been found to be highly effective at suppressing fugitive mineral dust emissions from mine tailings (Lee et al., 2020). However, in this study, the performance of product P3 was highly variable between replicates. In two replicates, neither the u^*_t for PM_{10} emissions or saltation were reached (Figure 4.5a, 4.5c). However, in the remaining two replicates u^*_t and erosion rate estimates were comparable to the equivalent volume of water treatment (Figure 4.6; Figure 4.7a; Appendix 4.3). This may imply that the application rate used was in the vicinity of a threshold application rate at which the product becomes effective on high-SOM peat soils. The surface shear strength for this treatment was significantly stronger than controls (Shear strength = 0.83 ± 0.05 kPa; Figure 4.4) but not significantly different from the equivalent volume of water treatment. There was therefore little evidence to suggest that surface crusting could explain the observed erosion resistance. The other soil physical properties measured also provided no explanation for the dichotomous behaviour of the replicates in this treatment. As such, Hypromellose emulsion should remain a candidate option for erosion mitigation on high SOM content agricultural peat soils but requires testing at application rates greater than the rate used in this study (Table 4.2). Tall oil emulsion (P2) resulted in a significantly lower u^*_t than an equivalent volume application of water (W2; Figure 4.5d) and produced higher mean erosion rates than controls (though this was not significant; Figure 4.6b, 4.6c; Figure 4.7a). This result was surprising as tall oil pitch and emulsions derived from it have previously been shown to suppress wind erosion from fine textured alluvium (Kavouras et al., 2009), burned (post-wildfire) soils (Robichaud et al., 2017) and mine tailings (Preston et al., 2020). However, it is unclear from our data why tall oil pitch emulsion performed so poorly on high SOM content peat soils.

It is notable that water applications of 2.7, 3.5 and 10 mm, when applied with relatively low kinetic energy through spray bottles and watering cans significantly increased the surface shear strength of high SOM content peat soils relative to untreated controls (Figure 4.4). Water applications also significantly increased u^*_t (Figure 4.5b) and decreased the cumulative count of saltating particles (Figure 4.7c) relative to untreated controls. Whilst cumulative PM_{10}-C emissions from water treatments were not significantly different from controls, PM_{10}-C fluxes were significantly lower than controls at u^* of 0.55 m s^{-1}. This agrees to some extent with the findings of Campbell et al. (2002), who found that low kinetic energy irrigation avoided erosion from milled peat soils from an extraction site in wind tunnel tests. However, Campbell et al. (2002) observed erosion mitigation from low kinetic energy water application at far higher friction velocities than in this study. It is, therefore, unclear whether irrigation offers a practical erosion control method. Kohake et al. (2010) found that at higher kinetic energy levels, comparable to rainfall, irrigation (simulated rainfall) increased the erodibility of agricultural peat soils. Therefore, it seems likely that low intensity applications of small volumes would be most likely to provide erosion control benefits. Consequently, the period for which increased moisture concentration and weak surface crusting resulting from irrigation persist, may well be too short to provide practical erosion mitigation benefits without very frequent spraying (Bergametti et al., 2016). This may partly explain why Preston et al. (2020) found that under field conditions on mine tailings, water spraying did not reduce erosion rates significantly relative to controls.

Whilst not the focus of this study, applications of chemical soil stabilisers also have the potential to influence soil structure, nutrient cycling, plant health and GHG emissions. PAM is generally associated with positive agronomic effects on mineral soils (Sojka et al., 2007). However, there may be interactions with other management practices (Mulualem et al., 2021) and these effects have not been studied on peat soils to our knowledge. It would also be essential that the use of chemical soil stabilisers did not exacerbate the substantial GHG emissions from agricultural peat soils (Leifeld and Menichetti, 2018). The amount of C added even with multiple treatment applications would represent very small additions relative to the total C stock of agricultural peatlands (Taft et al., 2017) but if addition of labile C or micronutrients removed constraints on microbial respiration then this could have a priming effect, increasing the vulnerability of peat C to mineralisation. PAM application has been found not to increase maize residue decomposition rates relative to controls, which suggests any priming effects may be minor (Awad et al., 2013). However, it is also possible that applications of chemical soil stabilisers could affect soil respiration rates by modifying soil moisture conditions (Fekadu et al., 2024). Finally, there is some evidence that PAM applications can reduce soil nitrogen surpluses and thus nitrous oxide emissions in mineral soils, suggesting there may even be some potential for favourable effects on the GHG balance of agricultural peatlands (Wu et al., 2019). Overall, the wider effects of chemical soil stabilisation on peat are unclear, so agronomic and climate consequences would need to be evaluated before any recommendations for large scale adoption could be considered.

4.6. Conclusions

Laboratory testing identified anionic PAM, applied at a high application rate of 45 kg ha^{-1}, as a commercially available chemical soil stabiliser with the potential to mitigate wind erosion from high SOM content agricultural peat soils. This treatment produced a resistant surface crust which increased the threshold friction velocity of entrainment beyond the maximum tested (0.82 m s^{-1}) and prevented erosion even though the soil still contained a large reservoir of erodible aggregates. There was no evidence of erosion mitigation at lower application rates of PAM, suggesting that stronger applications would be required for erosion control on high-SOM content peat soils than for mineral soils. Field studies would be required as a next step, to evaluate the persistence of erosion control in the presence of imported abraders, environmental stresses and vehicle traffic. The high application rate required means that PAM use on high SOM content peat soils may be expensive. In practice it would likely only be suitable for short periods of acute erosion vulnerability, particularly those where its use provided economic benefits (e.g. minimising crop damage/contamination). It may also have utility for one-off treatments to stabilise bare soil during peatland restoration. Hypromellose emulsion produced mixed results and remains a candidate for further testing. However tall oil pitch emulsion performed poorly on high SOM content peat soils. Water alone did provide some erosion mitigation benefits at low friction velocities ($\leq$0.55 m s^{-1}) but this is unlikely to translate into practical benefits without very frequent spraying. By providing evidence that anionic PAM can stabilise high SOM content agricultural peat soils under laboratory conditions, we hope that this study will provide a first step towards the development of chemical soil stabilisers as a practical erosion mitigation option for lowland agricultural peat soils. Chemical soil stabilisation is unlikely to be a standalone measure but if targeted and implemented correctly, it may represent an important option to assist land managers in protecting valuable peat soil C, whilst supporting production of high-quality food crops.

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Chapter 5

Climate change mitigation potential of relocating vegetable production from organic soil to mineral soil.

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BWJF conducted the investigation and analysis, and wrote the manuscript. DS, DRC, DLJ and CDE reviewed the manuscript.

Abstract

With global food demand expected to rise this century, there is a pressing need for climate change mitigation measures to limit food system greenhouse gas (GHG) emissions. Drained organic soils make an outsized contribution to agricultural GHG emissions given their small area. However, they also make important contributions to the production of certain high value crops (e.g. fresh vegetables in the United Kingdom). The large climate change impact (CCI) of agricultural peatlands makes rewetting and restoration an appealing climate change mitigation strategy. However, this would have implications for food production in many areas and agriculture would likely be relocated rather than discontinued. This could result in supply chains with higher GHG emissions in addition to emissions due to land use change. We undertook an expanded boundary life cycle assessment (LCA) to evaluate the net climate change impact (NCCI) of rewetting agricultural peatlands, using lettuce production for the UK market as a case study. We estimated environmental footprints for the ten major supply chains feeding into the UK lettuce market, considering sixteen environmental impact categories to explore co-benefits or trade-offs alongside NCCI. Using trade/production data we undertook a simple upscaling exercise, to assess the potential NCCI of rewetting policies. The CCIs of organic soil systems were two to four times higher than mineral soil systems and were similar to the CCIs of year-round protected cultivation in artificially heated/lit glasshouses. Cultivated organic soil systems also had higher nutrient footprints than mineral soil systems. We found a strong climate benefit from rewetting organic soil and relocating production to UK mineral soil, with NCCI estimates ranging from -0.71 kg CO_2 eq. kg^{-1} fresh matter (FM) for rewetting thick peat (>1 m depth), to -0.28 kg CO_2 eq. kg FM^{-1} for rewetting wasted peat (≤40 cm depth). We estimate that UK organic soil cultivation accounts for 33% of the CCI of the UK lettuce supply, versus 23% of production. However, economic analysis suggests that abatement costs for rewetting organic soil and relocating high value crop production are high and this strategy is unlikely to be viable at the current carbon price. Policy for responsible management of the UK's lowland agricultural peatlands should therefore focus on relocating low value crop production to mineral soil to allow opportunities for rewetting. On thick peat, the development of compromise strategies (e.g. intermediate WTD management) should be prioritised, to optimally balance production of high value crops against GHG emission reductions.

5.1. Introduction

Greenhouse gas (GHG) emissions from the global food system have been estimated at 18 Gt CO_2 eq. yr^{-1} and account for 34% of all anthropogenic GHG emissions (Crippa et al., 2021). Global food demand is expected to increase by 30-62% by 2050 (from 2010 levels; van Dijk et al., 2021) necessitating increased food production. Whilst there is some evidence for decoupling of agricultural GHG emissions and production (Bennetzen et al., 2016), it is estimated that food production alone could lead to nearly 1°C of warming by 2100 (from 2020 levels; Ivanovich et al., 2023). This clearly presents a major challenge for achieving the 1.5-2°C warming limits laid out in article 2 of the Paris Agreement (UNFCCC, 2015). Increasing food production could be expected to increase GHG emissions through further expansion of agricultural area and additional food value chain activities (Searchinger et al., 2018; Crippa et al., 2021). This could be mitigated by dietary change, waste reduction, climate change mitigation measures and increases in production efficiency (Bennetzen et al., 2016; Springmann et al., 2018; Sun et al., 2022; Zhu et al., 2023). However, behavioural obstacles may constrain the efficacy of the multi-strategy approach required for climate change mitigation (Springmann et al., 2018; Eker et al., 2019; Hegwood et al., 2023). Land use and land use change contributes 32% of food system GHG emissions, predominately through deforestation and the degradation of organic soils (Crippa et al., 2021). Due to their relatively large climate change impact (CCI) the rewetting of cultivated organic soils represents a globally important climate change mitigation strategy (Amelung et al., 2020; Barbier and Burgess, 2021; Huang et al., 2021).

Organic soil found in global peatlands represents a large and important carbon (C) store (>600 Gt C; Yu et al., 2010). However, ~11% of the global peatland area has been drained for productive use; half of this for agriculture (Joosten and Clarke, 2002; Joosten, 2010). Drainage aerates peat soil, stimulating mineralisation of soil organic matter (SOM) and resulting in substantial C losses (Freeman et al., 2022). Consequently, drained organic soils emit an estimated 1.91 Gt CO_2 eq. yr^{-1} (Leifeld and Menichetti, 2018). Drained peatlands account for only 0.5% of the global agricultural area (FAO, 2022, 2023b) and produce just 1.1% of total crop kilocalories (Carlson et al., 2017). However, they are disproportionately vulnerable to C loss and produce 32% of cropland GHG emissions (Carlson et al., 2017). Depleted peatland C stocks are practically irrecoverable; C lost as a consequence of drainage cannot be replenished over human-relevant timescales (Goldstein et al., 2020; Noon et al., 2021). Therefore, conventional agricultural systems on organic soil are effectively both highly extractive (Anderson and Rivera-Ferre, 2021) and unsustainable (economically and environmentally; Wijedasa et al., 2016). To date the only robustly evidenced strategy for reducing GHG emissions from agricultural peatlands is rewetting them by returning the water table to a level close to the peat surface (Wilson et al., 2016; Evans et al., 2021; Huang et al., 2021; Freeman et al., 2022). The effects of rewetting can be variable between sites (due to differences in implementation methods, soil physical properties, site hydrology and nutrient availability) and full ecological restoration is neither immediate nor certain (Kreyling et al., 2021). However, there is evidence that the favourable GHG balance of rewetted peatlands can be resilient even under drought conditions, suggesting that on average, substantial benefits could be achieved with consistent and appropriate management (Koebsch et al., 2020; Beyer et al., 2021; Schwieger et al., 2021, 2022). Due to the longer atmospheric persistence of CO_2 than CH_4, there is a strong argument to rewet drained peatlands as soon as possible, to limit their long-term warming effect (Günther et al., 2020).

However, drained organic soils under agricultural use are also highly productive in the short term. Rochette et al. (2010) estimated that SOM mineralisation in a drained thick peat soil supplied nitrogen (N) at a rate of 250-571 kg N ha^{-1} yr^{-1}, comfortably exceeding the requirements of most crops. Peatlands form under wet conditions, often in topographical depressions, and so with adequate drainage to avoid flooding, can have relatively favourable water availability (Freeman et al., 2022). Peat also has excellent water storage capacity; this declines over time following drainage due to mineralisation and compaction but remains superior to that of mineral soil until SOM loss is so advanced that the soil is no longer classifiable as peat (Liu et al., 2022). Agriculture on drained peatlands therefore benefits from substantial natural assistance in overcoming the key abiotic constraints of water and nutrient availability (Liliane and Charles, 2020). This allows drained organic soil to support highly profitable agricultural enterprises with vegetable production in particular demonstrating extremely high profitability due to the high value of many vegetable crop products (Rebhann et al., 2016). It is important to note that the high profitability of agriculture on drained organic soil does not reflect the long-term and large-scale external costs imposed by its CCI (Pieper et al., 2020). However, particularly where organic soil is used to produce high value crops, financial losses with rewetting could be substantial and must be balanced against beneficial climate effects.

When assessing the benefits of rewetting cultivated organic soil it is essential to account for the relocation of food production elsewhere, particularly for high value crops (Rawlins and Morris, 2010). Optimal relocation of cropland has been posited as an option to reduce the environmental impacts of food production (Beyer et al., 2022) and peatland rewetting may contribute to these benefits. However, replacing lost cropland with newly developed areas (e.g. former grasslands or forest lands) will be associated with land degradation and associated C losses, which must be factored into analyses of the net climate change impact (NCCI) of rewetting agricultural peatlands (Searchinger et al., 2018). Additionally, a shift to alternative supply chains may be associated with increases in GHG emissions associated with energy use, infrastructure and transport (Notarnicola et al., 2017). The debates around the utility of the 'food miles' concept and the viability of reliance upon local production remain unresolved (Coley et al., 2011; Schmitt et al., 2017; Kinnunen et al., 2020; Crippa et al., 2021; Li et al., 2022). However, additional food system emissions are still an essential consideration where substituted production requires either long transport distances (if new production locations are remote from consumers) or additional energy use for protected cultivation (e.g. artificial lighting/heating). Finally, caution must be taken to avoid a narrow view where only CCIs are considered in analysing the environmental effects of rewetting cultivated organic soils, as food production can have a wide range of environmental and socioeconomic costs/benefits. Consequently, there is a risk that any CCI benefits attributable to rewetting may be offset by changes to the costs/benefits of other environmental (e.g. land use, water consumption, fossil resource depletion; eutrophication; Brodt et al., 2013; Clark and Tilman, 2017; Poore and Nemecek, 2018) or socioeconomic impacts (e.g. rural economic activity, local culture; Rawlins and Morris, 2010; Ferré et al., 2019), presenting challenging trade-offs for decision-makers.

Relocation of cultivation following rewetting is especially likely to occur where lost organic soil production accounts for a large share of a highly profitable market because of the highly attractive economic opportunity created. Lettuce (*Lactuca sativa*) cultivation in the United Kingdom (UK) is therefore an excellent case study to assess the potential environmental impacts of rewetting cultivated organic soil. Lettuce is a high value crop accounting for ~15% of the value of UK field horticultural production (2012-2022; Defra, 2022a) and has an estimated net margin of £47k GBP ha^{-1} yr^{-1} on UK thick peat soil (Evans et al., 2023b). Approximately 70% of UK field grown lettuce is produced on drained organic soil (G's Fresh) and lowland agricultural peatlands have become a priority for climate change mitigation policy in the UK (Defra, 2021, 2023a). The UK lettuce market also relies heavily on imports, which account for ~60% of the total supply (2012-2022; Defra, 2022a). Vegetable consumption is associated with relatively high ‘food mile’ GHG emissions due to the short shelf-life of many vegetable products and the requirement for refrigeration during transport and storage (Li et al., 2022). Lettuce therefore represents a context for peatland rewetting in which (i) relocation of cultivation would be highly probable, (ii) a wide range of alternative supply chains already exist and (iii) production emissions from alternative supply chain and cultivation systems have the potential to offset the direct benefits of rewetting. We used life cycle assessment (LCA) for the analysis of this system; LCA is a holistic approach to modelling product systems, which allows comparative analysis of a range of environmental impacts (Rebitzer et al., 2004; Pennington et al., 2004; Notarnicola, et al., 2017). Our aim is to provide an assessment of the potential impacts of rewetting cultivated organic soil, and to support policymakers/food producers to balance socioeconomic and environmental considerations, in developing responsible management strategies for the UK’s agricultural peatlands.

5.2. Methods

5.2.1. Goal, scope and boundary definition

Our first goal was a comparative assessment of the environmental impacts of the supply chains associated with major cultivation systems supplying the UK lettuce market. We therefore conducted an attributional, cradle-to-gate LCA, with a focus on supply to large retailers, and with a functional unit of 1 kg of fresh lettuce delivered to a regional distribution centre (RDC) in the UK. Large retailers supply ~80% of UK fresh produce, so the downstream stages of the lettuce life cycle are highly consistent and were not included in our analysis (Hospido et al., 2009). The system boundary includes all processes relevant to cultivation, packaging and transport of lettuce up to the point of arrival at a RDC (Figure 5.1). We modelled field cultivation of lettuce on thick, thin and wasted organic soil in the UK (Table 5.1). We also modelled field cultivation on mineral soil in the UK, Northern Europe, Southern Europe, and the USA. Finally, we modelled protected cultivation in the UK, Northern Europe and Southern Europe. The open-source software OpenLCA v1.10.3 (GreenDelta, 2020) was used to calculate environmental impacts using the Environmental Footprint v2.0 life cycle impact assessment suite (European Commission, 2018) and Ecoinvent v3.7.1 database (Wernet et al., 2016). Environmental footprints were assessed across all 16 impact categories recommended in the product environmental footprint guidelines (European Commission, 2018), allowing comparison of wider environmental impacts and avoiding a narrow focus on CCI. All identified inputs or emissions sources accounting for ≥1% of product impacts were accounted for in cultivation system and supply chain inventories. Our second goal was to evaluate the NCCI of rewetting cultivated organic soil and relocating this production to mineral soil. To achieve this, we performed a boundary expansion to include changes in GHG emissions resulting from differences between production systems, peatland rewetting and land use change on mineral soil. Our third goal was to contextualise our LCA results by providing an initial assessment of the potential NCCI of large-scale rewetting at the level of the UK lettuce market. We undertook a simple upscaling exercise based on trade and production data for the UK lettuce market, and CCI/NCCI values derived from our analyses.

Table 5.1. Summary of lettuce cultivation and supply chain scenarios. HGV = Heavy goods vehicle, SOM = Organic matter content.

Scenario	O_{THICK_UK}	O_{THIN_UK}	O_{WASTED_UK}	M_{UK}	M_{NE}	M_{SE}	M_{USA}	P_{UK}	P_{NE}	P_{SE}
Region	United Kingdom	United Kingdom	United Kingdom	United Kingdom	Northern Europe	Southern Europe	United States of America	United Kingdom	Northern Europe	Southern Europe
Cultivation	Field	Field	Field	Field	Field	Field	Field	Protected	Protected	Protected
Soil type	Organic	Organic	Organic	Mineral	Mineral	Mineral	Mineral	Mineral	Mineral	Mineral
Peat type	Thick	Thin	Wasted							
Peat thickness	>100 cm	40-100 cm	20-40 cm							
Peat SOM	~65%	~45%	~30%							
Transport	HGV	HGV	HGV	HGV	HGV	HGV	HGV & Plane	HGV	HGV	HGV
Climate	Temperate	Temperate	Temperate	Temperate	Temperate	Semi-arid	Mediterranean/ Arid	Temperate	Temperate	Semi-arid

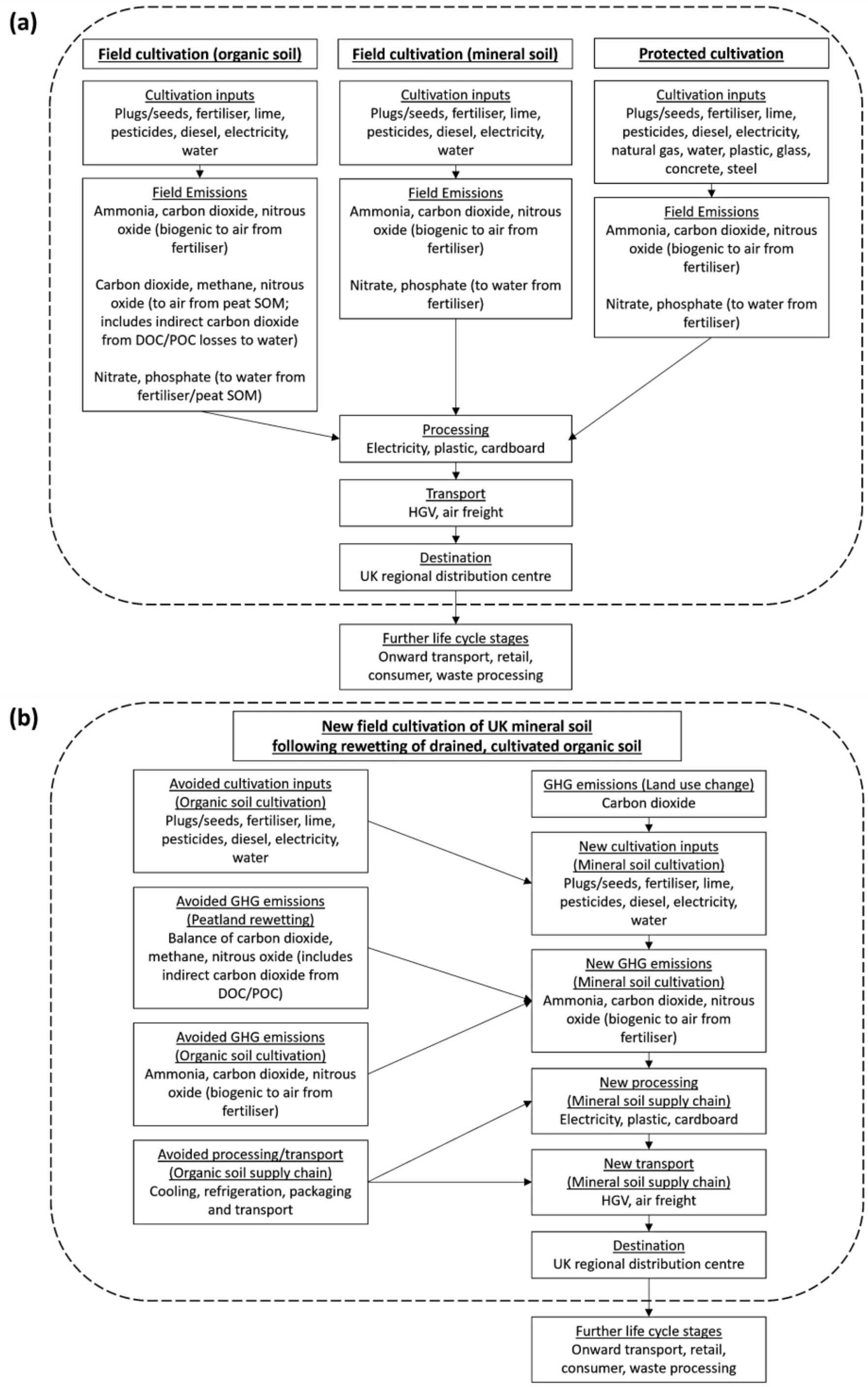

Figure 5.1. System boundaries and main processes for lettuce cultivation and supply chain scenarios. (a) For comparative analysis of environmental footprints between cultivation and supply chain scenarios. (b) For boundary expansion to assess the net climate change impact of rewetting cultivated drained organic soil and relocating production to mineral soil. The dashed line indicates the system boundary; all processes within this are included in our analyses and all processes outwith this are excluded. DOC = dissolved organic carbon, HGV = heavy goods vehicle, SOM = soil organic matter, POC = particulate organic carbon.

5.2.2. Lettuce cultivation and supply chain inventories

Activity data for UK field cultivation of lettuce on both organic and mineral soil were based on primary sources. Activity data for field cultivation of lettuce in Southern Europe were predominately extracted from Milà i Canals et al. (2008). Field cultivation data for the USA focused on Californian systems and were drawn largely from Turini et al. (2011) and Tourte et al. (2017). Protected cultivation of lettuce in the UK was based primarily on data extracted from Milà i Canals et al. (2008), whilst the scenario for protected cultivation in Southern Europe was constructed using data from Torrellas et al. (2012), Romero-Gámez et al. (2014) and Bartzas et al. (2015). Northern European cultivation inputs for both field and protected cultivation were assumed to be the same as those for equivalent UK systems but with larger transport distances. Data were adapted and supplemented as necessary to produce individual inventories representing the cultivation systems described in Table 5.1. Where variety-specific data were available, data describing iceberg lettuce cultivation were used to ensure consistency between inventories. Iceberg is a widely consumed, crisphead lettuce variety which is relatively robust and is well suited to the long transport distances involved in several scenarios (Geisseler and Horwath, 2012). Additionally, generic lettuce input values are often more similar to those for iceberg than other specialist varieties.

UK field cultivation data were obtained from four large farms in England, specialising in the production of salad vegetables. Three of these were situated on organic soil of varying depth and SOM content in the East Anglian Fens. Rosedene Farm (52°32’06’’N 0°27’27’’E) is situated on thick peat (1-2 m in thickness) with high topsoil SOM content (~65%) and was used to model scenario O_{THICK_UK}. Redmere Farm (52°26’39’’N 0°24’33’’E) is located on thin peat, predominantly <1 m in thickness, with intermediate topsoil SOM content (~45%) and was used to model scenario O_{THIN_UK}. Plantation Farm (52°28’11’’N 0°21’48’’E) represents predominantly wasted peat soil (humic clays) <0.4 m thick with low topsoil SOM content (~30%) and was used to model scenario O_{WASTED_UK}. TLC Sussex (50°49’42’’N 0°50’08’’W) operates on silty mineral soil on the South Coast of the UK and was used to model scenario M_{UK}. All farms grow a range of salad vegetables including lettuce. Two lettuce crops are produced on average each year, except for Plantation Farm where heavy humose clay soil limits the growing season to a single crop. Due to their formation processes, large areas of peat in the East Anglian Fens are underlain by clay (Waller and Kirby, 2021) and humose clay accounts for 65% of wasted peat in East Anglia (Seales, 1975). We also modelled a wasted peat scenario producing two crops per year, to account for the remaining 45% of lighter soils ($O_{WASTED\_UK\_2}$). Harvest is manual and lettuce crop residue is left on the field surface as a wind erosion mitigation measure. Fertiliser inputs and spray irrigation water volumes are tailored to crop requirements. In the Fens, irrigation water is pumped from drainage ditches adjacent to fields, using portable diesel pumps, whilst in Sussex water is pumped from a river up to a reservoir with a central electric pump distributing water to fields. The drainage of organic soil requires water to be pumped out of low-lying areas to prevent surface flooding (Freeman et al., 2022) and we used a value of 45.5 kWh ha^{-1} yr^{-1} for this, derived from primary data obtained from the Ely Group of Internal Drainage Boards.

Inventory descriptions for Southern European (M_{SE}) and US field cultivation (M_{USA}) have previously been published in Casey et al. (2022; Appendix 5.1). Briefly, cultivation inputs for Southern European field production were calculated as the average of values for two large farms in the Murcia region of southeast Spain (Milà i Canals et al., 2008). The US field cultivation scenario was constructed to represent large scale intensive production in the state of California, which accounts for approximately 75% of US lettuce production (Geisseler and Horwath, 2016). The scenario presented is a production weighted average of cultivation in both the Central Coast and Southern Desert regions. Monterey County (Central Coast) and Imperial County (Southern Desert) account for 57% and 13% of Californian lettuce production respectively (2007 figures; Geisseler and Horwath, 2016) and thus these two counties are representative of inputs for >50% of US lettuce production.

Protected cultivation in the UK (P_{UK}) was based on data for a farm undertaking year-round glasshouse lettuce cultivation and producing five crops annually, from Milà i Canals et al. (2008). We used an average yield per crop for UK greenhouse lettuce production between 2010-2017 of 32.37 t ha^{-1} (Defra, 2019). Fertiliser N applications were apportioned as ammonium or urea using the proportions in general UK agricultural use for 2002-2005 (FAO, 2023). We based irrigation water pumping energy on scenario M_{UK} and used the greenhouse irrigation energy value for cucumber production from Plappally and Lienhard V (2012). Southern European protected cultivation (P_{SE}) was also assumed to produce five crops annually with a yield per crop of 31.5 t ha^{-1} (Romero-Gámez et al., 2014). We used average crop N, P and K inputs from the greenhouse scenarios in Bartzas et al. (2015). The proportions of ammonium and urea for N inputs were calculated as for M_{SE} (Appendix 5.1). Diesel and pesticide use were assumed to be the same as

P_{UK}. We averaged irrigation water volumes from Bartzas et al. (2015; both protected scenarios) and Romero-Gámez et al. (2014). Irrigation water pumping energy use was assumed to be the same as for M_{SE} (Appendix 5.1) and greenhouse irrigation energy the same as P_{UK}. For both protected scenarios, we used the value of 63.1 m^{-3} ha^{-1} of concrete for the base of the structure (Torrellas et al., 2012). For P_{UK} we assumed the use of glasshouses (Milà i Canals et al., 2008). We used values of 15 kg m^{-2} of steel and 11 kg m^{-2} of glass for these structures (Bartzas et al., 2015). For P_{SE}, cultivation was assumed to take place in plastic covered tunnels, using steel and plastic structural inputs from Torrellas et al. (2012). For both protected scenarios, we used the value of 418 kg ha^{-1} for the weight of plastic irrigation piping, with greenhouse material lifespans from Torellas et al. (2012).

Transplantation of juvenile plug plants is a widespread planting method used in lettuce cultivation across the UK and Europe. Nursery production takes place under protected conditions due to the fragility of seedlings. Primary activity data for the nursery production of plug plants in the UK were collected from a large nursery in the east of England. Infrastructure materials for glasshouses used in UK nursery production were assumed to be the same as for UK protected cultivation of mature crops. Infrastructure (e.g. iron, plastic, concrete) data for nursery production under plastic covered tunnels in Southern Europe were based on the values used for P_{SE}. Electricity use and irrigation water inputs used secondary activity data from Ilari and Duca (2018). Lettuce plug weight (18 g) was estimated using the organic matter per plug (Cumming, 2018), with an estimated SOM content of 80% for the imported peat and moisture content of 1 g g^{-1}. A transportation distance of 40 km from the nursery to the farm was obtained from Bartzas *et al*. (2015) and this was used for all nursery production scenarios.

For all scenarios, initial vacuum cooling of lettuce was estimated to use 0.086 MJ kg^{-1} FM (Plawecki et al., 2014). We used primary data obtained from UK producers for packaging inputs. Transport was assumed to be via HGV except for M_{USA} where air freight is required. We followed the approach of Milà i Canals et al. (2008), in assuming a 200 km distance from UK farms (or point of entry to the UK for US air freighted imports) onward to a UK RDC. We used the distance of 2600 km from Milà i Canals et al. (2008) for Southern Europe and a value of 1300 km for Northern Europe. Vehicle freight within the US was estimated as 128 km for the Central Coast (Salinas Valley, CA to San Jose International Airport, CA) and 200km for the Southern Desert (Imperial Valley, CA to San Diego International Airport, CA) using Google Maps. Air freight distances were estimated as 8626 km (San Jose to London) and 8817 km (San Diego to London) respectively, again using Google Maps.

Field emissions from cultivated organic soil systems include substantial emissions derived from SOM decomposition. Direct CO_2 emissions result from mineralisation of SOM under aerobic, drained conditions and can be predicted using the depth of the drained peat layer (WTDe; Evans et al., 2021). We used the UK regression from Evans et al. (2021) to calculate direct CO_2 emissions for the organic soil cultivation scenarios. Based on existing peat thickness and water table depth measurements (Evans et al., 2016, 2021), we used effective water table depths (WTDe) of 0.3 m for wasted peat (midpoint of 0.2-0.4 m, peat layer fully drained; O_{WASTED_UK}), 0.55 m for thin peat (midpoint of 0.4-0.7 m, peat layer fully drained; O_{THIN_UK}) and 0.85 m for thick peat (midpoint of 0.7-1.0 m; O_{THICK_UK}). For comparison the measured values of WTDe are 0.40 m for Plantation Farm, 0.57 m for Redmere Farm and 0.83 m for Rosedene Farm (Evans et al., 2021). We used measured values for indirect CO_2 emissions from leaching of dissolved organic carbon (DOC) and particulate organic carbon (POC), and evasion of dissolved CO_2 for arable peat in East Anglia from Evans et al. (2016). Direct emissions of CH_4 were calculated using IPCC equation 2.6 from IPCC (2014). We used the Tier 2 emissions factor for CH_4 field emissions from Evans et al. (2017), and the CH_4 ditch emissions and ditch fraction from IPCC (2014). We applied the Tier 1 emission factor for N_2O emissions from cropland on drained organic soil from IPCC (2014).

Calculated field emissions for all scenarios included ammonia volatilisation from synthetic and organic fertilisers using volatilisation factors from IPCC (2019). Direct and indirect N_2O emissions from synthetic and organic fertilisers, and crop residue inputs were calculated using IPCC equations 11.2, 11.9 and 11.10 (IPCC, 2019). We used the disaggregated values for N type and climate where available. Protected cultivation was treated as irrigated dry climate production due to the obstruction of rainfall by glasshouses/tunnels. Leaching losses of nitrate were calculated using the leaching value from IPCC (2019). Phosphate leaching losses were calculated using the 1% loss value from Styles et al. (2015). CO_2 emissions resulting from lime and urea applications were calculated per IPCC (2006). Crop residue N inputs for the above calculations were estimated using the value of 52% of aboveground material remaining in the field after harvest (Taft et al., 2018). Aboveground residue N input was calculated using moisture content (3%), C content (40%) and C:N ratio (7.5:1; Baggs et al., 2000; Cumming, 2018). Belowground residue is assumed to be incorporated for the entire crop. Belowground residue N input was estimated using the ratio of lettuce biomass belowground to aboveground (0.35:1), and the ratio of N content between roots and leaves (0.52:1; Trinchera and Baratella, 2018). The same method was used to estimate residue inputs for all scenarios. For drained organic soil, N and P mineralised from SOM and total peat mineralisation, were estimated using annual CO_2 emissions, the C stock and C:N ratio of the aerated peat layer based on soil profile data from Taft et al. (2017) and the C:P ratio of drained minerotrophic peat under agricultural use from Pakuła et al. (2020).

5.2.3. Water consumption

Water consumption is an important environmental factor which can vary substantially between cultivation and supply chain scenarios. The environmental footprint values for dissipated water (Appendix 5.2) are calculated using global averages from the Ecoinvent database. Unlike GHG emissions where impacts are global and equivalent regardless of the location of the emission source, the impacts of water consumption vary substantially depending on geographic variations in water scarcity (Boulay et al., 2018; Schestak et al., 2022). Regionalisation of water consumption by all inventory processes (e.g. manufacturing, energy production) was beyond the scope of this study. However, as top irrigation water is either incorporated into crop plants or lost to evaporation, blue water consumption by top irrigation could be evaluated (Schestak et al., 2022). We regionalised blue water consumption using annual aggregated characterisation factors (CFs) in line with the Available WAter REmaining (AWARE) method (Boulay et al., 2018; Zampori and Pant, 2019). We used national scale CFs for irrigated agricultural land for general comparisons between cultivation and supply chain scenarios. For Northern Europe, we calculated a weighted average of the CFs for the 18 countries accounting for >99% of lettuce production in the region (2012-2021; EU, 2023). For Southern Europe, we calculated the production weighted average of the CFs for Spain and Italy which together account for 88% of the region’s lettuce production (EU, 2023). We applied watershed specific CFs, in line with the regionalisation of inventory data, to evaluate specific examples of water stress resulting from lettuce cultivation.

5.2.4. Boundary expansion

Rewetting of agricultural peatlands brings the WTD close to the peat surface and thus substantially reduces GHG emissions derived from SOM decomposition (Evans et al., 2021). Emissions of CO_2 and CH_4 from rewetted peatlands were calculated using the UK regressions from Evans et al.

(2021) for a target WTD of 0.03 m, which is optimal for C-derived GHG mitigation. As this represents a best-case scenario, we also undertook a sensitivity analysis to evaluate how WTDs of -0.1 m and 0.1 m, which reflect the likely range in practice, would affect our results. We used measured values for indirect CO_2 emissions from leaching of DOC and POC, and evasion of dissolved CO_2 for wet peatland in East Anglia from Evans et al. (2016). Finally, we used the Tier 1 value for N_2O emissions from rewetted organic soil (IPCC, 2014). UK household purchases of lettuce have remained consistent over the last decade with an average of 32 g person^{-1} week^{-1} (Range = 29-35 g person^{-1} week^{-1}; 2012-2022; Defra, 2023b).

Given consistent demand and the high profitability of lettuce production (Evans et al., 2023b), rewetting of organic soil used for lettuce cultivation would be highly likely to create demand for alternative cultivation areas, to substitute lost supply. Analysis by the UK Committee on Climate Change (2020) found that measures such as sustainable intensification, dietary change, and reducing food waste could free up ~22% of UK agricultural land, implying an excess of grassland in the UK. Consequently, our base case for land use change is that relocation of high value lettuce cropping following rewetting of organic soil, would have the indirect effect of displacing low value (e.g. cereal) cropping from cropland on mineral soil, resulting in conversion of grassland to meet increased demand for cropland (Rhymes et al., 2023). The total agricultural peat area in the UK is ~250,000 ha (Rhymes et al., 2023). This compares to an uncropped (croppable) area of ~300,000 ha and a temporary grassland (< 5 years) area of ~1,250,000 ha on mineral soils in the UK (Defra, 2022c). Therefore, land availability is unlikely to be a constraint on relocation of production from organic to mineral soils in the UK. The described land use change would be expected to result in 24 t ha^{-1} of soil C losses in total for grassland in England (Brown et al., 2023). Soil C losses follow an exponential decay curve, with an estimated half-life of 15 years for this land use change (Brown et al., 2023). This gives an approximate time to equilibrium of one hundred years, over which losses were annualised.

5.2.5. Market analysis

The UK lettuce market supply was estimated using the UK Horticulture Statistics (Defra, 2022a). Total supply has been relatively consistent over the period covered by the dataset (Mean = 313 kt; Quartiles = 285 kt, 321 kt; Period = 1988-2022), with a trend towards increasing imports and decreasing domestic production. In 2022 there was a sharp drop in imports related to supply chain disruptions, which resembles a previous period of volatility in trade associated with the signing of the Maastricht Treaty and the formation of the European Union in 1993. UK overseas trade data (HMRC, 2022) also indicate a reduction in imports beginning in 2020, reflecting supply chain disruptions associated with the COVID-19 pandemic and Brexit. It is unclear whether these represent a period of transient volatility or a trend change in the UK lettuce market. As we are not able to extrapolate market trends, we used mean values from the relatively stable period between 2012-2021 to estimate domestic field/protected production and import quantities for this analysis.

Cultivation of organic soil is estimated to account for ~70% of domestic field production (G's Fresh), with the remainder occurring on mineral soils and modelled as M_{UK}. We estimated the proportion of the total area of cropland on organic soil in England that represents wasted peat using data from Evans et al. (2017) and used this to estimate the proportion of organic soil production on wasted peat, accounting for yield differences. Of this, 65% was modelled as humose clay (O_{WASTED_UK}), with the remainder assumed to be lighter soils producing two lettuce crops annually ($O_{WASTED\_UK\_2}$). We assumed that areas of thin (O_{THIN_UK}) and thick peat (O_{THICK_UK}) followed the same area distribution pattern and estimated their production quantities accordingly. UK protected production was estimated using values from Defra (2022a) and modelled using scenario P_{UK}.

During the period 2012-2021, 99.6% of UK lettuce imports originated in the EU (HMRC, 2022). Of non-EU imports to the UK, >80% originated in North America and therefore all non-EU imports were modelled as M_{USA} for this analysis (HMRC, 2022). During the same period, Southern Europe accounted for ~80% of imports from the EU (HMRC, 2022). Field production accounted for ~90% of production in Europe and this was highly consistent between Northern and Southern Europe (EU, 2023). We used this proportion to estimate the contributions of field and protected production for both Northern and Southern Europe and modelled these as M_{NE}, M_{SE}, P_{NE} and P_{SE} as appropriate. In Northern Europe between 2012-2020, the mean area of cropland on drained organic soil was equal to ~8% of the total cropland area (FAO, 2022, 2023b). Due to a lack of specific data on the distribution of lettuce cultivation by soil type in Northern Europe, we allocated the proportion of Northern European field production to organic soil cultivation based on the cultivated organic soil area. The CCI of organic soil lettuce cultivation for Northern Europe ($O_{AVERAGE\_NE}$) was modelled as the arithmetic mean of UK organic soil cultivation scenarios, with transport emissions adjusted for the longer freight distances involved (in line with M_{NE}). We then used these data to evaluate the potential NCCI of large-scale rewetting of cultivated UK organic soil with relocation of cultivation to domestic mineral soil.

5.2.6. Uncertainty

To provide an estimate of uncertainty for our results, we performed Monte Carlo analyses with 999 iterations using OpenLCA v1.10.3 and R v4.2.2 (R Core Team, 2022). Uncertainty for direct and indirect peatland GHG emissions was derived from empirical data associated with emission estimates where available (IPCC, 2014; Evans et al., 2016, 2021). Uncertainty for other parameters followed a lognormal distribution (Ciroth et al., 2016; Muller et al., 2016). Empirical uncertainty values for trade and market data were not available. Therefore, to constrain our estimates, we assumed an upper error estimate of 20% for production/trade data parameters and 50% for parameters describing the distribution of lettuce production on UK organic soil, which were more dependent on informed assumptions or non-specific data. Values were then sampled from normal distributions specified so that approximately 95% of observations would fall within these error estimates. The distribution of production on organic soil in Northern Europe was sampled from a lognormal distribution to reflect a greater probability of underestimating as opposed to overestimating this value given our approach.

We further evaluated the effects of our modelling assumptions by conducting a sensitivity analysis; varying key input parameters to observe the resulting change in our estimates. Our base case for peatland rewetting is that land managers and policy goals would target an optimal annual average WTD for GHG mitigation of 3 cm. However, we recognise that in practice such precision may not be achieved. We, therefore, evaluated the effect that rewetted peatland WTDs of -10 cm and 10 cm would have on our NCCI estimates (kg CO_2 eq. kg FM^{-1}), as we believe these represent realistic boundaries for a range of target WTDs. Our mineral soil production scenarios (M_{UK} and M_{NE}) used input data from productive silty soil, capable of producing two lettuce crops annually. In the absence of empirical data, our base case used the simplifying assumption that lettuce was produced exclusively on mineral soil capable of producing two crops annually. This is not unreasonable as lettuce is a high value crop and would likely be preferentially allocated to higher quality soils. However, it is clearly plausible that some less productive mineral soils are used for lettuce production and we evaluated the effect of this on NCCI (kg CO_2 eq. kg FM^{-1}). Finally, whilst our base case for NCCI (kg CO_2 eq. kg FM^{-1}) calculation assumes conversion of UK grassland to cropland, we also evaluated reasonable scenarios in which (i) sustainable intensification of production on mineral soil avoids land use change and (ii) land use inefficiencies lead to the clearance of new agricultural land (including clearance of native vegetation in addition to soil carbon stock depletion; Searchinger et al., 2018).

Given the epistemic uncertainty associated with reliance on estimates and assumptions for the distribution of production across UK organic soils, we also conducted a sensitivity analysis on our market scale NCCI estimates (kt CO_2 eq. yr^{-1}). In the base case, peat soil accounts for 70% of UK field production, with the following distribution of production between peat soil thickness categories based on their relative areas: O_{THICK_UK} = 12%, O_{THIN_UK} = 31%, O_{WASTED_UK} = 38%, O_{WASTED_UK2} = 20%. We also considered scenarios, where the percentage of UK field production on peat soil was 50% and 90%, as we believe the true value is highly likely to lie within this range. Our reliance on area data for allocating production between peat thickness categories does not account for the potential for a high value crop like lettuce to be preferentially allocated to more productive soils. Therefore, we also considered scenarios in which production was evenly distributed between peat thickness categories (O_{THICK_UK} = 33%, O_{THIN_UK} = 33%, O_{WASTED_UK} = 22%, O_{WASTED_UK2} = 12%), slightly preferentially allocated to thicker peat (O_{THICK_UK} = 40%, O_{THIN_UK} = 35%, O_{WASTED_UK} = 16%, O_{WASTED_UK2} = 9%) and strongly preferentially allocated to thicker peat (O_{THICK_UK} = 50%, O_{THIN_UK} = 40%, O_{WASTED_UK} = 6.5%, O_{WASTED_UK2} = 3.5%).

5.2.7. Financial analysis

Lettuce is a high value crop and consequently, the financial implications of climate change mitigation strategies are highly salient for both food producers and policymakers. We used income/cost values from a recent cost-effectiveness analysis (Evans et al., 2023b) to examine the financial implications of alternative lettuce production systems. Abatement costs were calculated by dividing the change in net margin between land uses by the associated GHG emission reduction. For rewetted peatlands, we used the costs for wet reedbed management. For relocation of lettuce production from thick peat to UK mineral soil, we adjusted costs for mineral soil production using the ratio of inventory cost values (Table 5.2) and included a variable term to account for potential increases due to the costs (e.g. land purchase) and inefficiencies (e.g. less developed and potentially longer supply chains) associated with relocation. Comparative analysis of flux data from a range of sites suggests that partial reductions in annual average WTD on drained peatlands should also produce partial reductions in GHG emissions whilst permitting continued agricultural use (Evans et al., 2021). We modelled several scenarios for the adoption of intermediate WTDs (30 cm and 50 cm). Production costs on thick peat were left stable, whilst income was adjusted in line with the proportion of the original yield retained under intermediate WTD conditions. Where yield retention was <100%, we also modelled scenarios for relocation of lost production to mineral soil, with relocation cost inefficiencies of 20% and 30%.

5.3. Results

5.3.1. Comparison of environmental impacts

We observed a high degree of variability in environmental footprints between different cultivation and supply chain scenarios (Table 5.2; Appendix 5.2). Scenario M_{USA} had the highest values for nine out of sixteen impact categories, often by substantial margins, reflecting the large environmental impacts associated with long-distance air freight (Table 5.2; Appendix 5.2). Among the remaining scenarios, there was a general pattern for the highest impacts to be from protected cultivation scenarios (Figure 5.2). However, there were a few notable exceptions. High organic fertiliser inputs in the inventory for M_{SE} resulted in relatively high terrestrial eutrophication and acidification impacts (Figure 5.2). Field cultivation on mineral soil in the UK and Northern Europe was generally associated with low environmental impacts. Cultivation of organic soil resulted in notably higher CCI, nutrient footprints and fossil resource depletion that cultivation of mineral soil (Table 5.2). Scenario O_{WASTED_UK} also had a high land use impact, as only one crop was grown per year (Figure 5.2). The relatively larger environmental impacts of cultivated organic soil relative to mineral soil are particularly clear on a per capita normalised basis (Figure 5.3). The larger environmental impacts of cultivating organic soil are strongly driven by mineralisation of SOM (Figure 5.4). It is also notable that production of energy and materials to support year-round protected cultivation in artificially heated and lit glasshouses (P_{UK}) results in relatively large freshwater eutrophication and fossil resource depletion footprints (Figure 5.3; Figure 5.4).

Table 5.2. Summary of environmental impacts and inventory costs per kg of lettuce. Including only impact categories with higher quality input data. Full results for all impact categories are presented in Appendix 5.2. Note $O_{WASTED\_UK\_2}$ models production of two crops per year on UK wasted peat.

Scenario	**Climate change (kg CO2 eq.)**	**Acidification (mmol H^+ eq.)**	**Freshwater eutrophication (mg P eq.)**	**Marine eutrophication (g N eq.)**	**Terrestrial eutrophication (mmol H^+ eq.)**	**Fossil resource depletion (MJ)**	**Land use (Point)**	**Inventory net costs (USD)**
$\mathbf{O_{THICK\_UK}}$	1.02	5	90	4	20	9.3	40	0.70
$\mathbf{O_{THIN\_UK}}$	0.78	4	96	4	19	7.1	39	0.76
$\mathbf{O_{WASTED\_UK}}$	0.88	4	101	4	18	9.5	62	0.80
$\mathbf{O_{WASTED\_UK\_2}}$	0.58	4	97	3	15	6.7	39	0.80
$\mathbf{M_{UK}}$	0.25	3	87	2	12	3.5	40	0.76
$\mathbf{M_{NE}}$	0.35	3	93	3	14	5.0	43	0.89
$\mathbf{M_{SE}}$	0.45	8	105	4	33	6.2	40	1.04
$\mathbf{M_{USA}}$	4.02	22	142	9	89	55.4	33	3.63
$\mathbf{P_{UK}}$	0.89	3	174	2	9	13.3	25	1.95
$\mathbf{P_{NE}}$	0.99	4	181	2	11	14.9	28	2.09
$\mathbf{P_{SE}}$	0.53	3	152	2	12	9.8	31	1.45

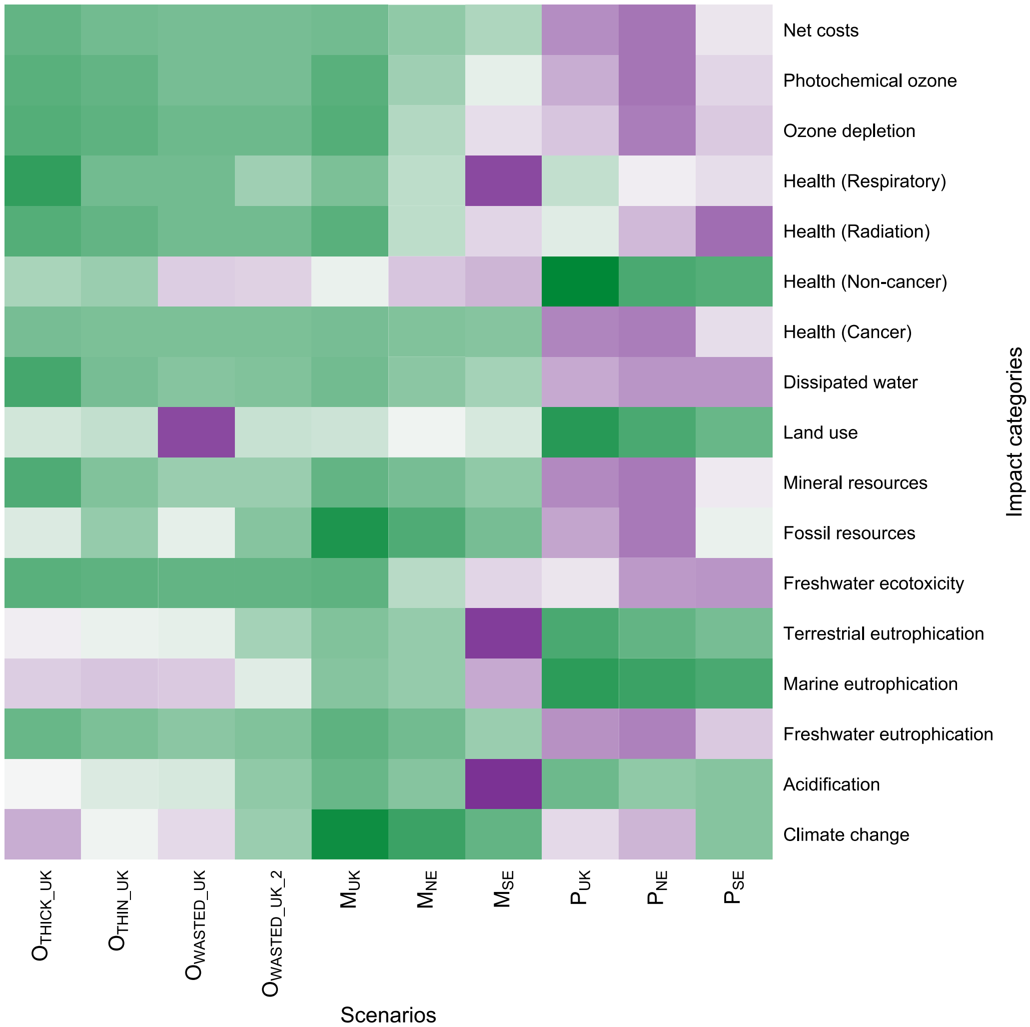

Figure 5.2. Heatmap of environmental footprints. Darker shades of green indicate relatively lower values within rows, pale shades indicate intermediate values and darker shades of purple indicate higher values. Scenario M_{USA} was omitted due to extremely high values for some impact categories which prevented resolution of differences between other scenarios. A version of the figure including M_{USA} is available in Appendix 5.2. Note $O_{WASTED\_UK\_2}$ models production of two crops per year on UK wasted peat.

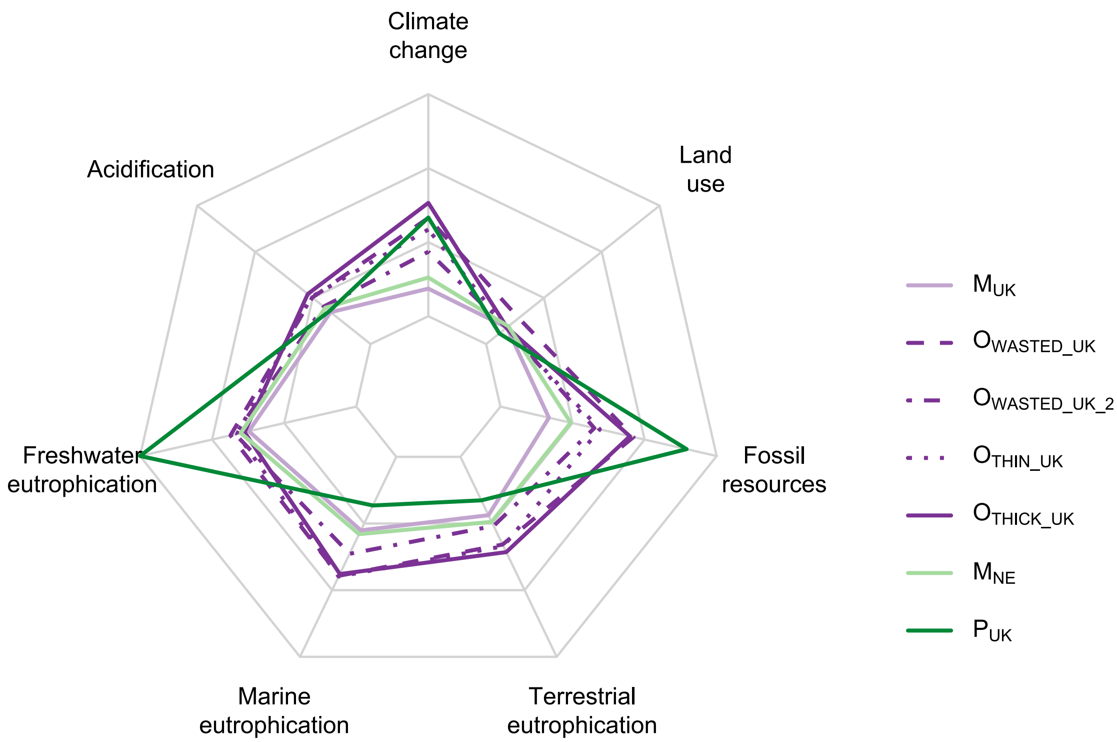

Figure 5.3. Comparison of major environmental impacts between selected scenarios. Data are normalised values using per capita normalisation factors from Sala et al. (2017), presented on a scale of zero to the maximum value in the dataset. Note $O_{WASTED\_UK\_2}$ models production of two crops per year on UK wasted peat.

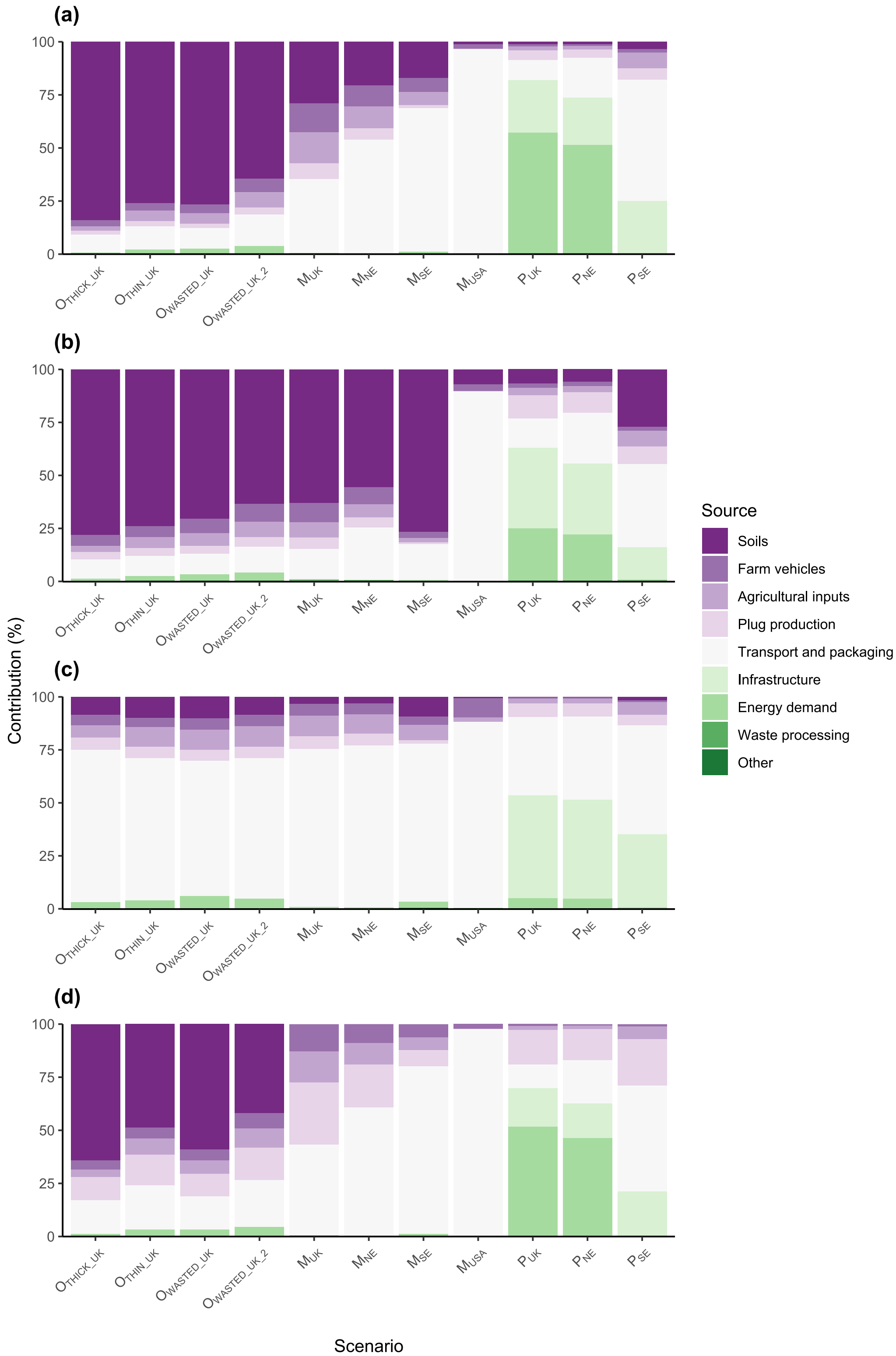

Figure 5.4. Contribution analysis for selected impact categories. (a) Climate change impact. (b) Acidification (Terrestrial and freshwater). (c) Freshwater eutrophication. (d) Fossil resource depletion. Note $O_{WASTED\_UK\_2}$ models production of two crops per year on UK wasted peat.

The CCI of cultivated organic soil production is substantially greater than that of UK mineral soil cultivation and comparable to that of year-round protected cultivation in artificially heated and lit glasshouses (Figure 5.5). The CCI of organic soil systems exceeded that of imported lettuce grown on mineral soil in Europe where produce is transported in HGVs but was substantially lower than that of US imports which are inflated by emissions associated with long-distance air freight (Figure 5.4; Figure 5.5). The CCI of organic soil systems was dominated by SOM-derived GHG emissions (~80%), whilst transport and packaging account for much of the CCIs of mineral soil cultivation systems (e.g. M_{UK} = 35%, M_{SE} = 68%), with a strong dependence of magnitude on the transport distances involved (Figure 5.4; Figure 5.5). For protected cultivation systems, infrastructure, energy production and transport processes tend to dominate environmental footprints (Figure 5.4).

Estimates of blue water consumption through top irrigation were substantially lower for UK and Northern European production systems than for U.S.A and Southern European systems (Figure 5.6). Southern European field production had the highest water consumption at 3.8 m^3 eq. kg FM^{-1}, though this was even higher if watershed specific data for Murcia, Spain were used (5.7 m^3 eq. kg FM^{-1}). In comparison, all UK systems had blue water consumption <0.17 m^3 eq. kg FM^{-1} (Figure 5.6). The lowest blue water consumption was 0.03 m^3 eq. kg FM^{-1} for P_{UK}, though the value of 0.05 m^3 eq. kg FM^{-1} for O_{THICK_UK} was also relatively low among UK systems.

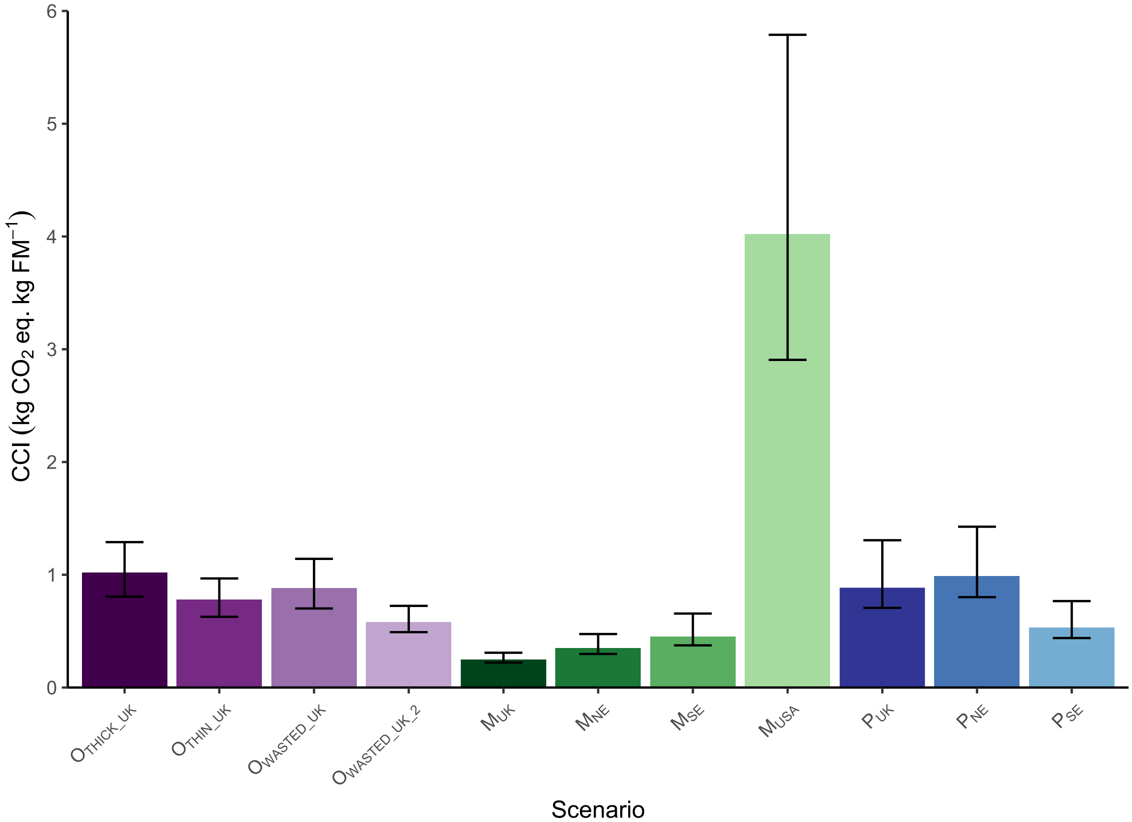

Figure 5.5. Climate change impact of lettuce cultivation and supply chain scenarios. Bars represent deterministic values from attributional life cycle assessment. Error bars represent the 95% simulation intervals of Monte Carlo simulations. Note $O_{WASTED\_UK\_2}$ models production of two crops per year on UK wasted peat.

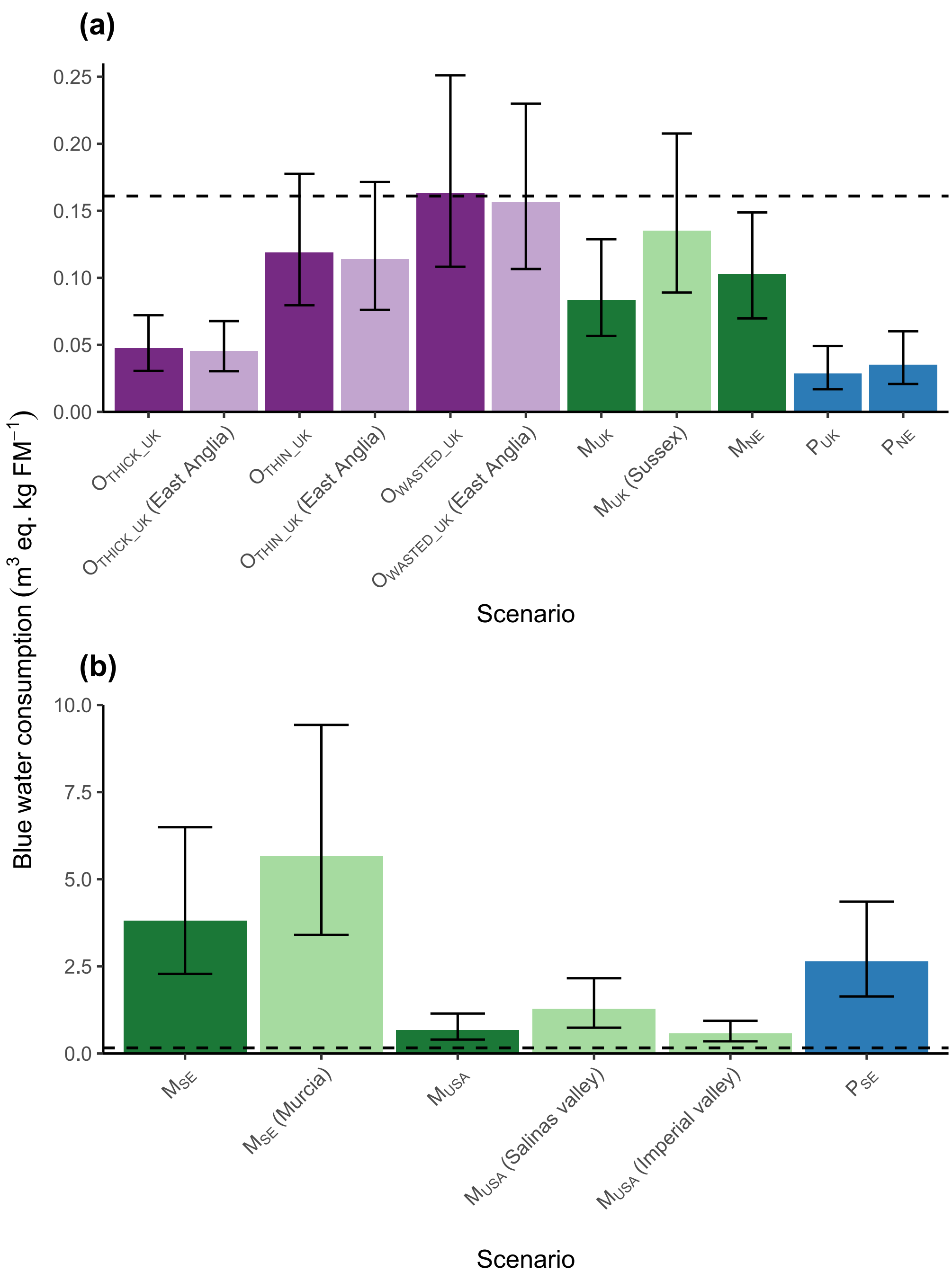

Figure 5.6. Blue water consumption by top irrigation. (a) For scenarios in areas of lower water stress. (b) For scenarios in areas of greater water stress. Bars show inventory values for top irrigation water use, adjusted for geographical variation in water scarcity using AWARE characterisation factors (Boulay et al. 2018). Error bars represent the 95% simulation intervals of Monte Carlo simulations. The dashed horizontal line indicates the non-regionalised global average blue/green water footprint of lettuce production estimated by Mekonnen and Hoekstra (2014). Where watershed specific CFs were used, the localised cultivation region is specified in brackets after the scenario name. Note the difference in scale for the y-axis between panels.

5.3.2. Rewetting organic soil and relocating production

Rewetting cultivated organic soil and relocating production to mineral soil in the UK would be expected to substantially reduce the overall GHG emissions associated with lettuce production for the UK market. Lettuce produced on mineral soil as a substitute following rewetting of peatlands would therefore have a negative NCCI relative to the baseline of continued production on organic soil (Figure 5.7). Estimated NCCI values range from -0.28 kg CO_2 eq. kg FM^{-1}, where lighter wasted peat soils supporting two crops annually are rewetted ($O_{WASTED\_UK\_2}$), to -0.71 kg CO_2 eq. kg FM^{-1}, where thick peat is rewetted (O_{THICK_UK}). The NCCI of rewetting heavy wasted peat supporting only a single crop annually (O_{WASTED_UK}; -0.54 kg CO_2 eq. kg FM^{-1}) was notably larger in magnitude than the NCCI of rewetting thin peat (O_{THIN_UK}; -0.48 kg CO_2 eq. kg FM^{-1}). This reflects a more concentrated allocation of annual peat derived GHG emissions with lower annual yields. The 95% simulation intervals of NCCI estimates for all peat thickness categories did not include zero (Figure 5.7). It is therefore reasonable to expect net climate benefits in most cases from rewetting UK cultivated organic soil and relocating production to UK mineral soil. This also indicates that the climate benefits of rewetting peat outweigh carbon losses from conversion of UK grassland to cropland (even for wasted peat). Our NCCI estimates were relatively robust to modelling assumptions (Table 5.3). However, there was one exception. A WTD of -10 cm on rewetted sites (indicating inundation and standing surface water) was associated with very large reductions in the magnitude of NCCI, producing values for wasted peat scenarios that were indistinguishable from zero (Table 5.3).

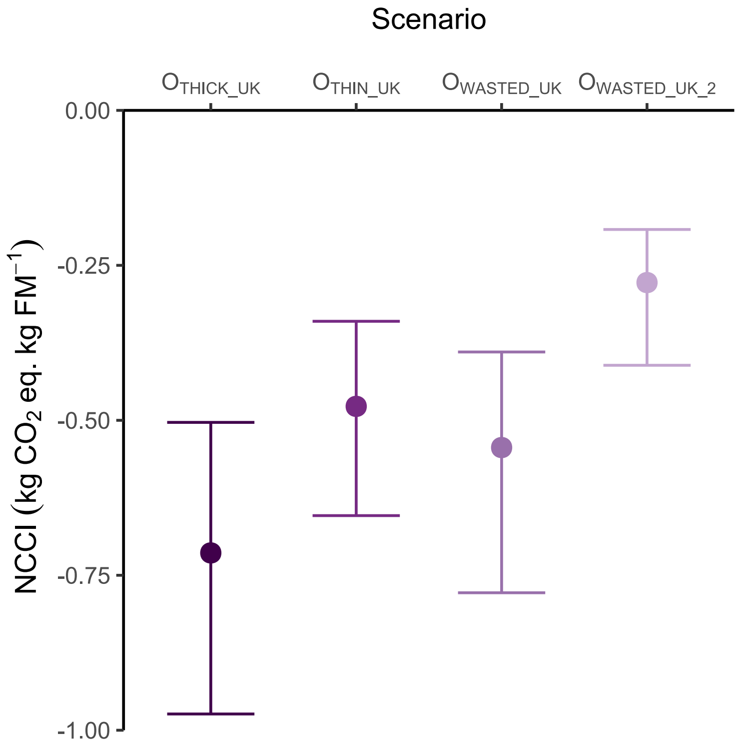

Figure 5.7. Net climate effect of rewetting UK organic soil and relocating cultivation to UK mineral soil. Net climate change impact (NCCI) is the change in average annual emissions over a 100-year period, when new mineral soil production is compared to continued production on organic soil. Results shown per kg fresh matter of lettuce. Points show deterministic values. Error bars represent the 95% simulation intervals of Monte Carlo simulations. Note $O_{WASTED\_UK\_2}$ models production of two crops per year on UK wasted peat.

Table 5.3. Sensitivity analysis for net climate change impact per kg of lettuce. For rewetting of cultivated organic soil and relocation of production to UK mineral soil. Values presented as kg CO_2 eq. kg FM^{-1} of lettuce. Change is presented as a percentage relative to the base case values. Includes the effects of varying the water table depth (WTD) of rewetted peatlands from the base case of 3 cm to values of 10 cm and -10 cm. Also includes the effects on the land use change component of calculations if mineral soil only supports a single crop per year, compared to base case of two crops per year. Finally, we evaluated the alternative land use change scenarios of sustainable intensification (land use change avoided) and clearance of new agricultural land (clearance of native vegetation in addition to soil carbon stock depletion). These were compared to a base case scenario of UK Grassland to Cropland conversion with associated soil carbon losses. Note the scenario $O_{WASTED\_UK\_2}$ models production of two crops per year on UK wasted peat.

Scenario	**O_{THICK_UK}**		**O_{THIN_UK}**		**O_{WASTED_UK}**		**$O_{WASTED\_UK\_2}$**	
	Value	Change	Value	Change	Value	Change	Value	Change
Base case	-0.71		-0.48		-0.54		-0.28	
Rewetted WTD = 10 cm	-0.72	-0.6%	-0.48	-0.8%	-0.55	-1.4%	-0.28	-1.3%
Rewetted WTD = -10 cm	-0.43	+40%	-0.21	+57%	0.00	+99%	-0.01	+97%
M_{UK} = One crop	-0.70	+2%	-0.46	+3%	-0.53	+3%	-0.26	+6%
Sustainable intensification	-0.73	-2%	-0.49	-3%	-0.56	-3%	-0.29	-6%
Land clearance	-0.69	+4%	-0.45	+6%	-0.52	+5%	-0.25	+10%

5.3.3. UK lettuce market analysis

Our analysis suggests that the total CCI of the UK market is 184 kt CO_2 eq. yr^{-1} (95% simulation interval = 149-221 kt CO_2 eq. yr^{-1}). Lettuce produced on UK cultivated organic soil makes an outsized contribution of 60 kt CO_2 eq. yr^{-1} (95% simulation interval = 40-86 kt CO_2 eq. yr^{-1}) to the CCI of the UK lettuce supply, whilst accounting for only 23% of production (Figure 5.8). This is 33% of the total CCI and is comparable in scale to the CCI of imports from Southern European field cultivation on mineral soil (68 kt CO_2 eq. yr^{-1}; 95% simulation interval = 47-97 kt CO_2 eq. yr^{-1}), which alone account for 43% of the total UK lettuce supply (Figure 5.8). Thick peat has the greatest CCI per kg of lettuce among organic soils (Figure 5.5). However, in our base case calculations, wasted peat makes the largest contribution to the overall CCI of cultivated organic soil systems due to its larger area (Figure 5.8). According to our estimates, production on UK organic soil supplies around 2.3 times the quantity of lettuce supplied by UK mineral soil but does so at the cost of producing 7.6 times the GHG emissions.

We used the ratio of percentage contribution to CCI and market supply (CCI:Production) to identify cultivation and supply chain systems which were making overweight/underweight contributions to the total CCI of the UK lettuce supply (Figure 5.9). By far the highest ratio was observed for non-EU imports (M_{USA}; 7.4) but the overall contribution from these systems is low (1.7% of total market CCI) due to the small quantities imported (0.2% of total market supply; Figure 5.8). Field cultivation of mineral soil both domestically and in Europe (North and South) is associated with CCI:Production ratios <1, indicating that these supply chains are relatively advantageous on a climate change basis. In contrast, all cultivated organic soil scenarios have ratios >1 (Figure 5.9), though the value of 1.06 for $O_{WASTED\_UK\_2}$ suggests that the CCI of the most productive wasted peat may approximate the production weighted market average CCI. The CCI:Production profile of protected cultivation in cooler climates is similar to that of field cultivation on organic soil but the quantities of lettuce produced are smaller (Figure 5.8; Figure 5.9). However, the reduced light and heat requirements for protected cultivation in southern Europe result in a notably lower contribution to the market CCI even after accounting for transport emissions.

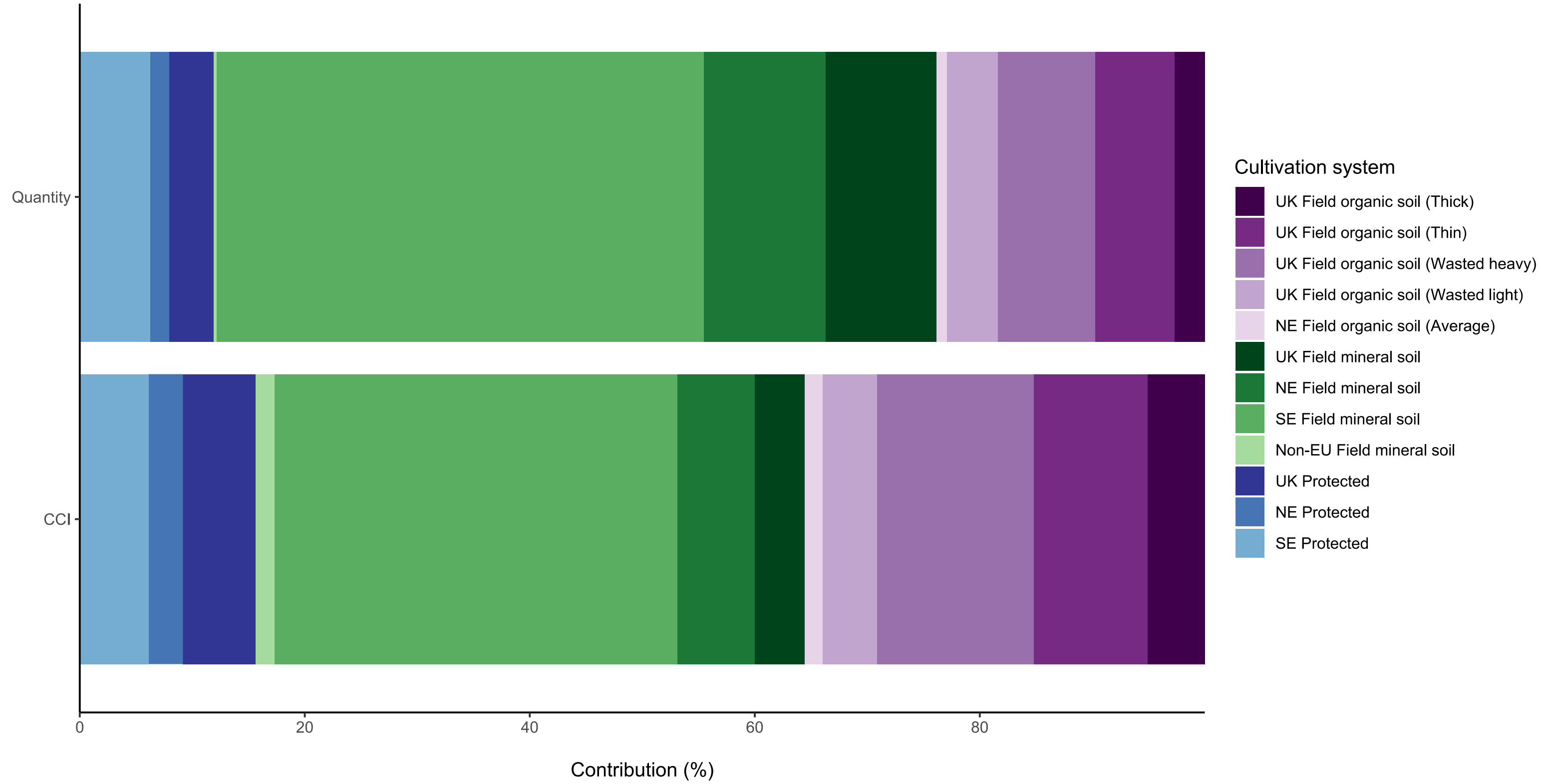

Figure 5.8. UK Lettuce market contribution analysis. Shows the percentage contribution to production/import quantity and climate change impact (CCI) disaggregated by individual cultivation and supply chain systems.

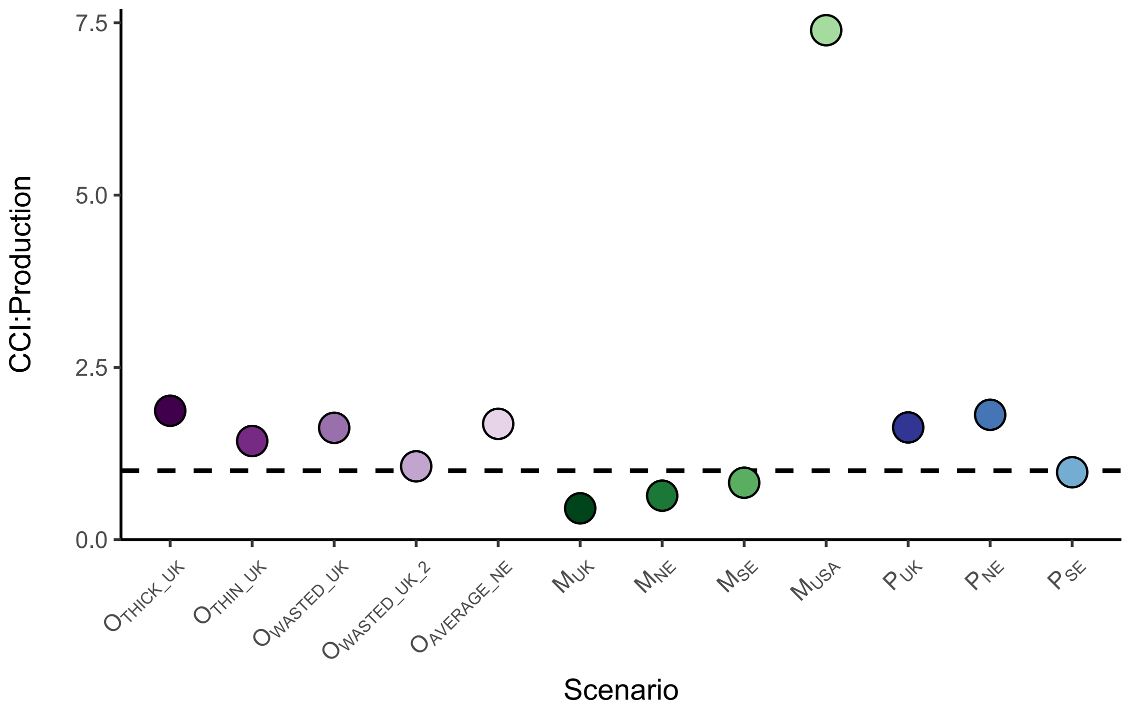

Figure 5.9. Ratio of climate change contribution to production quantity contribution for the UK lettuce market. The dashed line indicates a 1:1 ratio. Values >1 indicate scenarios which make an oversized contributions to the total climate change impact (CCI) of the UK lettuce market. Values <1 indicate scenarios which make an undersized contribution to total CCI. Note the scenario $O_{WASTED\_UK\_2}$ models production of two crops per year on lighter UK wasted peat and scenario $O_{AVERAGE\_NE}$ uses the mean of UK cultivated organic soil scenarios to estimate the average CCI of lettuce production on cultivated organic soil in Northern Europe.

5.3.4. Potential effects of rewetting at UK market scale

The single largest factor affecting the overall NCCI of rewetting organic soil and relocating cultivation to mineral soil would be the scale at which rewetting was implemented; rewetting a greater area of organic soil would be expected to produce a greater climate benefit. Where 100% of peat used for lettuce cultivation is rewetted, with lettuce production relocated to UK mineral soil, the estimated potential NCCI of this land use change would be -36 kt CO_2 eq. yr^{-1} (95% simulation interval = -23 to -55 kt CO_2 eq. yr^{-1}). If rewetting were evenly distributed by thickness category, the benefits could be expected to scale linearly with the rewetted area, such that rewetting 50% of organic soil under lettuce cultivation would produce a NCCI of -18 kt CO_2 eq. yr^{-1} and rewetting 25% would produce a NCCI of only -9 kt CO_2 eq. yr^{-1}. Wasted peat is the thickness category associated with the largest magnitude of potential net climate benefits following rewetting and relocation under our base case where production is assigned based on relative areas (Table 5.4). However, if lettuce production is preferentially allocated to thicker peat, due to the high value of the crop, rewetting thick peat rapidly produces the largest net climate benefits due to the high CCI of cultivating drained thick peat (Table 5.4; Figure 5.5; Figure 5.7). We found that our estimate for the total NCCI of rewetting UK organic soil was relatively robust to changes in modelling assumptions (Table 5.4). However, the NCCI estimates for individual peat thickness categories were highly sensitive (Table 5.4).

Table 5.4. Potential net climate change impact at UK market scale of complete rewetting of cultivated organic soil with relocation to UK mineral soil. Values presented as kt CO_2 eq. yr^{-1}. Main calculation results presented as base case values, alongside a range of sensitivity analyses. Change is presented as a percentage relative to the base case values for sensitivity analysis scenarios. In the base case, peat accounts for 70% of UK field production, with the following distribution of production between peat soil thickness categories based on their relative areas: O_{THICK_UK} = 12%, O_{THIN_UK} = 31%, O_{WASTED_UK} = 38%, O_{WASTED_UK2} = 20%. We also considered scenarios, where the percentage of UK field production on peat was 50% and 90%. We also considered scenarios in which production was evenly distributed between peat thickness categories (O_{THICK_UK} = 33%, O_{THIN_UK} = 33%, O_{WASTED_UK} = 22%, O_{WASTED_UK2} = 12%), slightly preferentially allocated to thicker peat (O_{THICK_UK} = 40%, O_{THIN_UK} = 35%, O_{WASTED_UK} = 16%, O_{WASTED_UK2} = 9%) and strongly preferentially allocated to thicker peat (O_{THICK_UK} = 50%, O_{THIN_UK} = 40%, O_{WASTED_UK} = 6.5%, O_{WASTED_UK2} = 3.5%). Note the scenario $O_{WASTED\_UK\_2}$ models production of two crops per year on lighter UK wasted peat.

Scenario	**O_{THICK_UK}**		**O_{THIN_UK}**		**O_{WASTED_UK}**		**$O_{WASTED\_UK\_2}$**		**Total**	
	Value	Change	Value	Change	Value	Change	Value	Change	Value	Change
Base case	-6.2		-10.8		-14.9		-4.0		-35.9	
Peat soil production = 50%	-4.4	+29%	-7.7	+29%	-10.7	+29%	-2.9	+29%	-25.6	+29%
Peat soil production = 90%	-7.9	-29%	-13.8	-29%	-19.2	-29%	-5.2	-29%	-46.1	-29%
Even distribution on peat	-17.4	-182%	-11.6	-8%	-8.7	+42%	-2.3	+42%	-40.0	-12%
Slight thick preference	-20.9	-239%	-12.2	-14%	-6.5	+57%	-1.8	+57%	-41.3	-15%
Strong thick preference	-26.1	-323%	-14.0	-30%	-2.6	+83%	-0.7	+83%	-43.3	-21%

5.3.5. Financial implications

The estimated abatement cost for cessation of lettuce production and rewetting of thick peat was £912 GBP t CO_2 eq^{-1}. If GHG reductions are converted to £ eq. using the current UK Emissions Trading Scheme carbon price (£83; UK Government, 2022), this would represent a net loss of approximately £43,000 ha^{-1} yr^{-1}. Where lettuce production is relocated to mineral soil, the abatement costs of rewetting thick peat are substantially lower but depend on the scale of the additional costs/inefficiencies associated with relocation (Figure 5.10a). At the current carbon price, relocation costs/inefficiencies would need to be ≤2.4% of the normal costs for production of lettuce on UK mineral soil for carbon credits to adequately compensate farmers. For reasonable costs/inefficiencies in the range of 20-30%, a substantially higher carbon price of £229-312 would be required and this strategy would represent a net loss of £5,500-8,600 ha^{-1} yr^{-1}. Intermediate WTDs would have zero abatement costs if 100% yield retention was possible (Figure 5.10b) and under these circumstances would likely be very attractive to both policymakers and food producers operating on thick peat soil. However, abatement costs would rise substantially under more conservative scenarios involving yield retention of 75-85% at a WTD of 30 cm (£422-703) and 85-95% at a WTD of 50 cm (£217-651) if yield losses were not replaced. With a WTD of 30 cm, relocating production to UK mineral soil to offset a 20% yield loss would produce abatement costs of £310-336 with relocation costs/inefficiencies of 20-30%. With a WTD of 50 cm, relocating production to UK mineral soil to offset a 10% yield loss would produce abatement costs of £233-252 with relocation costs/inefficiencies of 20-30%.

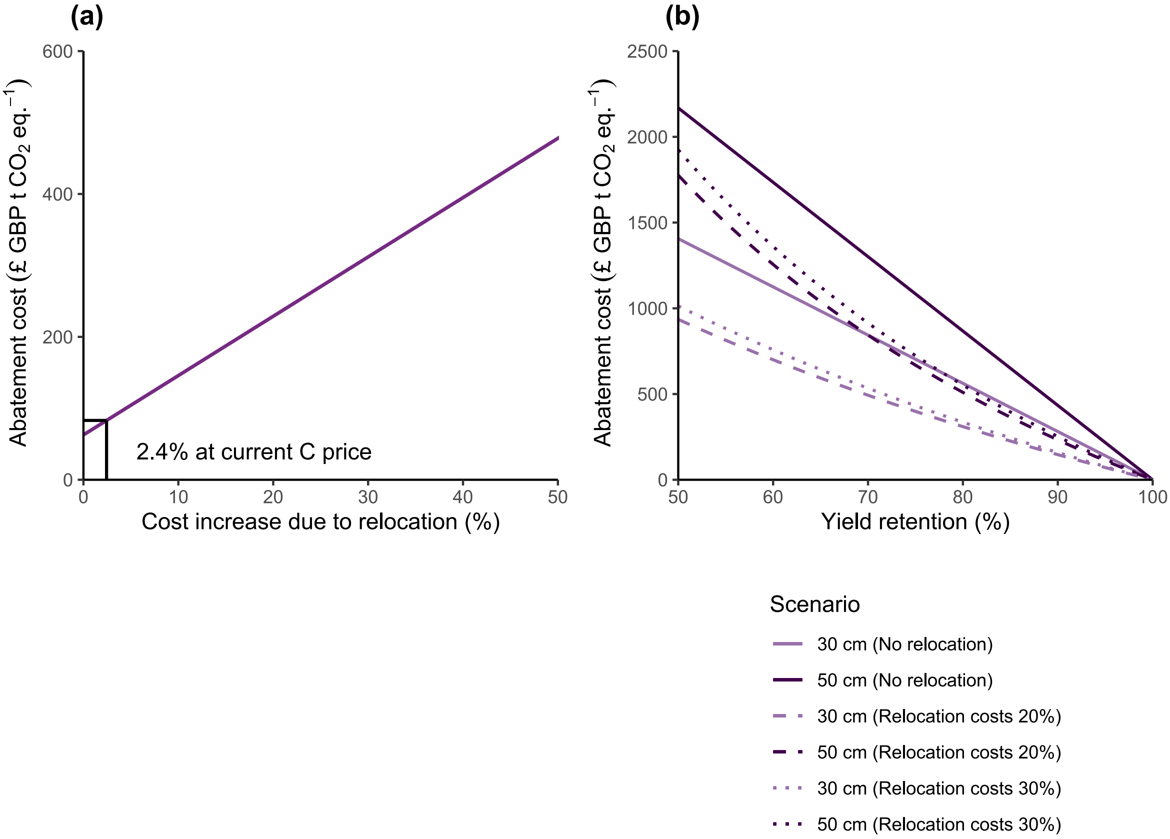

Figure 5.10. Financial implications of climate change mitigation strategies. (a) Abatement costs for rewetting thick peat and relocating lettuce production to UK mineral soil. Shown for a range of values for the additional costs/inefficiencies associated with relocating production to mineral soil. (b) Abatement costs for implementing intermediate WTDs under continued lettuce production on thick peat. Shown for a range of values for the percentage of the yield under fully drained conditions which is retained under intermediate WTDs. Scenarios presented both where yield reductions are not replaced (No relocation) and where yield reductions are replaced by relocating lost production to mineral soil (Relocation). Relocation was modelled as scenarios with additional costs/inefficiencies associated with relocation equal to 20% and 30% of normal mineral soil production costs.

5.4. Discussion

Overall, and on a normalised per capita basis, our analysis suggests that field cultivation of lettuce on UK organic soil has larger environmental impacts than production on UK mineral soil, for most impact categories (Table 5.2; Figure 5.2; Figure 5.3; Appendix 5.2). Specifically, the CCI of lettuce grown on organic soil is two to four times higher than that of lettuce grown on UK mineral soil, due to the substantial GHG emissions resulting from drainage-induced mineralisation of SOM (Figure 5.4; Figure 5.5). Our analysis also suggests that peat mineralisation leads to higher nutrient and fossil resource depletion footprints for lettuce grown on UK organic soil than on UK mineral soil (Figure 5.3). The CCI for lettuce grown on organic soil is similar to that for year-round protected cultivation in artificially heated and lit glasshouses, though other environmental impacts (e.g. freshwater eutrophication, fossil resource depletion) are lower than for protected systems (Figure 5.2; Figure 5.3 Figure 5.5). These findings should incentivise the development and prompt implementation of mitigation strategies to reduce the rate of SOM mineralisation in agricultural peatlands and limit the environmental impacts of food production on organic soil.

Full rewetting of drained organic soil is currently the only mitigation measure with a robust empirical evidence base (Freeman et al., 2022). The results of our boundary expansion analysis suggest that rewetting of cultivated organic soil, with relocation of lettuce production to mineral soil in the UK, would have a strongly beneficial NCCI (Figure 5.7). This indicates that the climate benefits from reduced emissions due to peatland rewetting, would outweigh C losses associated with land use change from grassland to cropland, when production is relocated to mineral soil. The difference in production emissions between lettuce cultivation on organic and mineral soils was relatively small in comparison to peatland SOM-derived GHG emissions. There is currently insufficient evidence available on changes in aquatic nutrient losses following peatland rewetting to allow robust boundary expansion analyses for nutrient footprints. However, nutrient footprints for production on organic soil were higher than for mineral soil (Figure 5.3). Therefore, it appears likely that rewetting of cultivated organic soil and relocation of lettuce production to mineral soil, would offer wider environmental benefits over the longer-term (Table 5.2; Figure 5.7). Where relocation is to UK mineral soil, this should also retain the majority of the economic and food self-sufficiency benefits the UK currently receives from this production.

Our analysis indicates that rewetting thick peat (O_{THICK_UK}) would have the largest climate benefits both on a per ha and per kg FM basis (Figure 5.7). However, lettuce production on domestic thick peat is also the most profitable system for supplying the UK market, as the large supply of nutrients from SOM mineralisation, reduces the need for inorganic nutrient inputs and provides a competitive advantage. The net inventory costs in Table 5.2 do not represent a full cost-analysis and omit several fixed costs (e.g. labour, maintenance, repairs, asset price depreciation and insurance). However, they do account for the differences in nutrient inputs between systems and so provide a reasonable indication of the relative profitability advantage of lettuce production on thick peat. The balance of CCI and wider environmental impacts against profitability, along with their relatively smaller area (Evans et al., 2017), may incentivise policymakers to target rewetting away from thick peat. However, in practice targeted rewetting could prove challenging and rewetting may need to be implemented opportunistically in response to favourable circumstances. The use of bunds and impermeable membranes can reduce lateral water movement but these are rarely fully effective (Freeman et al., 2022). Issues with hydrological management may be exacerbated by the uneven topography of many drained peatland landscapes, as the extent of subsidence will vary spatially, with time since drainage (Dawson et al., 2010). This may lead to areas of inundation or deeper WTDs than intended, constraining GHG emission benefits from rewetting (Table 5.3). Access to the water required for rewetting and the ability for land managers to regulate WTDs on their land may also depend on the actions of other stakeholders and could lead to conflict between land managers with differing priorities (Ferré et al., 2019; Buschmann et al., 2020). Implementation of wetter peatland management in agricultural landscapes will therefore require substantial development before widespread adoption can be recommended.

Given the potential for partial reductions in GHG emissions to be obtained from partial reductions in drainage depth (Evans et al., 2021), policymakers may instead target management for intermediate WTDs with continued agricultural use of thick peat (Freeman et al., 2022). With this approach, C losses and wastage would continue but at a reduced rate, ‘buying time’ and delaying controversial decisions about rewetting until the relative profitability advantage of thick peat is lost. The NCCI where the WTD of thick peat was reduced from 0.85 m to 0.5 m with no effect on lettuce yield, would be approximately -0.33 kg CO_2 eq. kg FM^{-1}. This is ~50% of the NCCI for full rewetting and relocation of production to mineral soil. Given the potential social and economic impacts of fully relocating production, this strategy may be highly appealing. However, the NCCI of this strategy is sensitive to yield changes. Wen et al. (2020) and Matysek et al. (2022) found that raising the WTD from 0.5 m to 0.3 m in peat mesocosms, reduced lettuce yields by 37%

and 32% respectively. In practice yield decreases with WTD change from 0.85 to 0.50 m may be smaller, as there will be less intrusion of groundwater into the root zone at greater WTDs. With a 15% yield reduction, the NCCI of this strategy is -0.21 kg CO_2 eq. kg FM^{-1}, equivalent to only 30% of the climate benefit of full rewetting and relocation. However, if yield reductions were offset by relocation of lost production to UK mineral soil, the NCCI is -0.29 kg CO_2 eq. kg FM^{-1}. This is similar to the NCCI with 100% yield retention, suggesting that the climate benefits of intermediate WTD management maybe relatively resilient to yield losses if these are mitigated by relocation of the lost portion of production to mineral soil. Intermediate WTD management needs thorough field-testing before such systems can be considered as a serious alternative to full rewetting of cultivated thick peat. However, they may provide a useful compromise strategy to balance climate benefits against food production and economic profitability on thick peat.

Paludiculture has also been proposed as an option to maintain some reduced level of productivity on wet peatlands (Wichtmann et al., 2016). There is evidence that paludiculture could reduce peatland GHG emissions relative to conventional drained agricultural use (de Jong et al., 2021; Lahtinen et al., 2022; Thers et al., 2023). However, the NCCI of paludiculture adoption on agricultural peatlands, with relocation of food production to mineral soil remains unclear. This is because: (i) carbon loss via biomass export (particularly for biomass crops such as reeds that form the basis for most current paludiculture trials) may limit GHG emission reductions compared to rewetting and restoration (Günther et al., 2015; Kandel et al., 2017); (ii) food production would be expected to be relocated, leading to land degradation elsewhere (Searchinger et al., 2018); and (iii) paludiculture for bioenergy or construction materials may provide additional climate benefits through avoided fossil fuel consumption or biogenic C storage respectively (Shurpali et al., 2010; de Jong et al., 2021). Development of viable paludiculture systems is a priority for UK policymakers wary of the low economic productivity of restored peatlands (Defra, 2023a). Therefore, detailed evaluations of the expected climate benefits and socioeconomic impacts of paludiculture are urgently required.

In contrast, to thick peat, lettuce production on heavy wasted peat soil (O_{WASTED_UK}; humose clay) does not appear to have a clear profitability advantage when compared to mineral soil (Table 5.2). The environmental impacts (per kg FM) of lettuce production on heavy wasted peat are also relatively high (comparable to thick peat; Table 5.2; Figure 5.3). Heavy wasted peat dries slowly in spring, wets early in autumn, and produces only a single lettuce crop annually.

Therefore, whilst per ha rates of GHG emissions and nutrient losses are lower than for thick peat (due to the lower SOM stock per ha), environmental impacts are concentrated across a lower yield, resulting in comparable values per kg of lettuce produced. The shorter growing season on heavy wasted peat, therefore, makes lettuce production on these soils relatively inefficient in environmental terms. For comparison, when lighter wasted peat capable of supporting two crops annually ($O_{WASTED\_UK\_2}$) is modelled, environmental impacts are lower, and intermediate between mineral soil and thick peat (Table 5.2; Figure 5.3). Consequently, as lettuce production on heavy wasted peat production does not appear to benefit from the competitive advantage held by thicker peat, there is an incentive to reduce lettuce production on these soils. Instead, heavy wasted peat could be preferentially allocated to other high value vegetable crops, which produce only a single harvest annually even on thicker peat, and where yields are comparable to yields on thicker organic soil.

In 2020, the UK's annual GHG emissions due to agriculture were ~45 Mt CO_2 eq. (Defra, 2022b), whilst annual onsite emissions of CO_2 and CH_4 from agriculturally used UK peat soil were approximately 9 Mt CO_2 eq. (Evans et al., 2017, 2023a, 2023c). Lettuce production occupies ~5% of the UK's planted area for vegetables on organic soil and ~1% of the UK's total agricultural peatland area (2012-2022; Defra, 2022a; Rhymes et al., 2023). If the results of our upscaling exercise (Table 5.4) were tentatively assumed to apply to all vegetable crops, then rewetting UK peatlands cropped for vegetables and relocating this production to mineral soil would be expected to produce an emissions reduction equivalent to ~1% of the UK's annual agricultural (and agricultural peatland land use) emissions. However, this substantial potential climate benefit must be balanced against the economic implications of rewetting. The high abatement cost for cessation of lettuce production and rewetting of thick peat (£912 t CO_2 eq^{-1}), reflects the high profitability of lettuce production. This abatement cost is substantially higher than the UK emissions trading scheme carbon price of £83.03 for 2023 (UK Government, 2022). Abatement costs for thinner peat may be higher still as similar financial losses result from smaller GHG emissions reductions. Relocation of production would substantially reduce the abatement costs of rewetting peatlands under high value vegetable cropping but this strategy is still very unlikely to be viable at the current carbon price, once costs/inefficiencies associated with relocation are accounted for (Figure 5.10a). Whilst intermediate WTDs appear to have potential to deliver robust NCCI benefits, our analysis suggests that abatement costs rise steeply if yields are reduced (Figure 5.10b). Therefore, implementation of intermediate WTD systems would need to be optimised before they could be presented as a viable alternative land use for thick peat. It should also be noted that inventory costs based on secondary sources for M_{UK} were higher (\$0.81 USD kg FM^{-1}) than the estimate based on primary sources (Table 5.2). Therefore, this analysis may underestimate the competitive advantage of production on organic soils and may also consequently slightly underestimate abatement costs. Overall, our results agree with previous findings (e.g. Ferré et al., 2019) and suggest that even though the potential climate benefits of rewetting agricultural peatlands are substantial, rewetting peat used for high value cropping is currently unlikely to be financially viable due to high abatement costs and the low carbon price.

In contrast to our results for lettuce, cessation of wheat production and rewetting of wasted peat soil would have a much lower abatement cost of approximately £73 t CO_2 eq^{-1}, based on the values in Evans et al. (2023b). This is equivalent to a small net benefit from rewetting of £185 ha^{-1} yr^{-1} at the current carbon price and is comparable to other estimates of abatement costs for low value agriculture on peatlands (e.g. Krimly et al., 2016). This abatement cost would be expected to be lower still on thicker peat (greater GHG reductions), if lost wheat production were relocated to mineral soil (smaller profit losses) or if other environmental co-benefits were included (e.g. water quality). Consequently, rewetting of organic soil used for low value agriculture may already represent a net benefit in some cases, when balancing profit and NCCI. However, it is unclear whether the current policy and financial environment would allow such benefits to be realised by land managers in practice. The costs of relocation (e.g. land, infrastructure, supply chains and inefficiencies) may be relatively low when amortised over long periods and large areas. However, initial costs would represent a large financial risk and may be insurmountable for smaller producers. Large-scale rewetting would also have acute socioeconomic impacts in areas like the East Anglian Fens, which are heavily dependent on industrial agriculture (Rawlins and Morris, 2010). Responsible management policies for UK lowland peatlands might therefore specifically aim to encourage/support relocation of low value agriculture away from peat to facilitate rewetting. However, there may be a need for government support/funding, in the form of grants/loans/insurance, separate from carbon credits, specifically to support relocation of food value chains and maintenance of sustainable rural communities. Where peat remains drained, responsible management policies might advocate for concentration of high value cropping, so that the costs of GHG emissions are maximally offset by the economic benefits of food production.

Blue water consumption due to top irrigation for UK and Northern European field cultivation scenarios was similar in magnitude to the global average green/blue water footprint for lettuce production of 0.161 m^{-3} kg^{-1} estimated by Mekonnen and Hoekstra (2014; Figure 5.6). Among the UK field cultivation scenarios, O_{THICK_UK} had the lowest top irrigation water consumption (Figure 5.6). However, this likely reflects the use of a sub-surface irrigation system at Rosedene Farm. Annual evapotranspiration has been found to be lower at Rosedene than Redmere Farm (O_{THIN_UK}; Evans et al., 2016), which suggests that water consumption for this system may be lower. However, whilst Rosedene uses only 40% of the top irrigation water that Redmere does, it sees 85% of the annual evapotranspiration. This suggests that total blue water consumption at Rosedene is greater than our estimate due to additional inputs from sub-surface irrigation, which are not captured in top irrigation data. It should also be noted that not all thick peat cultivation will use sub-surface irrigation. Therefore, our blue water consumption estimate for O_{THICK_UK} should be treated with caution. Water consumption was generally lower for protected cultivation than geographically similar field cultivation systems (Figure 5.6), which likely reflects higher water use efficiency and reduced potential evaporation inside glasshouses (Méndez-Cifuentes et al., 2020). Overall, differences in water consumption between UK field cultivation scenarios were small in magnitude given the uncertainty in the data (Figure 5.6). This suggests that relocation of production from UK organic soil to mineral soil is unlikely to substantially alter water demand.

Water consumption values for Southern Europe and the United States were strongly influenced by regional water scarcity (Figure 5.6). This suggests that importing lettuce from water stressed regions imposes significant environmental burdens upon these areas, which they are poorly equipped to handle. The aquifers of the Salinas Valley in California are already experiencing significant saltwater intrusion due to groundwater pumping (Jasechko et al., 2020). The Colorado River, which provides irrigation water to the Imperial Valley has seen a 20% decrease in flow in recent years (Udall and Overpeck, 2017). In Murcia, Southern Spain, there is severe aquifer depletion and eutrophication, which has led to sociopolitical conflict, as agricultural use places heavy demands on scarce water resources against the backdrop of a changing climate (Pedreño et al., 2015; Alonso-Sarría et al., 2016; Cabello and Brugnach, 2023). In the context of the UK lettuce supply, Southern Europe and California largely represent winter supply or emergency supply in times of shortages. Therefore, they would be unlikely to be developed as alternatives to domestic organic soil cultivation following rewetting. Nonetheless, our results highlight the potential for substantial environmental and sociopolitical impacts to occur when produce is imported from environmentally vulnerable or stressed regions (e.g. much of the UK's winter lettuce supply; HMRC, 2022). Such environmental impacts are often poorly reflected by food prices, which then creates a challenging information deficit for consumers looking to make responsible choices (Pieper et al., 2020).

We deliberately did not present 95% confidence intervals of mean estimates in this chapter. Monte Carlo simulation is an infinite-precision method, which can produce misleadingly precise but inaccurate estimates when used on input data containing considerable uncertainty (e.g. LCA inventories; Heijungs, 2020). Therefore, the 95% simulation intervals presented simply show the 2.5th and 97.5th quantiles of the estimates produced by the 999 Monte Carlo simulations. They may best be interpreted as an indicator of the range of simulated values with a reduced sensitivity to extreme values. They reflect both uncertainty due to data quality, estimation error and model assumptions, and variability due to variation in real world circumstances (e.g. distance from RDC or fertiliser use; Huijbregts, 1998). They should not be taken as a probabilistic statement about the location of the true population value as they are unsuitable for this purpose.

The largest contribution to uncertainty in our NCCI estimates per kg lettuce is from peatland CO_2 emissions. This results from the relatively small sample size of the dataset from which the UK regression was estimated (n=16; Evans et al., 2021), which produces relatively large uncertainty in the parameter estimates for the regression. However, these parameter estimates agree well with those from a global model using a larger dataset (n=65; Evans et al., 2021), and more recent UK flux measurements are generally in good agreement with the original dataset (Evans et al., 2023a, 2023b). In addition, this regression represents a between-site analysis of the aggregate relationship between WTDe and CO_2 emissions. Consequently, model residuals account for variation in third variables (e.g. temperature, precipitation, C:N ratio) that might be expected to vary less in the context of within-site WTD changes. Therefore, the 95% simulation intervals for NCCI are relatively conservative and might be expected to narrow with the addition of further empirical flux measurements. Consequently, we can have a relatively high degree of confidence in our main finding: Rewetting organic soil and relocating cultivation to UK mineral soil would on average be expected to have a beneficial NCCI on the UK lettuce market.

In the results of the upscaling exercise, the 95% simulation intervals represent multiplicative effects of uncertainty in both the CCI estimates and the production/trade data. Given the relatively cautious approach used to estimate the maximum potential error in production/trade data, this uncertainty dominates the range of the 95% simulation intervals, which are consequently rather wide. However, our sampling approach for the UK domestic field cultivation inventories relied exclusively on growers operating under the mantle of a single large producer. The remaining inventories relied on secondary data, where we were often constrained to using single data points as parameter estimates. Results for M_{UK} produced from primary inventory data agree well with values produced using secondary data covering three UK farms (Milà i Canals et al, 2008; Casey et al., 2022; Appendix 5.2), which suggests that inventory values are relatively consistent across larger growers. However, clearly our sampling procedure was not statistically representative, and did not include small producers, which results in a degree of epistemic uncertainty, as is often the case in LCA studies (Igos et al., 2019). As such, the cautious approach to uncertainty in the upscaling exercise also reflects an awareness that extrapolating from non-representative samples creates potential for misleading inaccuracy if excessive precision is attempted (Heijungs, 2020). The upscaling exercise still provides useful context for the interpretation of the LCA results. However, whilst the overall estimate of the NCCI of peatland rewetting was relatively robust, the relatively high uncertainty of the resulting market-scale estimates for individual peat thickness categories should be noted (Table 5.4). Nonetheless, this study shows that LCA can provide a powerful tool to evaluate the potential environmental effects of changes in peatland land use. When presented alongside reasonable uncertainty estimates and economic analyses, LCA results can provide valuable support for policymakers and land managers, working to develop responsible management strategies to protect the UK's peatlands, and balance meeting climate targets against the retention of socioeconomic benefits.

5.5. Conclusions

In the case of lettuce production for the UK market, rewetting and restoration of agricultural peatlands, and relocation of production to UK mineral soil could have substantial climate change benefits. Our results also indicate the potential for wider environmental benefits (e.g. reduced acidification, eutrophication, fossil resource depletion footprints). However, our analysis of the financial implications suggests that peatland rewetting has high abatement costs where land is used for high value crop (e.g. lettuce) production. The balance of financial costs and climate benefits would currently represent a substantial net loss when evaluated relative to the UK carbon price. The viability of management for intermediate WTDs with continued high value cropping on thick peat appears to be highly sensitive to yield reductions. However, if yield retention can be optimised, this strategy could have significant value as a compromise between retaining production and reducing GHG emissions. The balance of climate benefits and financial costs would appear to be far more favourable for rewetting of peatlands used for low value cropping. Consequently, responsible management of the UK's agricultural peatlands might be best served by policies with a focus on supporting/encouraging the relocation of low value crop production away from organic soil to allow rewetting. There remain substantial uncertainties about the climate change impacts and financial implications of rewetting for many crops and alternative land use systems (e.g. paludiculture). Addressing these uncertainties and extending the current analysis to a wider range of agricultural systems, is therefore essential to developing robust strategies for the responsible management of agricultural peatlands. LCA has proven to be a powerful tool to evaluate land use change strategies for agricultural peatlands, and can provide valuable decision support for policymakers and land managers as they attempt to balance meeting climate targets with current economic realities.

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Chapter 6

Research synthesis

6.1. Overview

The overarching question that the presented programme of research has sought to address is:

How can the environmental impacts of peatland agriculture best be balanced with food production and economic benefits?

Of course, such a broad and complex question cannot be entirely resolved by the work presented in a single thesis. Nonetheless, the individual chapters presented all contribute to the wider effort by researchers, food producers, policy makers and other stakeholders to address this question. Ultimately, the research pathway that we followed resulted in the consideration of two relatively distinct themes. The first of these (predominately addressed by Chapters 2 and 5) can be summarised as: *Water table management for climate change mitigation.* The second of these (predominately addressed by Chapters 3 and 4) can be summarised as: *Wind erosion vulnerability and mitigation options.* For the sake of clarity, we consider these themes individually in the following sections, contextualising our findings in the wider evidence base and reflecting on the next steps required to progress towards answering the initial overarching research question.

6.2. Water table management for climate change mitigation.

6.2.1. The rationale for wetter peatland management

The substantial climate change impact of agriculture on drained peatlands is well established and is perhaps best demonstrated by global, regional and national greenhouse gas (GHG) emission factors (EFs; IPCC, 2014; Evans et al., 2023a; Chapter 2; Appendix 2.3). In Chapter 2, we observed that carbon dioxide (CO_2) emissions resulting from mineralisation of soil organic matter (SOM) account for ~80% of onsite GHG emissions and (along with the subsidence rate) are strongly controlled by water table depth (WTD; Evans et al., 2019, 2021). Consequently, we concluded that WTD management will be the single most efficacious option for reducing GHG emissions (and wider impacts) resulting from the agricultural use of peatlands.

Full rewetting (returning the WTD close to the peat surface) and restoration of wetland vegetation would be expected to be associated with the greatest and most reliable GHG emissions reductions (Evans et al., 2021; Huang et al., 2021). However, we also noted that a range of responsible agricultural strategies, which adapt to the wetland character of peatlands by reducing drainage intensity under continued productive use may represent important compromises. These would include paludiculture – the productive use of wet peatlands – which is becoming a relatively well-established research field (Wichtmann et al., 2016). Less well researched alternatives might include management for intermediate WTDs or seasonal rewetting, which could be expected to reduce annual GHG emissions by reducing the annual average depth of drained peat (Evans et al., 2021).

Evaluating these options will require future research to provide robust answers to three main questions: (i) What potential benefits could these strategies be expected to deliver? (ii) Do the benefits of these strategies outweigh their costs? (iii) What are the challenges facing implementation of these strategies and how can they be overcome?

6.2.2. Potential benefits

Using UK Tier 2 EFs (Evans et al., 2023a, 2023b) and IPCC AR5 global warming potential values (GWP; Mhyre et al., 2013), the difference in GHG emissions between cropland and rewetted fen peatland, when balancing CO_2, methane (CH_4) and nitrous oxide (N_2O) can be estimated as -20.4 t CO_2 eq. ha^{-1} yr^{-1} for wasted peat and -31.5 t CO_2 eq. ha^{-1} yr^{-1} for thicker peat. Consequently, if all cropland on peat in England were rewetted (based on areas from Evans et al., 2017), this would be expected to equate to a net GHG emission reduction of approximately -4 Mt CO_2 eq. yr^{-1}, equivalent to a 40% reduction in direct GHG emissions from peatlands in England. These are substantial effect sizes and explain why agricultural peatlands have become a policy priority in the UK (Defra, 2021, 2023) in the context of climate targets (UK Government, 2021, 2023).

Given higher mineralisation rates and larger drained areas of tropical peatland, benefits could be expected to be proportionally larger on a global basis (Leifeld and Menichetti, 2018). The potential climate benefit (CO_2 and CH_4 only) from fully and optimally rewetting global peat cropland has been conservatively estimated at -670 Mt CO_2 eq. yr^{-1}, whilst optimally rewetting global agricultural peatlands would be estimated to produce net GHG emission reductions of -800 Mt CO_2 eq. yr^{-1} (Evans et al., 2021). These figures were estimated using 100-year GWPs but in practice, the climate benefits of peatland rewetting also depend on the schedule of rewetting (Günther et al., 2020). The longer atmospheric lifespan of CO_2 relative to CH_4 means that the eventual warming impact of drained peatlands will be greater, the longer they remain drained, which creates some urgency in developing and implementing peatland agri-climate strategies (Günther et al., 2020).

Evans et al. (2021) regressed peatland CO_2 and CH_4 emissions against the depth of drained peat (WTDe) for a range of sites under different land uses and found that for WTDe >0.3 m, every -0.1 m difference in drainage depth was associated with a GHG emission reduction of approximately -3 t CO_2 eq. ha^{-1} yr^{-1}. Extrapolating this relationship globally, they estimated that halving the WTDe of drained agricultural croplands could produce a GHG emission reduction of -410 Mt CO_2 eq. yr^{-1}, which represents ~60% of the GHG benefits of optimal rewetting (the equivalent value for global agricultural peatlands is -510 Mt CO_2 eq. yr^{-1}; Evans et al., 2021). Therefore, the current evidence base suggests that full or partial rewetting of drained agricultural peatlands could have significant emissions reduction.

Whilst the broad conclusions based upon currently available evidence are reasonable, there remain several weaknesses in the evidence base, which require addressing to ensure that the research community continues to deliver the robust information required to support decisionmakers. Firstly, there is substantial unexplained variation in CO_2 emission estimates between published datasets (Tiemeyer et al., 2020; Evans et al., 2021; Koch et al., 2023). The impact of these differences is particularly pronounced at intermediate WTDs and therefore predominantly affects grassland EFs (Evans et al., 2023b). Consequently, estimates of the benefits of rewetting peat cropland are likely to be more robust but these discrepancies need resolving to ensure future management and policy decisions are based on accurate figures.

Secondly, the estimates of Evans et al. (2021) do not consider N_2O or GHG emissions from aquatic sources, which they suggest could account for a further 270 Mt CO_2 eq. yr^{-1} in emissions from agricultural peatlands. Estimates of N_2O emissions from peat cropland in particular are highly variable, which is reflected by the wide confidence intervals for cropland N_2O EFs (see Figure 2.3; IPCC, 2014; Tiemeyer et al., 2020; Evans et al., 2023b). This variation partly reflects the limitations of interpolating fluxes between manual chamber measurements, which can struggle to capture substantial spatiotemporal variation and may lead to underestimation of N_2O emissions for agricultural peatlands (Anthony and Silver, 2021). Therefore, the use of automated chambers and eddy covariance approaches will be important to produce more robust N_2O EFs for agricultural land uses on peatlands (Evans et al., 2023c).

The interactions of soil properties, anthropogenic nitrogen (N) inputs, vegetation and climatic influences, and hydrology driving N_2O emissions also appear to be substantially more complex than the drivers of CO_2 and CH_4, impeding the development of simple Tier 3 approaches based on WTD alone (Couwenberg et al., 2011; Leppelt et al., 2014; Tiemeyer et al., 2016). N_2O emissions from peatlands with WTDs near the surface are generally low (Couwenberg et al., 2011). However, there remains a need to refine our understanding of the drivers of N_2O emissions to ensure WTD changes and fluctuations associated with reductions in drainage intensity do not exacerbate N_2O emissions and negate C-derived GHG emission reductions (Tiemeyer et al., 2016; Anthony and Silver, 2021).

Thirdly, as discussed briefly in Chapter 5, the CO_2 emissions regressions presented by Evans et al. (2021) represents a between-site analysis of the aggregate relationship between WTDe and CO_2 emissions. Linear regression of CO_2 emissions against a single independent variable (WTDe) cannot fully account for differences in vegetation influences between land uses. Therefore, given the very low losses of vegetation C from sites with near-surface WTDs, the analysis could overestimate the benefits of partial within-site reductions in WTDe for productive land uses. This could have important implications for the viability of intermediate WTD management which currently appears promising as a responsible management strategy for high-value agriculture on peat if benefits are calculated using the Evans et al. (2021) regressions (Chapter 5). Initial results from within-site, between-field trials (Evans et al., 2023c), appear to suggest that vegetation influences may dominate the net CO_2 emissions responses to small WTD changes under productive land use. There is therefore an urgent need for controlled experimental studies assessing the within-field effects of intermediate WTD management to ensure the hypothesised benefits can be realised in practice (Evans et al., 2023c).

Chapter 5 used literature values for peatland GHG emissions as part of a wider life cycle assessment of the production of lettuce for the UK market. We used the UK regression from Evans et al. (2021) to estimate CO_2 emissions, which is derived from UK eddy covariance measurements of CO_2 emissions and can therefore be considered representative for the scenarios modelled. We used a combination of the IPCC (2014) Tier 1 EF for N_2O emissions from drained organic soils and calculated N_2O emissions contributions associated with N inputs, allowing at least partial resolution of between-farm differences; measurements by Taft et al. (2017) suggest that peat-derived N_2O emissions are similar across the three peatland farms. Finally, we focused the main analysis on full rewetting to non-productive, restoration land use for peatland scenarios. This limited the influence on our main analysis of issues around overestimation of the benefits of partial drainage depth reductions. Instead, we included simple estimates of the potential effects of intermediate WTDs in our discussion to contextualise our main results and highlight the pressing need for practical empirical evaluation of these systems.

Overall, our findings in Chapter 5 suggested that even after accounting for changes in production emissions and indirect land use change consequences, rewetting organic soil under lettuce production and relocating production to mineral soil would produce substantial net climate benefits. Our findings also suggest that due to lower SOM mineralisation rates per ha in mineral soils, rewetting peatlands could also have wider environmental benefits in the longer-term by reducing the eutrophication, acidification and N deposition impacts of lettuce production. Agricultural peatlands are associated with substantial aquatic nutrient losses (Grenon et al., 2023). However, the exact effects of peatland rewetting on nutrient flows are not currently empirically well established and there is evidence that rewetting can at least initially produce increased rates of aquatic nutrient losses in response to hydrological changes (Pönisch et al., 2023). Therefore, at this stage caution should be exercised in claiming wider benefits for peatland rewetting until the duration and magnitude of these effects are understood better.

6.2.3. Costs vs. benefits

The results of our life cycle assessment in Chapter 5, suggest that the primary benefits obtained from agriculture on drained peatlands are socioeconomic, with peatland rewetting and mineral soil cultivation offering relative environmental benefits in comparison. The main direct benefits provided by the agricultural use of peatlands are food production and economic activity resulting from food value chains. Indirectly, agribusiness also provides wider social/cultural benefits by supporting rural communities. These are non-negligible benefits, which must not be taken lightly in discussions of peatland rewetting, as their loss would represent substantial costs.

Rural abandonment and loss of industry can create pronounced socioeconomic stresses in agriculture/resource/industry dependent communities (Hobor et al., 2012; Beatty and Fothergill, 2020; Quintas-Soriano et al., 2023). In the case of the East Anglian Fens, exploitation of peat resources supports agriculture which contributes ~7% of total agricultural production and 33% of fresh vegetable production in England (NFU, 2019). From farm to fork, the associated food chain employs ~80,000 people (NFU, 2019). The socioeconomic effects of widespread rewetting in the region could therefore be substantial, and it is essential that the development of responsible management policies accounts for this.

In comparison to research on peatland GHG emissions, academic understanding of the economics of agricultural peatland systems and the potential costs associated with proposed responsible agriculture systems is relatively limited. This is understandable, as measurement and elucidation of potential issues inevitably precedes and incentivises the development of solutions. However, it creates a situation where currently, scientific publications tend to centre on peatland GHG emissions, whilst food producers' main concerns centre on the challenging economic realities of agricultural production. The resulting divergence of attention, priorities and values risks stakeholder division over the most appropriate strategies for future management of agricultural peatlands (Rawlins and Morris, 2010; Taft, 2014; Reed et al., 2020).

To maximise stakeholder cohesion, we concluded in Chapter 2, that development of solutions for agricultural peatland management should be based on participatory research, and the co-creation of knowledge and workable solutions. Undertaking detailed economic analyses of alternative land use strategies for peatlands and developing financial valuation approaches for environmental benefits will be essential to this process. Financial valuation of ecosystem services has many limitations (Small et al., 2017) but could still have substantial utility in this context, by allowing researchers, food producers and policy makers to compare the costs and benefits of strategies directly, in the same terms. The simple economic analysis presented in Chapter 5 is limited and far from sufficient to resolve this issue. However, it provides a useful starting point and was necessary to ensure potential climate benefits from rewetting were presented in the context of potential socioeconomic costs.

Our results from Chapter 5 suggest that currently, rewetting agricultural peatlands under high-value crops would very likely result in financial losses for food producers that exceed payments for GHG emissions reductions, even if production was relocated to mineral soil. Full rewetting of high-value agricultural systems is therefore unlikely to be adopted in the near-term. Intermediate WTDs under continued high-value production might be financially viable if yield losses are minimal but these systems require field testing to ensure high yield retention and proposed climate benefits are obtainable in practice. Therefore, research should focus on evaluating and developing, functional, resilient and cost-effective strategies to reduce GHG emissions from continued high-value cropping through reductions in drainage intensity.

The above conclusion is of course, heavily dependent on the current low carbon price and the limitations of carbon markets (Ervine, 2018). It is therefore possible that future increases in the carbon price might shift the balance to favour peatland rewetting even under higher-value agricultural use (de Jong et al., 2021). This further incentivises development of reduced drainage intensity strategies to slow rates of peat loss now, to reduce emissions, retain optionality and maximise the remaining C stock that might potentially be rewetted in future.

Notably, we also found some evidence based on figures for wheat production from Evans et al. (2023c) that relocation of low-value agricultural production to mineral soils, to permit rewetting of peatlands, may be financially viable, even at the current low carbon price. This agrees with other research suggesting that abatement costs for low-value agriculture on peat are competitively priced relative to other GHG mitigation strategies (Krimly et al., 2016). However, this suggestion is based on very limited data and should be treated with caution until more comprehensive and detailed analyses are available.

If future studies confirm that continued low-value agriculture is not net-beneficial when compared to peatland rewetting, then policy for responsible management of peatlands might benefit from a focus on relocating low-value agriculture away from peatlands. This could have the effect of increasing profit per ha and thus profit per t CO_2 eq. of GHG emissions on the remaining drained peat area. The aim would be to optimise the cost/benefit ratio associated with peat resource depletion and associated GHG emissions from peatland areas under agricultural use.

There is a strong and urgent need for detailed economic analyses to strengthen the evidence base and inform decision making with regard to agricultural peatland land use and climate change mitigation. The economic implications of peatland rewetting are likely to depend upon the carbon price, the value of current agricultural production and local circumstances. Therefore, fluctuations over time can be expected and a one-size-fits-all approach is unlikely to be adequate. Agri-climate policy for peatlands will therefore need to be flexible and adaptable to ensure just and optimal outcomes.

6.2.4. Implementation challenges

As discussed in Chapter 2, there are likely to be substantial drainage and water resource management challenges associated with wetter management of agricultural peatlands. The experimental work conducted in this thesis did not examine this topic directly. However, such challenges represent essential context for evaluating the potential benefits of the climate change mitigation strategies considered in Chapter 5. Therefore, we provide a brief discussion here with a view to highlighting research priorities.

Hydrological challenges are inevitable with management for wetter peatlands due to spatiotemporal variability in water supply. Specifically, management will be required to mitigate flood risk when supply is excessive and mitigate the risk of water shortages when supply is limiting. These challenges will need to be addressed at field, farm and landscape scales. There will be a need both to mitigate impacts from extremes of precipitation, and ensure efficient and equitable distribution of water within management units. Successful implementation will therefore likely require both technological and social innovations.

The first major research/management challenge will be achieving the necessary accuracy in management for target WTDs and ensuring that variation around annual average WTDs can be limited to acceptable levels. Substantial intra-annual WTD variability has been observed on both drained and wet peatland sites in the UK (Evans et al., 2016). Failure to control WTDs could lead to waterlogging of crop root zones and poor trafficability under intermediate WTD systems with potentially large economic consequences (Madramootoo and Abbasi, 2022). Chapter 5 also highlighted the potential for substantially reduced net climate benefits from rewetting if poor WTD control leads to inundation of rewetted sites, given the exponential relationship between CH_4 emissions and WTD (Tiemeyer et al., 2020; Evans et al., 2021).

The solutions to within-field WTD control will likely differ between land uses. On rewetted sites, in widely drained landscapes, management will likely need to prioritise water retention through the construction of bunds and the use of impermeable membranes – though neither strategy is foolproof (Evans et al., 2016). For agricultural land uses, there is growing research into using subsurface drainage systems to improve WTD control (Couwenberg, 2018; Weideveld et al., 2021; Boonman et al., 2022; Madramootoo and Abbasi, 2022; Offermanns et al., 2023). There is some evidence for improved WTD control (Weideveld et al., 2021; Boonman et al., 2022) but WTD will still vary spatially (Offermanns et al., 2023) and the longevity of these benefits remains uncertain (Couwenberg, 2018). Continued research to develop field/site scale WTD control solutions will therefore be essential if the WTD recommendations made for GHG mitigation are to be viable as practical management options for peatland systems.

Water management challenges will also need to be overcome at the landscape scale, where the focus is on flood management, and the storage and redistribution of water resources. Retention of a larger volume of water within peatland systems will inevitably reduce the available soil storage available to mitigate flood risk during high rainfall periods. This may necessitate increases in pumping capacity or the creation of washlands for short-term surface storage (Mulholland et al., 2021). It is also possible that increased water demands associated with maintaining shallower WTDs will create water supply challenges during drier summer periods, which may increase the requirement for reservoirs within water management systems (Querner et al., 2012).

However, rewetting peatlands would also increase the total volume of water stored within peatland areas (Stachowicz, et al., 2022), possibly allowing some flexibility for redistribution to mitigate the economic impacts of dry periods on agricultural production. This suggests that wetter peatland management would present both challenges and opportunities, with the net-benefits depending on the strategies selected. Initially, there is a strong need to improve our understanding of hydrological stocks and flows in agricultural peatland landscapes to identify where these challenges and opportunities lay for individual systems.

The solutions to the hydrological challenges of wetter peatland management still remain largely unclear and any changes would have the potential to cause stakeholder conflict at least initially (Ferré et al., 2019; Buschmann et al., 2020). The uneven topography and complex water management systems present in many agricultural peatland areas will likely require equally complex and locally adapted solutions. Nonetheless, the sophisticated, resilient and effective water management systems found in lowland peat landscapes such as the East Anglian Fens and the Netherlands indicate substantial capacity for engineering advancements to meet such challenges, when the benefits justify the investment.

6.3. Wind erosion vulnerability and mitigation options

6.3.1. Erosion rates

Wind erosion losses from agricultural peatlands remain poorly quantified (Chapter 2; Freeman et al., 2022). Recent field monitoring studies have estimated horizontal mass transport rates above agricultural peat soils in East Anglia (Cumming, 2018; Newman, 2022). However, in the absence of information about net detachment/deposition rates, horizontal mass transport estimates cannot reliably be used to estimate C loss rates per unit area for comparison with GHG EFs. Warburton (2003) measured transport distances of marked particles, allowing horizontal mass transport data to be converted to an approximate soil loss estimate of 0.46-0.48 t ha^{-1} yr^{-1} for an isolated patch of bare upland peat in the UK. As far as we are aware, this is the only published estimate of the wind erosion rate for peat in the UK.

The estimate of Warburton (2003) is similar to the average net erosion rate estimated by caesium-137 tracing for mineral soils under agriculture in East Anglia by Chappell and Warren (2003; 0.6 t ha^{-1} yr^{-1}). However, it is lower than caseium-137 tracing estimates for dryland agricultural soils in Australia (Chappell and Baldock, 2016; 4.4 t ha^{-1} yr^{-1}) and the USA (Van Pelt et al., 2017; 3.7-6.6 t ha^{-1} yr^{-1}). Given these admittedly limited data, it seems likely that soil loss rates due to wind erosion from agricultural peat may be comparable to agriculturally managed mineral soils in similar climates. However, aeolian C losses from peat may be proportionally higher than from mineral soils due to their higher C content and the C enrichment of eroded material (Kohake et al., 2010; Webb et al., 2013). Nonetheless, overall, and given the high probability of partial redeposition within peatland complexes, this suggests that average annual C losses from UK agricultural peatlands due to wind erosion are likely to be substantially lower than losses due to mineralisation of SOM.

6.3.2. Erosion impacts and vulnerability

However, sediment losses resulting from wind erosion are rarely spatiotemporally consistent. Whilst Chappell and Warren (2003) found average erosion rates for East Anglia to be low, the maximum magnitude of erosion (-32.6 t ha^{-1} yr^{-1}) and accumulation (37.5 t ha^{-1} yr^{-1}) rates observed within their study area suggest that localised sediment redistribution may be substantial. Parent et al. (1982) observed much lower subsidence rates where organic soil cultivated for vegetable production was sheltered from wind erosion (cover-cropping/windbreaks; 0.99 cm yr^{-1}), than where fields were exposed (4.53 cm yr^{-1}), suggesting that sediment redistribution rates are highly management/context dependent. Severe episodic erosion events associated with storm events and bare soil have long been observed in the East Anglian Fens and are referred to as 'Fen blows' (Thompson, 1957). Cumming (2018) and Newman (2022) both identified temporal peaks in horizontal mass transport above agricultural peat soil in East Anglia resulting from exposure of bare soil to high wind speeds in the spring. Modelling of erosion rates for mineral soils in East Anglia also identified high erosion potential in the autumn (Böhner et al., 2003), which is likely explained by bare soil following harvest and the onset of early winter storms. Cumulative wind erosion losses therefore tend to be dominated by spatial hotspots of vulnerability or episodically severe erosion events, and aggregate measures may not clearly elucidate acute/concentrated local impacts.

Dust emissions can influence global climate (Kok et al., 2023) and have longer-range air quality (Prospero et al., 2014; Stafoggia et al., 2016) and ecological impacts (Brahney et al., 2015; Dansie et al., 2022). However, the severity of many wind erosion impacts will be geographically skewed towards the location of sediment detachment, and land degradation impacts are centred on the detachment site by definition (Chappell et al., 2019). This contrasts markedly with GHG emissions, which contribute to global climate change, the impacts of which are not geographically correlated with emissions (Harrington et al., 2016, 2018). Consequently, comparison of wind erosion and GHG emissions based on average annual C losses will likely underestimate the true costs of wind erosion. This is paralleled by the global (GHG emissions) and local (subsidence) distribution of costs from SOM mineralisation in drained peatlands. Subsidence results in substantial local costs by increasing drainage/flood management requirements and causing damage to infrastructure (Page et al., 2020), which are simply not captured if costs are framed in terms of GHG emissions.

In chapter 5 we highlighted that agriculture on drained peatlands currently appears justifiable for high-value crops, where economic benefits from production outweigh the (carbon price dependent) costs of GHG emissions. Any factor that reduces the profitability of agricultural production therefore has potentially important consequences for the balance between economic profitability and environmental costs. Wind erosion can have substantial impacts on both yield and crop value, by damaging crops and contaminating fresh produce, and vegetable crops are particularly susceptible to these effects (Riksen and de Graaff, 2001; Genis et al., 2013). Therefore, if responsible peatland agriculture strategies encourage a focus on high-value cropping (e.g. vegetables), erosion mitigation would increase in importance and research efforts should be focussed on identifying strategies with financial net-benefits.

The results of the study presented in Chapter 3 clearly showed that the potential vulnerability of bare agricultural peat soil to wind erosion increases throughout the crop management cycle. Farm management activities were associated with increases in erosion vulnerability including tillage and vehicle traffic (mechanical action), and irrigation (hydraulic action). This was an observational study and therefore, direct management influences cannot be entirely separated from the influences of prolonged exposure of surfaces to environmental stresses (e.g. abrasion by movement of surface particles or rainfall). If it were necessary to identify the exact causal contributions of different management operations to erosion rates, then controlled experiments would be required.

The benefit of our observational approach is that erosion vulnerability was estimated under the actual conditions found in the field (in practice a combination of management and exposure effects will always be present), providing a clear indication of how erosion vulnerability varies in practice. Our most practically important finding was that agricultural peat was broadly vulnerable to wind erosion following salad vegetable crop establishment and irrigation, regardless of SOM content. If as discussed above, the focus of erosion mitigation for responsible peatland management is primarily to minimise crop damage/contamination, then our results strongly support the use of erosion mitigation strategies to minimise bare soil during the establishment period of high-value vegetable crops.

6.3.3. Erosion mitigation options

Vegetation providing approximately 40% ground cover is typically sufficient to inhibit wind erosion (Funk and Engel, 2015). Cover cropping can therefore be highly effective for erosion mitigation but no-/min-till practices are required for ground cover (residue) benefits to be retained during the establishment period of subsequent crops (Schnarr et al., 2022). Hoepting et al. (2008) trialled a min-till, wheat cover crop system on peat, observing 60% ground cover prior to planting and 30% at full crop development with no significant effect on yield. The protective effect of flat residue varies with wind speed but it can offer substantial erosion mitigation between 30-50% ground cover (Lin et al., 2021). No-/min-till and cover cropping may therefore provide substantial erosion mitigation benefits for vegetable production on agricultural peatlands. However, the effects on yield would need to be confirmed for a wider range of crops. Additionally, the effects of cover cropping on the GHG balance of agricultural peatlands are poorly understood (Newman, 2022) and may be highly species dependent (Wen et al., 2019), so further research is required before robust recommendations can be made.

Companion crops – interplanted cereals between row crops – can substantially reduce wind speed across the soil surface. Schultz and Carlton (1959) observed wind speed reductions of ~25% (relative to unprotected soil) when wind direction was perpendicular (90°) or parallel (0°) to companion crop rows, and 35-40% for intermediate wind directions (0-90°). The wind speed reductions seen with companion crops are similar to the reductions expected at a downwind distance equal to 10 times the height of a tree shelterbelt (Schultz and Carlton 1959) and larger reductions are seen with better established companion crops (Schultz et al., 1963). Given low shading, the wide range of directional protection and the ease of removal once cash crops are established, companion cropping would appear to be a very attractive erosion mitigation strategy for vegetable production on peat and may complement no-/min-till and cover-cropping practices.

No-/min-till methods have not been optimised for all crops (Defra, n.d.) and economic realities/risk aversion will understandably delay adoption of these methods until effects on crop yield/quality are well understood. Therefore, there remains a high likelihood of bare soil during vegetable crop establishment for the foreseeable future. Bare soil is also inevitable on areas subject to heavy vehicle traffic, which based on admittedly limited measurements appear to be highly vulnerable to erosion on high-SOM peat (Chapter 3). Erosion from these areas will also be further exacerbated by the mechanical action and turbulence created by vehicles themselves (Goosens et al., 2001; Kuhns et al., 2010). Whilst controlled traffic systems may help to limit these effects (Tullberg et al., 2007), some traffic will always be required and targeted erosion mitigation for remaining bare soil areas/periods would be highly desirable.

In Chapter 4, we evaluated the performance of several commercially available chemical soil stabilisers. Chemical soil stabilisation has generally been considered unsuitable for peat (Riksen et al., 2003) but we know of no published studies demonstrating this. Therefore, our study was intended as a first step in developing this strategy by evaluating whether stabilisation was possible even under optimal conditions; this was a laboratory study with measurements made shortly after application. Polyacrylamide (PAM) was observed to stabilise peat soil, significantly increasing the wind speed required to initiate erosion and significantly reducing erosion losses. However, these effects were only seen at a relatively high application rate, suggesting PAM stabilisation may be expensive in practice (Xiong et al., 2018). PAM stabilisation would therefore likely only be suitable for short periods/small areas of acute erosion vulnerability when its use provided clear economic benefits. It may also have utility for one-off treatments to stabilise bare soil during peatland restoration.

The performance of PAM could potentially change when surfaces are exposed to environmental stressors (e.g. rainfall, UV; Xiong et al., 2018), management effects (e.g. vehicle traffic, irrigation; Preston et al., 2020) and abrasive loose erodible surface material (Armburst, 1999; Preston et al., 2020). Field tests are therefore now essential to ascertain whether the erosion mitigation performance and effect duration of PAM in the field are adequate to provide a practical and cost-effective erosion mitigation option for agricultural peatlands.

Shelterbelts, hedgerows and windbreaks may also have a role to play in landscape scale management of wind erosion on agricultural peatlands. These interventions increase the aerodynamic roughness of the landscape, resulting in reduced wind speeds and surface shear forces (Bartus et al., 2017). Wind speed reductions due to shelterbelts can be substantial within a distance 5-10 times the height of the shelterbelt (Řeháček et al., 2017; Chang et al., 2021). Therefore, when combined with reductions in field size (especially field length in the direction of the prevailing wind), they have potential to substantially reduce erosive forces across the landscape (Bartus et al., 2017; Yang et al., 2018). The vast majority of eroded particles are transported at heights less than 0.3 m above the surface (Cumming, 2018, Newman, 2022). Consequently, these interventions can also act as a physical barrier to sediment transport onto/off of fields, which could facilitate spatiotemporally targeted erosion mitigation by minimising issues with sediment being imported from adjacent non-target areas.

The benefits of shelterbelts and other erosion mitigation options are often considered noneconomic, creating barriers to adoption where financial margins are tight (Rempel et al., 2017). However, if damage/contamination of high-value fresh vegetables could be reduced, agricultural peatlands may represent a context in which the primary benefit of erosion mitigation strategies is economically explicit. Erosion mitigation research for these systems should therefore evaluate mitigation measures on the basis of their cost-effectiveness, with a primary focus on explicitly financial local costs (e.g. crop damage) instead of diffuse global costs with valuation challenges (e.g. CO_2 emissions associated with aeolian C losses).

6.4. Concluding remarks

The environmental impacts of drainage-based agriculture on peatlands are substantial and are justifiably a policy priority both globally and within the UK. The programme of research presented in this thesis suggests that responsible peatland management strategies will need to align with the wetland character of peatlands by reducing average drainage depths. However, they must also acknowledge the substantial socioeconomic benefits of peatland agriculture and provide solutions for the significant water management challenges that wetter peatland landscapes could create.

Responsible peatland management will likely produce land use mosaics, with: (i) fully rewetted areas to maximise climate change mitigation; (ii) partially rewetted areas under agriculture to balance climate and economic benefits; (iii) fully drained areas, where production of high-value, wet-intolerant crops provides a net-benefit even with high GHG emissions. Optimising the cost-benefit balance for agricultural peatlands will likely favour the concentration of high-value agriculture on these highly productive soils and may support reducing the area of low-value agriculture on peat.

Wind erosion rates appear to indicate substantially lower aeolian C losses from UK agricultural peatlands than losses due to GHG emissions. However, other impacts of erosion can be substantial on a local scale. In particular, if crop damage/contamination is severe, wind erosion may have important effects on profitability (e.g. during establishment of high-value vegetable crops), and so mitigation should be part of comprehensive responsible management strategies aiming to optimise the cost/benefit ratio of peatland agriculture.

The development of responsible peatland management strategies should actively involve land managers/food producers, to ensure proposed solutions are workable in practice. The communities occupying agricultural peatlands have a long history of overcoming environmental management challenges and this adaptive capacity should be drawn upon to ensure the success of efforts to adapt peatland management to the changing sociopolitical and environmental context.

It is easy to focus on the risks associated with change and therefore dismiss the opportunities. However, well-designed responsible peatland management strategies could increase the resilience of food production systems, deliver environmental benefits, protect our valuable peat resources and create an investment in our future. The UK has the opportunity to be a global leader by creating thriving, innovative, green and profitable peatland landscapes, and delivering an important contribution to international climate change mitigation efforts.

6.5. References

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Appendices

Appendix 2.1

We have broadly defined mid-latitude peatlands as non-tropical and non-polar for the purposes of this analysis. The 'tropical' boundary can be represented conveniently by the tropics of Cancer (23.5° N) and Capricorn (23.5° S), due to relatively low peatland coverage at these specific latitudes (Fig. A2.1.1). The 'polar' boundary is less easily defined in terms of latitude, due to the large peatland area seen at higher latitudes in the Northern Hemisphere. Therefore, we did not use the Arctic and Antarctic circles (66.5° N and S respectively) to guide our 'polar' boundaries. Given the importance of climatic conditions in modulating peatland function, a better indicator of the 'polar' boundaries is given by the limits of the polar climate zones (ET and EF in Fig. A2.1.2; Peel et al., 2007). This definition includes large areas of boreal peatland in the northern hemisphere particularly, where climatic conditions are not currently viable for agricultural use. However, given the potential for expansion of the global agricultural area towards the poles by 1200 km by the year 2099 (an area of 5 million km^2), these peatlands could be highly vulnerable to future drainage (Unc et al., 2021). Their inclusion thus represents an acknowledgement that climate zones and viable activities are not fixed over time. It also represents – due to the vast carbon stocks in these regions – a clear reason why a policy of 'no further peatland drainage' is essential if overall peatland greenhouse gas emissions are to be halted. An argument can be made for defining the 'tropical' boundary using climate zones also. The main boundary case of note would be the Florida Everglades (USA). The inclusion of this location in our definition is likely constructive as their socio-economic circumstances are more closely aligned with other mid-latitude peatlands in the USA. However, due to climatic conditions, soil organic matter mineralisation and soil respiration rates may be higher than at most other mid-latitude sites and it may represent something of an outlier. It is important to note that our analysis was not heavily reliant on data from this location and so this boundary decision does not affect our conclusions.

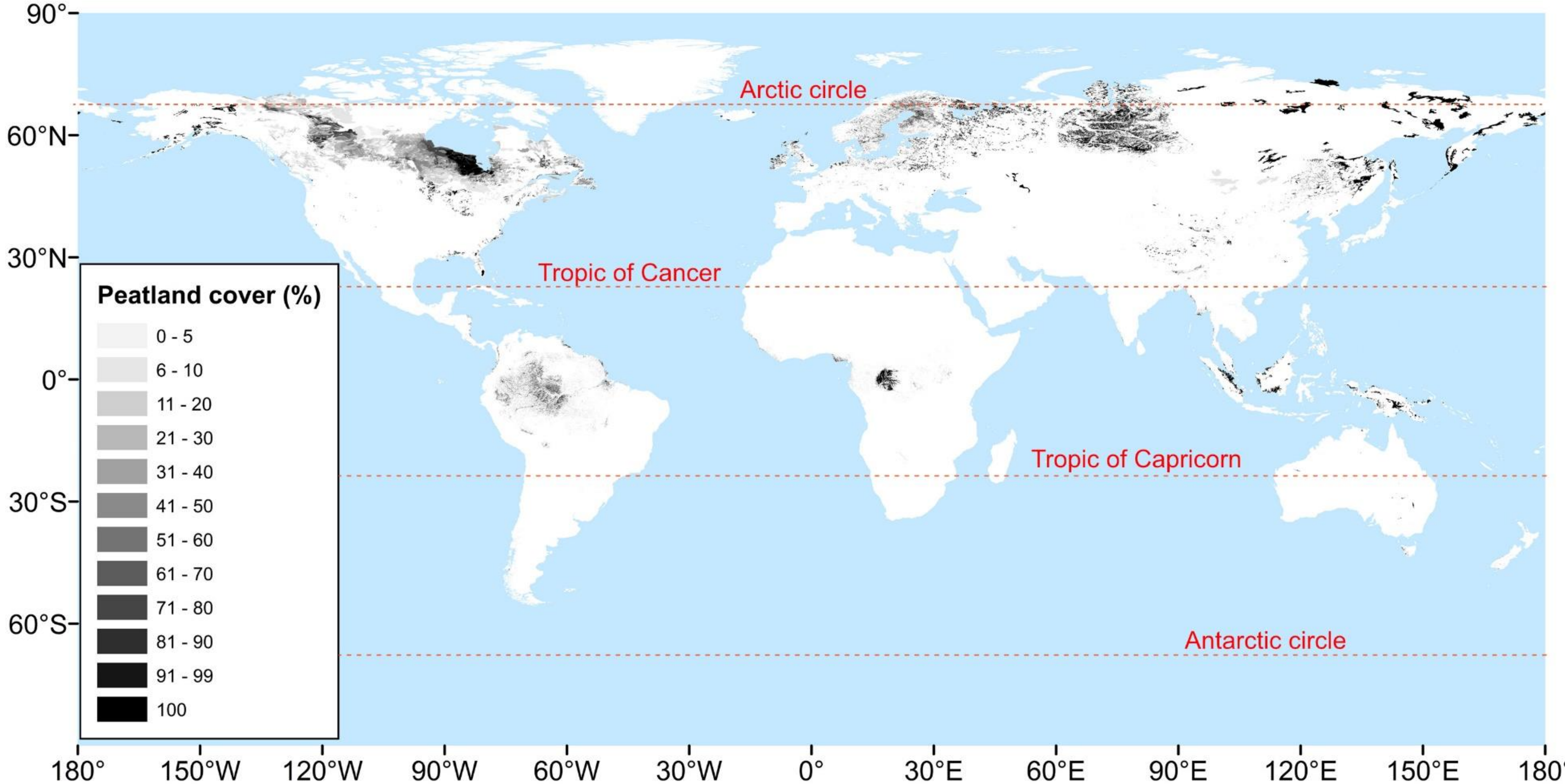

Figure A2.1.1. Global peatland distribution derived from PEATMAP. The Arctic and Antarctic circles (66.5° North and South respectively) along with the Tropics of Cancer and Capricorn (23.5° North and South respectively) are hand drawn additions to the original figure, shown as red dashed lines for the readers convenience (the locations are not exact but indicative). Reprinted with permission from Xu et al. (2018), © 2018 Elsevier.

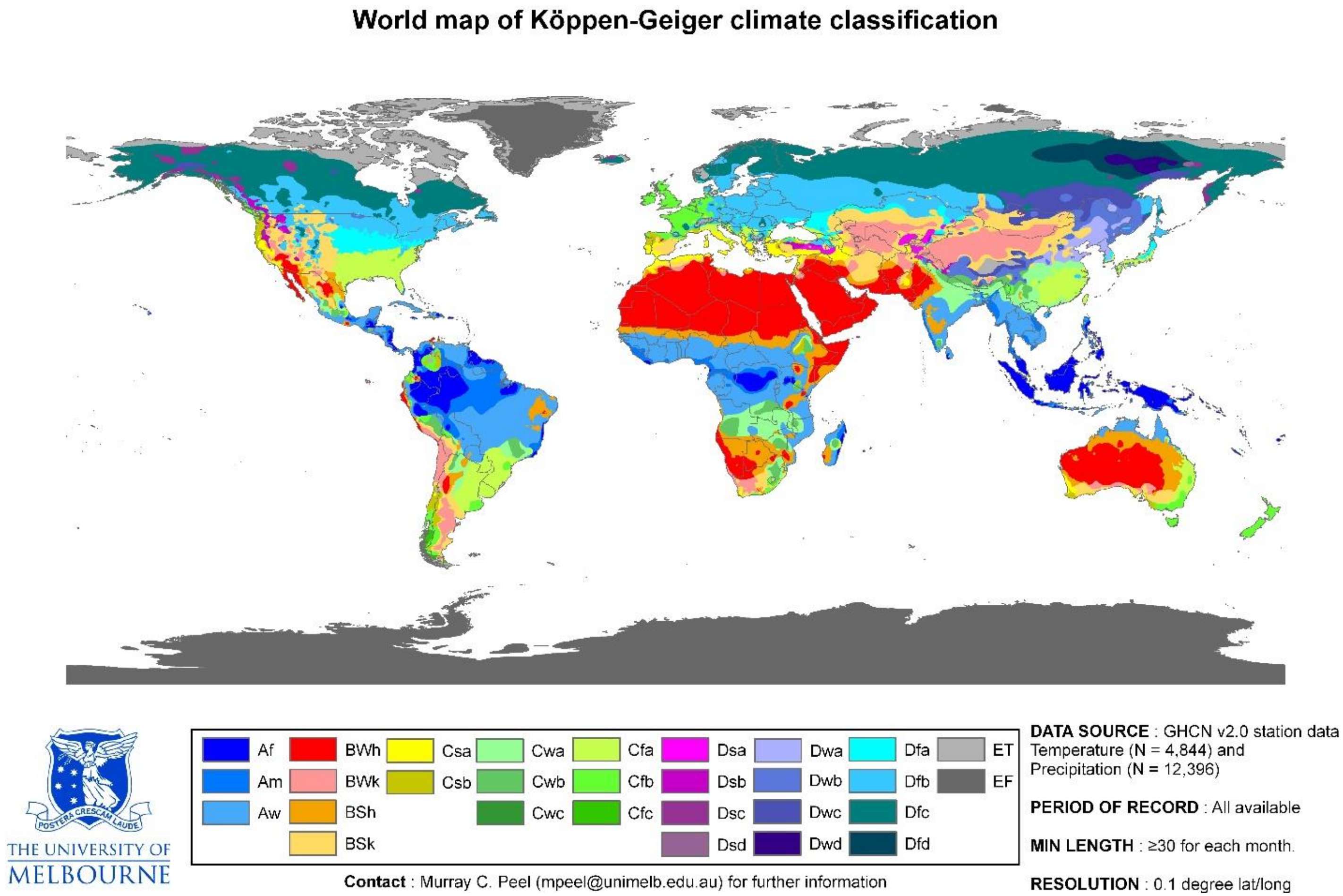

Figure A2.1.2. World map of the Köppen-Geiger climate classification. The first letters in each category signify: A = Tropical, B = Arid, C = Temperate, D = Cold (Boreal), E = Polar. Further letters indicate temperature/rainfall/habitat descriptors and are described fully in the original publication. The area shown as Dfc best approximates those regions that whilst not currently climatically favourable for agriculture, would see increasing development with climate warming. Image reprinted from Peel et al. (2007), © Peel, M.C., Finlayson, B.L., McMahon, T.A., 2007. This work is licensed under the Creative Commons Attribution-NonCommercial-ShareAlike 2.5 License. Original image downloadable at: https://hess.copernicus.org/articles/11/1633/2007/hess-11-1633-2007-supplement.zip

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Appendix 2.2

Subsidence values from a number of studies are collated in Table **A**2.**2**.1. The median subsidence rate observed is 2.0 cm yr^{-1} (Quartiles = 1.3 – 2.7, n = 48). Median subsidence rates observed under cropland were slightly higher (2.1 cm yr^{-1}; Quartiles = 1.5 – 3.0, n = 21) than for grassland (1.0 cm yr^{-1}; Quartiles = 0.7 – 1.5, n = 13). Any difference would likely be a reflection of differences in drainage intensity between land uses, as water table depths appear to be the dominant factor determining subsidence rates (Evans et al., 2019). A higher median subsidence rate was observed in North American studies (2.5 cm yr^{-1}; Quartiles = 1.9 – 3.1, n = 19) than European studies (1.4 cm yr^{-1}; Quartiles = 0.8 – 2.1; n = 25). This result is almost certainly biased by the inclusion of warmer climate peatlands (e.g. Florida Everglades and Sacramento-San Joaquin Delta) in the North American sample, where mineralisation rates would be greater due to enhanced reaction kinetics (Stephens and Stewart, 1969). However, it serves as a useful demonstration of the potential for future climate warming to increase environmental impacts resulting from drainage and highlights the universal fragility of peatlands.

Table A2.2.1. Annual subsidence values for mid-latitude peatlands drained for agriculture. This table is an update of previous research published in Evans et al. (2019).

Region	Location	Land use	Subsidence (cm yr^{-1})	Source
Asia	Japan	Arable	3.00	Miyaji et al. (1995)
Australasia	New Zealand	Grassland	3.40	Schipper and McLeod (2002)
	New Zealand	Grassland	2.56	Fitzgerald and McLeod (2004)
	New Zealand	Grassland	1.90	Pronger et al. (2014)
Europe	Belarus	Unknown	2.10	Armentano (1979)
	Germany	Arable	2.70	Eggelsmann and Bartels (1975)
	Germany	Grassland	0.67	Eggelsmann and Bartels (1975)
	Germany	Grassland	0.50	Eggelsmann (1976)
	Germany	Grassland	0.83	Kluge et al, (2008)
	Ireland	Unknown	1.80	Armentano (1979)
	Italy	Arable	1.75	Gambolati et al. (2005)
	Italy	Arable	0.75	Zanello et al. (2011)
	Netherlands	Grassland	0.88	Schothorst (1977)
	Netherlands	Unknown	0.70	Armentano (1979)
	Netherlands	Unknown	1.35	Armentano (1979)
	Norway	Grassland	2.50	Grønlund et al. (2008)
	Norway	Unknown	2.50	Armentano (1979)
	Poland	Grassland	0.60	Grzywna (2017)
	Sweden	Arable	2.50	Berglund and Berglund (2010)
	Sweden	Arable	1.50	Berglund and Berglund (2010)
	Sweden	Grassland	1.00	Berglund and Berglund (2010)
	Switzerland	Arable	1.26	Leifeld et al. (2011)
	Switzerland	Unknown	1.27	Wüst-Galley et al. (2019)
	Ukraine	Grassland	2.00	Lipka et al. (2017)
	UK	Arable	1.37	Richardson and Smith (1977)
	UK	Arable	3.05	Hutchinson (1980)
	UK	Arable	1.48	Dawson et al. (2010)
	UK	Grassland	0.62	Brunning (2001)
	UK	Unknown	2.75	Armentano (1979)
North America	Canada (Ontario)	Arable	3.30	Mirza and Irwin (1964)
	Canada (Quebec)	Arable	2.07	Millette (1976)
	Canada (Quebec)	Arable	2.50	Mathur et al. (1982)

	USA (CA)	Arable	0.83	Deverel et al. (2016)
	USA (CA)	Grassland	2.20	Deverel and Leighton (2010)
	USA (CA)	Unknown	5.35	Armentano (1979)
	USA (FL)	Arable	3.18	Stephens (1956)
	USA (FL)	Arable	3.00	Stephens et al. (1984)
	USA (FL)	Arable	1.45	Shih et al. (1998)
	USA (FL)	Arable	1.40	Wright and Snyder (2009)
	USA (FL)	Arable	1.82	Aich et al. (2013)
	USA (FL)	Unknown	2.70	Armentano (1979)
	USA (FL)	Unknown	3.45	Armentano (1979)
	USA (IN)	Arable	2.26	Jongedyk et al. (1950)
	USA (IN)	Unknown	1.85	Armentano (1979)
	USA (LA)	Unknown	3.00	Armentano (1979)
	USA (MI)	Unknown	1.85	Armentano (1979)
	USA (NC)	Arable	4.00	Ewing and Vepraskas (2006)
	USA (NY)	Unknown	2.50	Armentano (1979)

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Appendix 2.3

Data for this analysis were drawn from the IPCC Wetland Supplement (Drösler et al., 2014), and two other synthesis studies, estimating Tier 2 emission factors for the UK (Evans et al., 2017) and Germany (Tiemeyer et al., 2020). Total GHG balances were calculated as the sum of terrestrial carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O) emissions (Table A2.3.1). We used factors of 3.66412 to convert CO_2-C to CO_2 and 1.57112 to convert N_2O-N to N_2O. Values for GHGs were converted to CO_2-e by calculating 100-year global warming potentials using factors of 1, 265 and 28 for CO_2, N_2O and CH_4 respectively as described in IPCC AR5 (Myhre et al., 2013), in line with UN Framework Convention on Climate Change guidelines for national and international emissions reporting (UNFCCC, 2021). Average values for the land used categories were calculated as the arithmetic mean of the available literature values. For the extensive grassland category, we used the values for extensive grassland from Evans et al. (2017) and for shallow-drained, nutrient-rich grassland from the IPCC Wetland Supplement. For the intensive grassland category, we used the values for both deep-drained, nutrient-rich and drained, nutrient-poor grassland from the IPCC Wetland Supplement, along with the value for intensive grassland from Evans et al. (2017) and the value for grassland from Tiemeyer et al. (2020). This analysis focused on temperate and not boreal mid-latitude peatlands as (i) the IPCC tier 1 emission factors do not differentiate between grassland categories for boreal climates and (ii) the tier 2 emission factors available were predominately for temperate regions.

Table A2.3.1. Results of terrestrial greenhouse gas balance analysis for temperate agricultural peatlands. Values for greenhouse gas emissions are reported as t CO_2-e ha^{-1} yr^{-1} and as a percentage contribution to the total greenhouse gas balance. Sources: 1) Drösler et al. (2014), 2) Evans et al. (2017), 3) Tiemeyer et al. (2020).

Land use	CO_2	CH_4	N_2O	Total	% CO_2	% CH_4	% N_2O
Cropland[1]	29 (24-34)	0 (-0.1-0.1)	5.4 (3.4-7.5)	34.4	84.2	0	15.8
Cropland[2]	26 (15-38)	0 (0-0.1)	8.0 (2.8-13.1)	34.4	76.8	0.1	23.1
Cropland[3]	34 (11-41)	0.2 (0-0.5)	4.6 (0.7-16.9)	38.5	87.6	0.4	12.0
Grassland, deep-drained, nutrient-rich[1]	22 (18-27)	0.4 (0.1-0.8)	3.4 (2.0-4.6)	26.2	85.3	1.7	13.0
Grassland, drained, nutrient-poor[1]	19 (14-25)	0.1 (0-0.1)	1.8 (0.8-2.8)	21.3	91.3	0.2	8.4
Grassland, shallow-drained, nutrient-rich[1]	13 (7-20)	1.1 (-0.1-2.3)	0.7 (0.2-1.1)	14.9	88.2	7.3	4.5
Grassland intensive[2]	23 (14-33)	0.4 (-0.4-1.2)	2.5 (1.2-3.8)	26.4	88.9	1.6	9.5
Grassland extensive[2]	13 (8-19)	2.0 (0.5-3.6)	1.3 (0.1-2.6)	16.6	79.6	12.3	8.0
Grassland[3]	30 (5-40)	0.3 (0-2.4)	1.9 (0.1-9.2)	32.6	93.2	1.0	5.9
Cropland (average)	30	0.1	6.0	35.7	82.9	0.2	17.0
Grassland intensive (average)	24	0.3	2.4	26.6	89.7	1.1	9.2
Grassland extensive (average)	13	1.6	1.0	15.8	83.9	9.8	6.2

Additional references for Appendix 2.3

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Appendix 2.4

We generated an exponential function for the relationship between methane (CH_4) and water table depth (WTD) for Couwenberg et al. (2011) in order to estimate the WTD associated with minimising carbon-derived greenhouse gas balances for this study. We digitised data with WTD > 0 m from Figure 2 of Couwenberg et al. (2011) using WebPlotDigitzer 4.5 (Rohatgi, 2010). This produced a dataset with n = 68 (Table A2.4.1). The original dataset is described as n = 99, with 3 points at WTDs > 0 m. We therefore obtained 71% of the original dataset. Data that we missed will likely have been less influential data on average, because outliers were easily digitised whereas data in well-represented (visually 'crowded') regions of the plot were more challenging to extract. Therefore, this dataset provides a reasonable representation of the main features of the original dataset. We fit an exponential function using the *nls* function in R v4.0.4 (R Core Team, 2021; Equation A2.4.1).

Equation A2.4.1:

$$CH_4 = -1.18 + 203.54e^{-7.74 \cdot WTD}$$

There was clear heterogeneity of variance and non-normality of residuals resulting from greater spread of values at near surface WTDs than at deeper WTDs (see Fig. A2.4.1). This is likely because vegetation mediates the relationship between WTD and CH_4 at near-surface WTDs. The original linear relationship from Couwenberg et al. (2011) used only sites with aerenchymatous shunt species and they did not include open vegetation without shunt species or sites with trees in their regression. We included all available data, without adjusting for vegetation because that information was not available. Visual inspection suggests the mean model prediction offers an acceptable description of the trend. It also agrees well with the Tiemeyer et al. (2020) model for rewetted sites, where variation was also clearly much higher at near-surface WTDs. As such, it was deemed adequate for our purpose. Investigation of the role of vegetation in mediating this relationship represents a potentially important future research target.

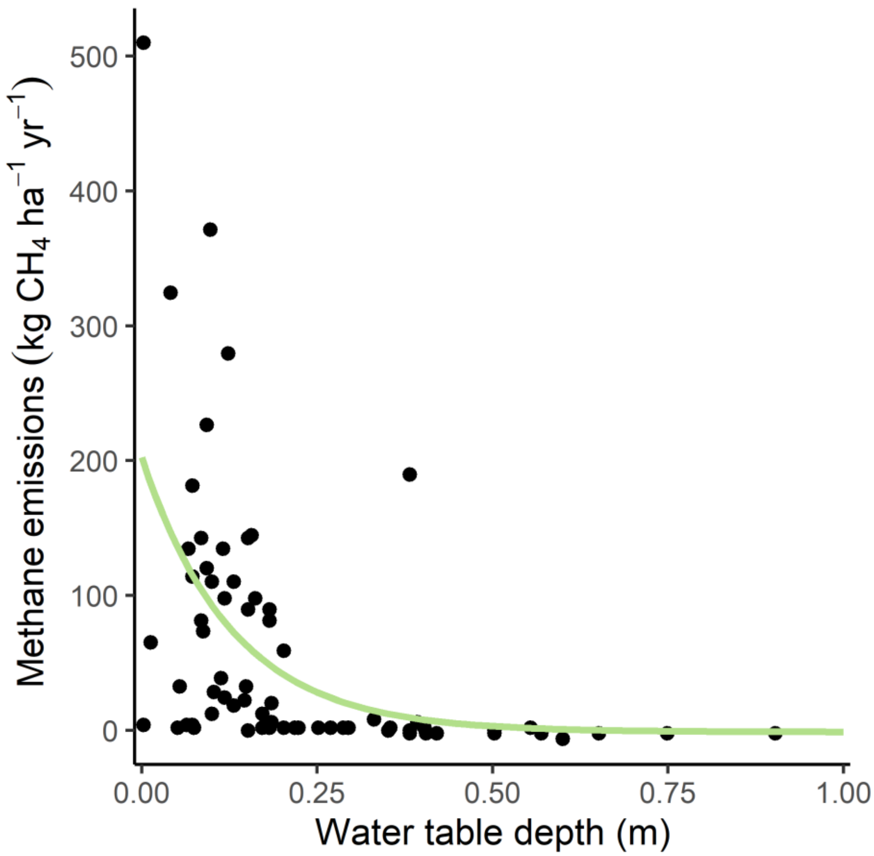

Figure A2.4.1. Exponential relationship between methane and water table depth estimated from a subset of Couwenberg et al. (2011) data.

Table A2.4.1. Digitised dataset.

Methane emissions (kg CH_4 ha^{-1} yr^{-1})	**WTD (m)**
-2.04082	0.902564
-2.04082	0.748718
-2.04082	0.651282
-6.12245	0.6
-2.04082	0.569231
2.040816	0.553846
0	0.502564
-2.04082	0.502564
-2.04082	0.420513
-2.04082	0.405128
2.040816	0.402564
6.122449	0.392308
189.7959	0.382051
-2.04082	0.382051
0	0.382051
2.040816	0.353846
0	0.351282
8.163265	0.330769
2.040816	0.294872
2.040816	0.287179
2.040816	0.269231
2.040816	0.251282
2.040816	0.223077
2.040816	0.217949
59.18367	0.202564
2.040816	0.202564
20.40816	0.184615
6.122449	0.184615
89.79592	0.182051
81.63265	0.182051
2.040816	0.182051
2.040816	0.171795

12.2449	0.171795
97.95918	0.161538
144.898	0.15641
142.8571	0.151282
89.79592	0.151282
0	0.151282
32.65306	0.148718
22.44898	0.146154
110.2041	0.130769
18.36735	0.130769
279.5918	0.123077
97.95918	0.117949
24.4898	0.117949
134.6939	0.115385
38.77551	0.112821
28.57143	0.102564
110.2041	0.1
12.2449	0.1
371.4286	0.097436
226.5306	0.092308
120.4082	0.092308
73.46939	0.087179
142.8571	0.084615
81.63265	0.084615
2.040816	0.074359
181.6327	0.071795
114.2857	0.071795
4.081633	0.071795
134.6939	0.066667
4.081633	0.064103
32.65306	0.053846
2.040816	0.051282
324.4898	0.041026
65.30612	0.012821
510.2041	0.002564
4.081633	0.002564

Additional references for Appendix 2.4

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Rohatgi, A., 2010. WebPlotDigitizer. Version 4.5 https://apps.automeris.io/wpd/

Appendix 3.1

This analysis involved the fitting of a linear mixed effects model to our dataset, to predict density-adjusted WEM (%) from SOM (%) and the proportion of soil mass < 0.85 mm in diameter ($P_{0.85}$; %), resulting in equation A3.1.1 ($p<0.0001$; Marginal $R^2 = 0.96$).

Equation A3.1.1:

$$WEM = -0.60728 + 1.10506(P_{0.85}) + 0.6247(SOM)$$

This was then applied to surface mean $P_{0.85}$ and SOM values from Zobeck et al. (2013) to estimate approximate density-adjusted WEM values for their surfaces. Finally, we adjusted the Run 0 PM_{10} fluxes from Zobeck et al. (2013) for C content. Run 0 was used as this represented an initial blow-off run on the freshly prepared surfaces. Runs 1 and 2 from Zobeck et al. (2013) involved the addition of abrasive material and so were deemed less comparable to our measurements on surfaces under field conditions. Clearly this approach is an approximation but Figure A3.1.1. shows relatively good agreement between our values and our estimates for the equivalent values from Zobeck et al. (2013). This provides further supporting evidence that density-adjusted WEM may be an important predictive variable for erosion rates from lowland agricultural peat soils.

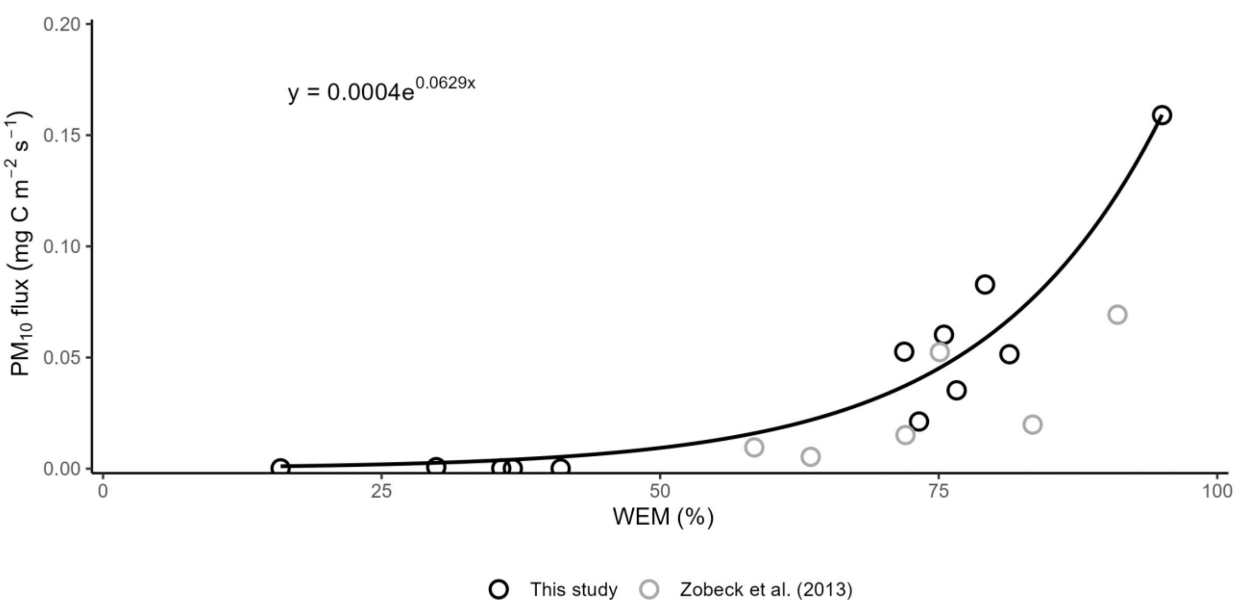

Figure A3.1.1. Comparison of estimated values from Zobeck et al. (2013) with our observations.

Appendix 4.1

Correlations between soil property variables (Table A4.1.1) and erosion variables (Table A4.1.2).

Table A4.1.1. Correlations between soil property variables. Spearman's Rho with Holm correction for multiple testing. Significance indicated as follows: * = p<0.05, ** = p<0.01.

Variable	Soil organic matter content	Wind erodible material	Dry aggregate stability	Crust strength	Volumetric moisture content
Bulk density	0.24	-0.07	0.06	0.23	0.82 **
Soil organic matter content		-0.48 *	0.54 **	0.11	0.62 **
Wind erodible material			-0.58 **	-0.33	-0.29
Dry aggregate stability				-0.02	0.30
Crust strength					0.30

Table A4.1.2. Correlations between dust and saltation dependent variables. Spearman's Rho with Holm correction for multiple testing. Significance indicated as follows: * = p<0.05, ** = p<0.01. Values in square brackets indicate the friction velocities of step phases (m s^{-1}).

Variable	Mean saltation rate [0.69]	Mean saltation rate [0.82]	Cumulative saltation count	Maximum saltation rate	Mean PM_{10}-C flux [0.55]	Mean PM_{10}-C flux [0.69]	Mean PM_{10}-C flux [0.82]	Cumulative PM_{10}-C emissions	Maximum PM_{10}-C flux
Mean saltation rate [0.55]	0.69 **	0.38	0.59 **	0.28	0.40	0.59 **	0.34	0.50 *	0.31
Mean saltation rate [0.69]		0.62 **	0.86 **	0.53 **	0.25	0.91 **	0.57 **	0.81 **	0.59 **
Mean saltation rate [0.82]			0.86 **	0.97 **	0.14	0.59 **	0.64 **	0.71 **	0.67 **
Cum. saltation count				0.79 **	0.26	0.84 **	0.70 **	0.89 **	0.72 **
Max. saltation rate					0.07	0.50 *	0.64 **	0.67 **	0.68 **
Mean PM_{10}-C [0.55]						0.29	-0.05	0.12	-0.03
Mean PM_{10}-C [0.69]							0.65 **	0.85 **	0.65 **
Mean PM_{10}-C [0.82]								0.90 **	0.94 **
Cum. PM_{10}-C emissions									0.89 **

Appendix 4.2

Parameter estimates from linear models describing variation in the threshold friction velocity of entrainment (u^*_t) for PM_{10} emissions (Table 4.2.1) and saltation (Table 4.2.2).

Table 4.2.1. Parameters of linear model describing u^*_t for PM_{10} emissions. For surfaces where u^*_t was determined ($n = 40$; $p = 0.001$; $R^2 = 0.29$).

Term	Fitted range	Estimate	Standard error	t-value	p-value
Intercept		1.006	0.103	9.74	<0.0001
SOM (%)	50.5 - 79.5	-0.003	0.001	-2.43	0.0202
WEM (%)	9.7 – 46.7	-0.004	0.001	-3.51	0.0012
Shear strength (kPa)	0 – 2.55	0.054	0.023	2.34	0.0248

Table 4.2.2. Parameters of linear model describing u^*_t for saltation. For surfaces where u^*_t was determined ($n = 40$; $p = 0.0004$; $R^2 = 0.34$).

Term	Fitted range	Estimate	Standard error	t-value	p-value
Intercept		1.078	0.104	10.39	<0.0001
SOM (%)	50.5 - 79.5	-0.004	0.001	-3.07	0.0040
WEM (%)	9.7 – 46.7	-0.005	0.001	-3.73	0.0007
Shear strength (kPa)	0 – 2.55	0.057	0.023	2.47	0.0184

Appendix 4.3

There was a significant overall effect of treatment on mean saltation rate at a u^* of 0.55 m s^{-1} (Phase S1; $p = 0.006$). All chemical stabilisers and equivalent volume water treatments had significantly lower mean saltation rates than the untreated control treatment ($p<0.01$; Figure A4.3.1a). There were no significant differences between chemical stabilisers and equivalent volume water treatments at this u^*.

There was a significant overall effect of treatment on mean saltation rate at a u^* of 0.69 m s^{-1} (Phase S2; $p = 0.006$). Treatment P5 had a significantly lower mean saltation rate than the untreated control treatment ($p = 0.017$; Figure A4.3.1b) but was not significantly different from the equivalent water volume application (W5; $p = 0.97$). Treatment P4b also had a significantly lower mean saltation rate than the control treatment ($p = 0.036$). However, this was an artefact from the discontinued run on the replicate with the highest erosion rate and should not be interpreted as evidence of erosion mitigation. There were no significant differences between any other treatments at this u^*.

There was a significant overall effect of treatment on mean saltation rate at a u^* of 0.82 m s^{-1} (Phase S3; $p = 0.004$). Treatment P5 had a significantly lower mean saltation rate than both the control treatment ($p = 0.003$; Figure A4.3.1c) and W5 ($p = 0.026$). There were no significant differences between any other treatments at this u^*.

There was a significant overall effect of treatment on cumulative saltation count over the test program duration ($p = 0.004$). Treatment P5 had a significantly lower cumulative saltation count than the untreated control treatment ($p = 0.0006$; Figure A4.3.1d) but not W5 ($p = 0.31$). However, P5 did not exceed the u^*_t for saltation in any replicate and the cumulative saltation count essentially represents the sum of background noise recorded by the optical gate sensors. It is clear from Figure 4.3.1d that P5 strongly outperformed W5 with regard to suppression of saltation as the cumulative saltation count was essentially zero. There were no significant differences in cumulative saltation count between any other treatments.

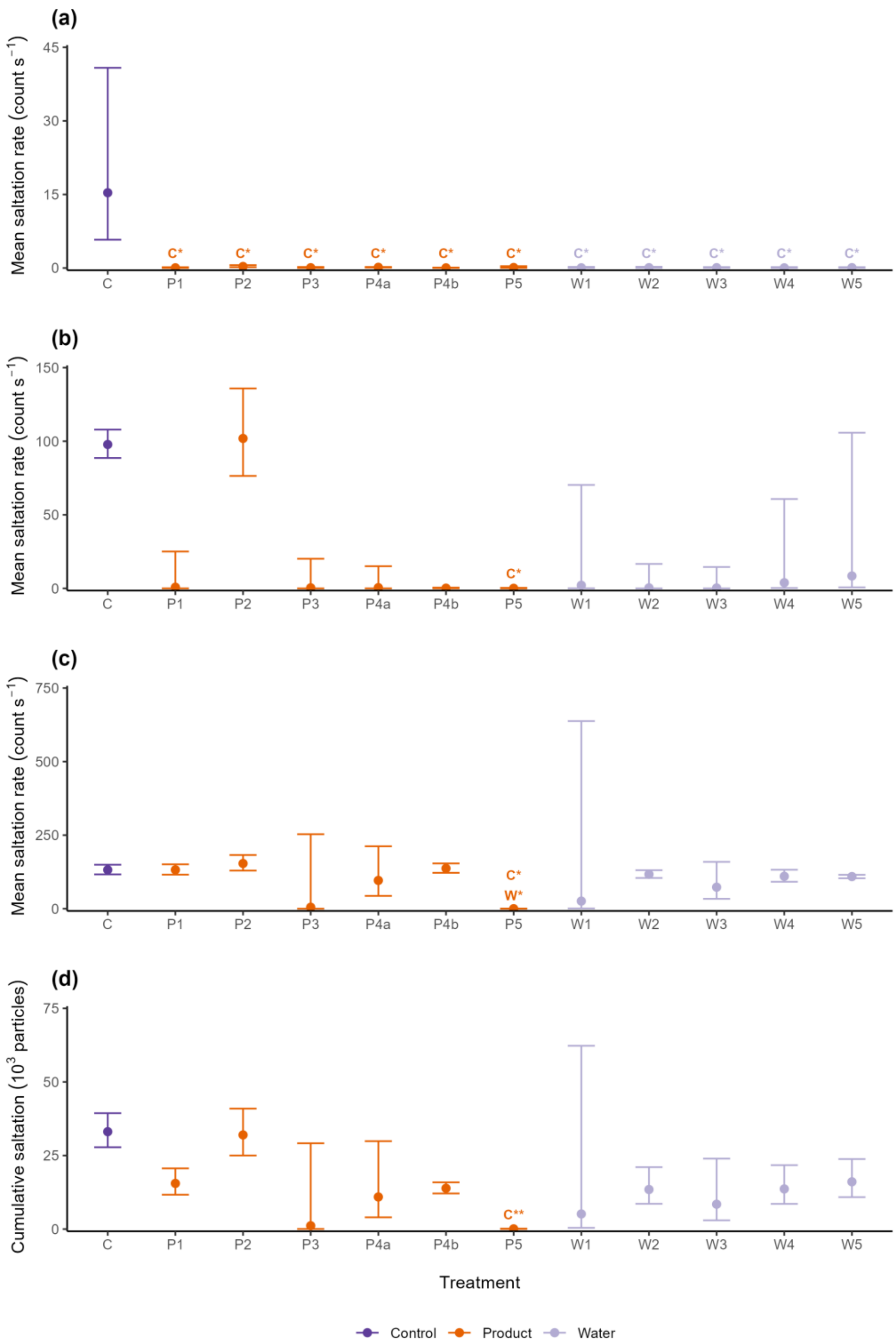

Figure A4.2.1 Treatment effects on saltation rates. (a) Step mean saltation rate for $u^* = 0.55$ ms^{-1}. (b) $u^* = 0.69$ ms^{-1}. (c) $u^* = 0.82$ ms^{-1}. (d) Cumulative saltation count over test period. Data presented as geometric means (points) and geometric standard deviations (error bars). Annotations indicate results of pre-planned contrasts: C indicates significant difference from control, W indicates significant difference from equivalent volume water treatment, * = $p<0.05$, ** = $p<0.001$, *** = $p<0.0001$. Note the differences in y-axis scale/unit between panels.

Appendix 5.1

A5.1.1. Southern European field production inventory details

The Spanish farms produce two crops of field lettuce annually and inputs were aggregated to produce annual values. The proportions of fertiliser N applied as ammonium and urea were estimated using Spanish production proportions for 2002-2011 (FAO, 2023). Organic fertiliser inputs were modelled as pig slurry using data from FNR (2010). We assumed no lime application due to the highly calciferous and basic soils in the region (García-Lorenzo et al., 2015). Irrigation water is largely obtained from groundwater sources and requires pumping to the surface (Milà i Canals et al., 2008) with an energy use of 0.4 kWh m^{-3} (Plappally and Lienhard V, 2012). We treated 20% of the water volume as spray irrigation to aid crop establishment and 80% as drip irrigation during the growing period. We used a value of 0.37 kWh m^{-3} for sprinkler irrigation energy use and 0.167 kWh m^{-3} for drip irrigation (Plappally and Lienhard V, 2012).

A5.1.2. United States field production inventory details

On the Central Coast, climate conditions allow production of two lettuce crops annually. However, the shorter growing season in the Southern Desert allows production of only a single crop. Yields were calculated using the iceberg regression equations for Monterey and Imperial Counties for 2011 from Simko et al. (2014), giving yields of 111.7 t FM ha^{-1} yr^{-1} and 40.0 t FM ha^{-1} yr^{-1} respectively. Inputs of N, phosphorous (P) and potassium (K) were derived from Turini et al. (2011). On the Central Coast, N applications range from 168-202 kg ha^{-1} for the first crop and 112-168 kg ha^{-1} for the second crop. Summing the average of both ranges gave an annual N input of 325 kg ha^{-1}. In the Southern Desert, N inputs depend on the growing season and thus temperature. Early season crops require ~168 kg ha^{-1} whilst late season crops can require 224-280 kg ha^{-1}; averaging these gave a representative input of 210 kg ha^{-1} yr^{-1}. Proportions of N applied as ammonium and urea were estimated using US agricultural use data for 2008-2016 (FAO, 2023). P inputs were 22 kg ha^{-1} prior to each planting for the Central Coast. However, soil P levels are substantially lower in the Southern Desert and pre-planting applications can reach 280 kg ha^{-1}. As this represents an extreme value, a mid-point value of 151 kg ha^{-1} was chosen to give a more robust estimate of general practices. Potassium applications equal to 134 kg ha^{-1} yr^{-1} were deemed adequate to maintain soil fertility, with no differences between regions. Lime application was based on the recommendations and median soil properties observed for Californian soils in Miller et al. (2005). Diesel use per lettuce crop was obtained from Tourte et al. (2017). We used the seeding rate for a single lettuce crop from Tourte et al. (2017) and estimated seed weight using an average value of 0.6175 g per 1000 seeds (Souza et al., 2019). Average pesticide application rates in kg active ingredient ha^{-1} were calculated using USDA NASS data for Californian lettuce crop production for 1992-2016 (USDA, n.d.). Crop water requirements vary considerably between the two regions due to climatic differences. An average lettuce crop in the Southern Desert requires 3700 m^3 ha^{-1} (Turini et al., 2011). Surface-drip irrigation is increasing on the Central Coast and covers at least 60% of the vegetable production area (Johnson, 2013). We assume the remaining 40% is sprinkler irrigated and that these proportions apply to lettuce. Using values of 500-700 m^3 ha^{-1} for a drip irrigated lettuce crop and 750-1000 m^3 ha^{-1} for a sprinkler irrigated lettuce crop, and assuming two crops per year on the central coast gives an average irrigation water requirement of 1450 m^3 ha^{-1} yr^{-1} for production in the Central Coast region (Turini et al. 2011). Irrigation energy use for the Central Coast was calculated using the values for drip and sprinkler irrigation above, with groundwater pumping energy of 0.24 kWh m^{-3} (Plappally and Lienhard V, 2012). Irrigation in the Imperial Valley is predominately furrow irrigation from canals (84%) with sprinkler irrigation making up the remainder (Scott et al. 2014). We used the value for sprinkler irrigation above and the value of 0.045 kWh m^{-3} for furrow irrigation, with no groundwater pumping energy use (Plappally and Lienhard V, 2012).

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Appendix 5.2

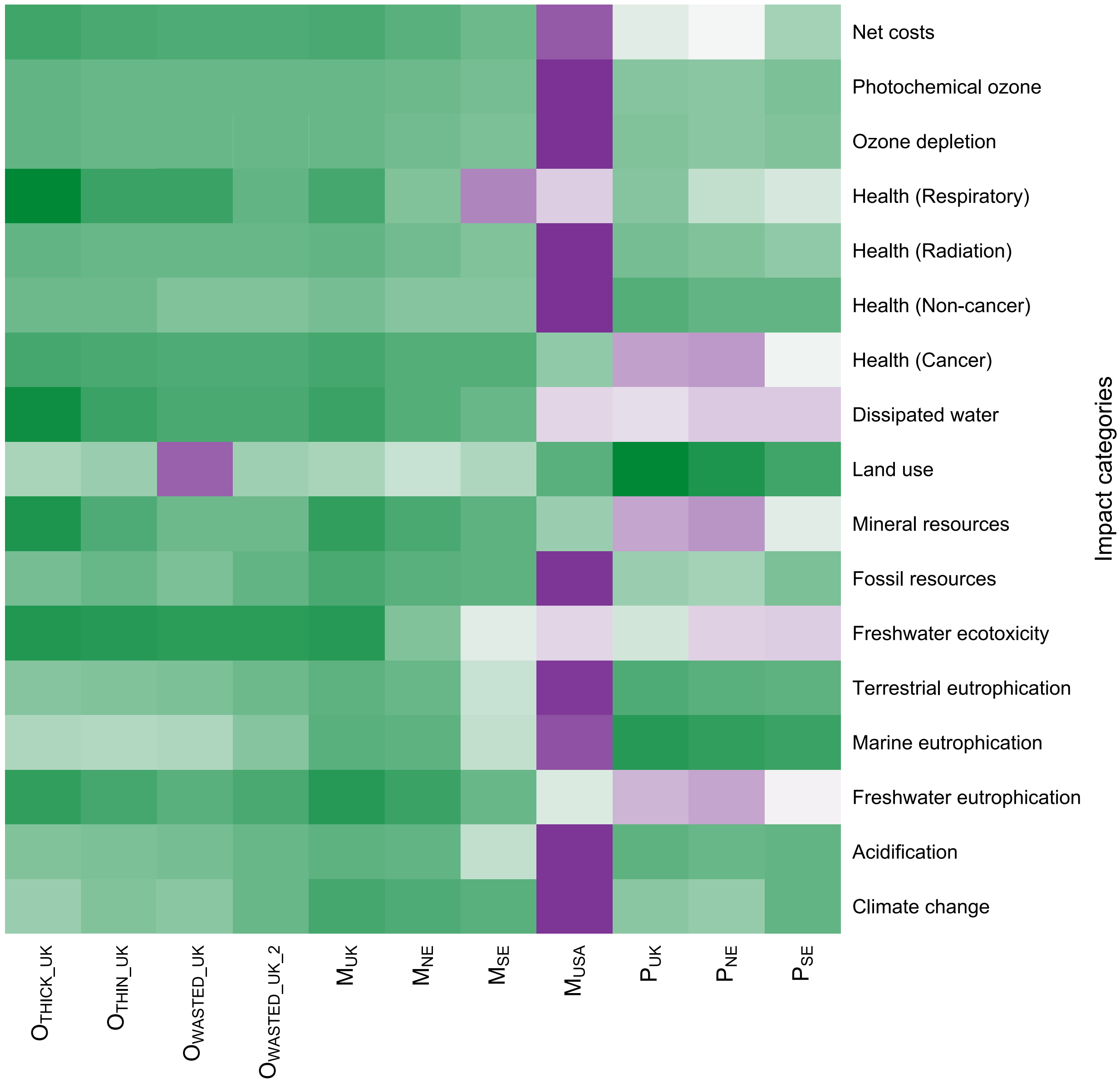

Figure A5.2.1. Heatmap of environmental burdens. Darker shades of green indicate lower values and darker shades of purple indicate higher values. Showing all scenarios, including M_{USA}.

Table A5.2.1. Uncertainty estimates of environmental footprints per kg of lettuce for domestic field mineral soil cultivation and supply chain scenarios. Shows deterministic values and the 95% simulation intervals of Monte Carlo simulations. Secondary source data inventory presented for comparison and based primarily on the average of values for three sites from Milà i Canals et al. (2008; see Casey et al., 2022 for inventory details).

Impact category	Unit	M_{UK} (Primary source data)			M_{UK} (Secondary source data)		
		Value	2.5%	97.5%	Value	2.5%	97.5%
Climate change	kg CO_2 eq.	0.25	0.22	0.31	0.25	0.22	0.35
Acidification (Terrestrial and Freshwater)	mol H^+ eq.	0.003	0.002	0.004	0.002	0.002	0.004
Freshwater ecotoxicity	CTUe	0.9	0	3.6	1.0	0	3.9
Freshwater eutrophication	kg P eq.	8.7E-05	4.0E-05	3.1E-04	8.8E-05	4.5E-05	3.3E-04
Marine eutrophication	kg N eq.	0.002	0.002	0.004	0.002	0.001	0.005
Terrestrial eutrophication	mol N eq.	0.012	0.010	0.018	0.009	0.006	0.016
Human health (Cancer)	CTUh	1.3E-08	0	3.9E-07	1.4E-08	0	5.1E-07
Human health (Ionising radiation)	kg ^{235}U eq.	0.01	0.01	0.05	0.01	0.01	0.06
Human health (Non-cancer)	CTUh	1.7E-07	0	3.9E-05	3.2E-07	0	5.9E-05
Ozone depletion	kg CFC11 eq.	3.0E-08	2.3E-08	6.0E-08	3.4E-08	2.5E-08	7.7E-08
Photochemical ozone formation	kg NMVOC eq.	0.0009	0.0008	0.0010	0.0012	0.0010	0.0022
Human health (Respiratory inorganics)	disease inc.	2.8E-08	2.4E-08	4.2E-08	2.2E-08	1.8E-08	3.9E-08
Dissipated water	m^3 eq.	0.14	0.12	0.17	0.13	0.11	0.16
Fossil resource depletion	MJ	3.5	3.1	4.3	3.9	3.5	4.9
Land use	Point	40	32	57	40	32	65
Mineral resource depletion	kg Sb eq.	2.0E-06	1.7E-06	4.4E-06	2.3E-06	1.9E-06	5.0E-06

Table A5.2.2. Uncertainty estimates of environmental footprints per kg of lettuce for domestic field wasted organic soil cultivation and supply chain scenarios. Shows deterministic values and the 95% simulation intervals of Monte Carlo simulations. $O_{WASTED\_UK\_2}$ models two crops per year.

Impact category	Unit	O_{WASTED_UK}			O_{WASTED_UK2}		
		Value	**2.5%**	**97.5%**	**Value**	**2.5%**	**97.5%**
Climate change	kg CO_2 eq.	0.88	0.70	1.14	0.58	0.49	0.72
Acidification (Terrestrial and Freshwater)	mol H^+ eq.	0.004	0.004	0.005	0.004	0.003	0.005
Freshwater ecotoxicity	CTUe	0.9	0	3.7	0.9	0	3.8
Freshwater eutrophication	kg P eq.	0.0001	0.0001	0.0003	0.0001	5.3E-05	0.0003
Marine eutrophication	kg N eq.	0.004	0.003	0.005	0.003	0.003	0.004
Terrestrial eutrophication	mol N eq.	0.018	0.016	0.023	0.015	0.012	0.018
Human health (Cancer)	CTUh	1.4E-08	0	4.8E-07	1.4E-08	0	4.73E-07
Human health (Ionising radiation)	kg ^{235}U eq.	0.01	0.01	0.07	0.01	0.01	0.07
Human health (Non-cancer)	CTUh	1.9E-07	0	4.8E-05	1.9E-07	0	4.9E-05
Ozone depletion	kg CFC11 eq.	3.5E-08	2.7E-08	7.5E-08	3.5E-08	2.8E-08	7.2E-08
Photochemical ozone formation	kg NMVOC eq.	0.001	9.4E-04	0.002	0.001	0.001	0.002
Human health (Respiratory inorganics)	disease inc.	3.8E-08	3.5E-08	5.4E-08	3.2E-08	2.9E-08	4.6E-08
Dissipated water	m^3 eq.	0.14	0.12	0.18	0.14	0.12	0.18
Fossil resource depletion	MJ	9.5			6.7	6.1	7.9
Land use	Point	62	47	91	39	31	56
Mineral resource depletion	kg Sb eq.	2.6E-06	2.2E-06	5.1E-06	2.6E-06	2.2E-06	5.3E-06

Table A5.2.3. Uncertainty estimates of environmental footprints per kg of lettuce for domestic field thicker organic soil cultivation and supply chain scenarios. Shows deterministic values and the 95% simulation intervals of Monte Carlo simulations.

Impact category	Unit	O_{THIN_UK}			O_{THICK_UK}		
		Value	**2.5%**	**97.5%**	**Value**	**2.5%**	**97.5%**
Climate change	kg CO_2 eq.	0.78	0.63	0.97	1.02	0.81	1.29
Acidification (Terrestrial and Freshwater)	mol H^+ eq.	0.004	0.004	0.005	0.005	0.004	0.006
Freshwater ecotoxicity	CTUe	0.9	0	3.5	0.9	0	3.7
Freshwater eutrophication	kg P eq.	9.6E-05	5.2E-05	0.0003	9.0E-05	4.8E-05	0.0003
Marine eutrophication	kg N eq.	0.004	0.003	0.005	0.004	0.003	0.005
Terrestrial eutrophication	mol N eq.	0.019	0.016	0.023	0.020	0.017	0.025
Human health (Cancer)	CTUh	1.3E-08	0	4.0E-07	1.3E-08	0	3.7E-07
Human health (Ionising radiation)	kg ^{235}U eq.	0.01	0.01	0.05	0.01	0.01	0.06
Human health (Non-cancer)	CTUh	1.5E-07	0	4.4E-05	1.6E-07	0	3.9E-05
Ozone depletion	kg CFC11 eq.	3.2E-08	2.5E-08	6.4E-08	2.9E-08	2.3E-08	6.0E-08
Photochemical ozone formation	kg NMVOC eq.	9.1E-04	8.3E-04	0.001	8.4E-04	7.5E-04	0.001
Human health (Respiratory inorganics)	disease inc.	2.6E-08	3.4E-08	5.2E-08	1.7E-08	3.6E-08	5.5E-08
Dissipated water	m^3 eq.	0.14	0.12	0.17	0.13	0.10	0.16
Fossil resource depletion	MJ	7.1	6.4	8.4	9.3	8.1	11.0
Land use	Point	39	31	56	40	32	56
Mineral resource depletion	kg Sb eq.	2.3E-06	2.0E-06	4.9E-06	1.8E-06	1.6E-06	3.8E-06

Table A5.2.4. Uncertainty estimates of environmental footprints per kg of lettuce for imported field mineral soil cultivation and supply chain scenarios. Shows deterministic values and the 95% simulation intervals of Monte Carlo simulations.

Impact category	Unit	M_{NE}			M_{SE}			M_{USA}		
		Value	**2.5%**	**97.5%**	**Value**	**2.5%**	**97.5%**	**Value**	**2.5%**	**97.5%**
Climate change	kg CO_2 eq.	0.35	0.30	0.47	0.45	0.37	0.66	4.02	2.91	5.79
Acidification (Terrestrial and Freshwater)	mol H^+ eq.	0.003	0.003	0.005	0.008	0.004	0.019	0.022	0.017	0.035
Freshwater ecotoxicity	CTUe	1.2	0	4.0	1.6	0	4.7	1.9	0	6.0
Freshwater eutrophication	kg P eq.	9.3E-05	4.9E-05	3.4E-04	1.1E-04	5.9E-05	3.1E-04	1.4E-04	7.4E-05	4.6E-04
Marine eutrophication	kg N eq.	0.003	0.002	0.004	0.004	0.002	0.011	0.009	0.007	0.013
Terrestrial eutrophication	mol N eq.	0.014	0.011	0.019	0.033	0.017	0.083	0.089	0.067	0.138
Human health (Cancer)	CTUh	1.5E-08	0	4.3E-07	1.5E-08	0	5.3E-07	2.5E-08	0	1E-06
Human health (Ionising radiation)	kg ^{235}U eq.	0.02	0.01	0.09	0.03	0.02	0.11	0.25	0.12	0.67
Human health (Non-cancer)	CTUh	1.9E-07	0	4.5E-05	2.0E-07	0	6.1E-05	7.8E-07	0	1.1E-04
Ozone depletion	kg CFC11 eq.	5.3E-08	3.8E-08	1.4E-07	7.9E-08	5.1E-08	2.2E-07	8.9E-07	4.8E-07	2.6E-06
Photochemical ozone formation	kg NMVOC eq.	0.001	0.001	0.002	0.002	0.001	0.003	0.021	0.016	0.034
Human health (Respiratory inorganics)	disease inc.	3.7E-08	3.2E-08	5.8E-08	7.5E-08	4.9E-08	1.6E-07	6.2E-08	5.2E-08	1.0E-07
Dissipated water	m^3 eq.	0.15	0.12	0.18	0.15	0.13	0.19	0.20	0.17	0.27
Fossil resource depletion	MJ	5.0	4.3	7.1	6.2	5.0	9.3	55.4	40.1	80.4
Land use	Point	43	33	61	40	32	63	33	24	65
Mineral resource depletion	kg Sb eq.	2.2E-06	2.0E-06	4.6E-06	2.5E-06	2.2E-06	4.8E-06	3.1E-06	2.7E-06	6.2E-06

Table A5.2.5. Uncertainty estimates of environmental footprints per kg of lettuce for domestic and imported protected cultivation and supply chain scenarios. Shows deterministic values and the 95% simulation intervals of Monte Carlo simulations.

Impact category	Unit	P_{UK}			P_{NE}			P_{SE}		
		Value	2.5%	97.5%	Value	2.5%	97.5%	Value	2.5%	97.5%
Climate change	kg CO_2 eq.	0.89	0.71	1.31	0.99	0.80	1.43	0.53	0.44	0.77
Acidification (Terrestrial and Freshwater)	mol H^+ eq.	0.003	0.003	0.004	0.004	0.003	0.005	0.003	0.003	0.005
Freshwater ecotoxicity	CTUe	1.5	0	5.4	1.9	0	6.6	1.9	0	6.0
Freshwater eutrophication	kg P eq.	1.7E-04	1.0E-04	5.0E-04	1.8E-04	1.0E-04	5.4E-04	1.5E-04	9.2E-05	4.8E-04
Marine eutrophication	kg N eq.	0.002	0.001	0.002	0.002	0.001	0.002	0.002	0.002	0.003
Terrestrial eutrophication	mol N eq.	0.009	0.008	0.013	0.011	0.009	0.015	0.012	0.010	0.017
Human health (Cancer)	CTUh	5.9E-08	0	1.1E-06	6.1E-08	0	1.2E-06	4.1E-08	0	9.5E-07
Human health (Ionising radiation)	kg ^{235}U eq.	0.02	0.01	0.10	0.03	0.02	0.13	0.04	0.02	0.17
Human health (Non-cancer)	CTUh	1.2E-07	0	1.2E-04	1.35E-07	0	1.2E-04	1.4E-07	0	9.7E-05
Ozone depletion	kg CFC11 eq.	8.7E-08	6.8E-08	1.9E-07	1.1E-07	8.2E-08	2.4E-07	8.6E-08	5.9E-08	2.4E-07
Photochemical ozone formation	kg NMVOC eq.	0.002	0.002	0.004	0.003	0.003	0.005	0.002	0.002	0.004
Human health (Respiratory inorganics)	disease inc.	3.7E-08	3.3E-08	1.5E-07	4.6E-08	4.2E-08	1.4E-07	4.9E-08	4.2E-08	8.1E-08
Dissipated water	m^3 eq.	0.20	0.16	0.24	0.21	0.17	0.25	0.21	0.17	0.27
Fossil resource depletion	MJ	13.3	10.5	18.8	14.9	12.1	21.0	9.8	8.3	14.4
Land use	Point	25	20	36	28	22	42	31	24	50
Mineral resource depletion	kg Sb eq.	5.1E-06	4.7E-06	8.0E-06	5.3E-06	4.9E-06	8.2E-06	3.8E-06	3.1E-06	8.2E-06

Table A5.2.6. Uncertainty estimates and environmental footprints per lettuce plug for plug production scenarios. Shows deterministic values and the 95% simulation intervals of Monte Carlo simulations.

Impact category	Unit	UK (and Northern Europe)			Southern Europe		
		Value	2.5%	97.5%	Value	2.5%	97.5%
Climate change	kg CO_2 eq.	0.006	0.005	0.007	0.004	0.003	0.007
Acidification (Terrestrial and Freshwater)	mol H^+ eq.	5.2E-05	4.5E-05	8.4E-05	4.2E-05	3.0E-05	8.8E-05
Freshwater ecotoxicity	CTUe	0.01	0	0.05	0.01	0	0.04
Freshwater eutrophication	kg P eq.	1.7E-06	9.5E-07	4.4E-06	1.1E-06	6.3E-07	3.5E-06
Marine eutrophication	kg N eq.	2.9E-05	2.1E-05	4.5E-05	2.7E-05	1.5E-05	6.1E-05
Terrestrial eutrophication	mol N eq.	1.8E-04	1.4E-04	3.5E-04	1.6E-04	1.0E-04	3.9E-04
Human health (Cancer)	CTUh	6.1E-10	0	1.5E-08	3.4E-10	0	1.0E-08
Human health (Ionising radiation)	kg ^{235}U eq.	4.0E-04	2.3E-04	0.002	3.6E-04	2.1E-04	0.002
Human health (Non-cancer)	CTUh	6.6E-10	0	1.8E-06	4.1E-10	0	1.2E-06
Ozone depletion	kg CFC11 eq.	1.1E-09	8.4E-10	2.0E-09	9.8E-10	6.1E-10	2.3E-09
Photochemical ozone formation	kg NMVOC eq.	2.8E-05	2.2E-05	6.4E-05	2.1E-05	1.5E-05	6.9E-05
Human health (Respiratory inorganics)	disease inc.	4.8E-10	4.1E-10	1.5E-09	3.4E-10	2.6E-10	6.6E-10
Dissipated water	m^3 eq.	0.003	0.002	0.003	0.002	0.002	0.003
Fossil resource depletion	MJ	0.33	0.28	0.41	0.32	0.27	0.42
Land use	Point	0.08	0.03	0.11	0.06	0.03	0.11
Mineral resource depletion	kg Sb eq.	7.7E-08	6.8E-08	1.3E-07	3.9E-08	2.9E-08	1.2E-07