Applying the natural capital approach to farm-scale land management decision-making and evaluation

Exploring the impacts of management intensity and organic agriculture on natural capital and ecosystem services

Submitted by Matthew Frederick Holden to the University of Exeter as a thesis for the degree of Doctor of Philosophy in Geography, January 2022

Abstract

The natural capital (NC) approach presents a structured framework for sustainable decision-making and evaluation, requiring an understanding of how different decisions impact NC and the flow of multiple ecosystem services (ES). The approach has been placed at the heart of delivering the UK Government's 25 year Environment Plan, which states their intention to "set gold standards in protecting and growing natural capital – leading the world in using this approach as a tool in decision-making".

There is now growing advocacy for its incorporation into local-scale land management decision-making (e.g. individual farm or estate businesses). Despite this growing interest, evidence of its application at the farm scale is limited. Existing studies have often only partially applied the approach and nearly always rely on existing data (irrespective of its suitability at local scales), modelled data or data from other studies. Previous research has suggested that failing to underpin the approach with site-specific, fit-for-purpose, data brings into question its usefulness in decision-making and evaluation at the local scale.

The research in this PhD represents one of the first attempts to implement a complete application of the NC approach, including detailed measurement of NC condition, ecosystem function (EF) and ES value at the farm scale. The study focuses on four ES pathways – climate regulation, food production, drinking water provision and pollinator services – in the context of land management decisions on the Clinton Devon Estate in Devon. Its core contributions are both methodological and empirical; it explores how the NC approach can be applied robustly at the farm scale and how the adoption of different land management practices, including organic agriculture and intensive farm management, impact NC and ES. The key findings are that: 1.) there are a number of significant challenges that need to be addressed before the NC approach will be practical in routine farm-management decision-making (e.g. availability of suitable data, access to expertise), 2.) land management intensity can degrade soil NC presenting on-going risks to future soil condition in the UK and 3.) organic farming has the capacity to increase soil carbon storage, enhance pollinator stocks and improve the supply of clean drinking water whilst delivering similar producer welfare compared to conventional farming.

Acknowledgements

The completion of this PhD was made possible by the support of a number of individuals and organisations to whom I am very grateful.

I would first like to thank my academic supervisors Prof. Richard Brazier and Prof. Brett Day from whom I learnt a great deal during my PhD. I would also like to thank my industry supervisors Dr Sam Bridgewater (Clinton Devon Estate) and Yog Watkins (Westcountry Rivers Trust) for their enthusiastic support on the ground and valuable insights into farm and estate management.

I would like to extend my thanks to Clinton Devon Estate for supporting the funding of the research and to their estate and farm staff for being so welcoming and encouraging throughout. A special thanks goes to farm managers Sam Briant-Evans and Barney Tremaine for their exchange of farming knowledge and advice during this time. This research would have been impossible without the support of estate tenants and their staff, for which I am very grateful - thank you Alan, Jerry, Chris, James, Robin, Pete and Di. I would also like to acknowledge all of the estate volunteers, with whom it was a pleasure to work on the bumblebee surveys across the estate – thank you for your dedicated assistance.

Thank you to Marcus Adams and Nik Whalley at South West Water for supporting the nitrate leaching work through funding the porous pot installation, providing invaluable advice on groundwater in the Lower Otter and enabling access to the appropriate data. Staff at Wessex Water were also integral to the development of this work and I would like to thank Tim Stephens, Dr Sophie Sherriff and the rest of their team for getting me up to speed with porous pot monitoring and running the IRRIGUIDE model simulations. I am also extremely grateful to members of the University of Exeter Bee Group (chaired by Prof. Juliet Osbourne). In particular thank you to Dr Ben Phillips for showing me the ropes of flower counts and to Dr Grace Twiston-Davies and Dr Matthias Becher for going above and beyond to help refine the application of the BEE-STEWARD software for my purposes.

A huge thanks goes to colleagues in the Brazier Research Group. There are too many names to list but I am extremely grateful for all of your input and advice. You kept me sane throughout the PhD and for that I will forever be grateful. A special mention to Paul Henderson (Seal King), Dr Dave Luscombe, Hugh

Graham, Dr Pia Benaud and Chryssa Brown for your specific inputs to the PhD. Thank you also to the Geography Department Lab Technicians team, with special mention to Angela Elliott, Dr Joana Zaragoza-Castells and Neville England. When I say I couldn't have done this without you all – I really mean it!

I would finally like to thank all of my friends and family who have supported me over the past four years. I am hugely grateful to those I have shared this PhD journey with; Roger Auster, Nicola Ellis, Rachel Manning and Oana Stefancu. Thank you for all the laughter, post-work swims and picnics on the beach – we all made it in the end! A massive thank you also to my brother and Ciara, Mum, Dad and Ali for being patient with me when I was pulling my hair out and for the many motivational pep talks.

Table of Contents

List of tables

List of figures

Definition of terms

Definition of terms and acronyms frequently used within the thesis.

Economic terms:

Natural capital (NC): The stock of renewable and non-renewable resources (e.g. plants, animals, air, water, soils, minerals) that combine to yield a flow of benefits to people (Natural Capital Coalition, 2017).

Natural capital approach: Defined here as a systematic approach to decision-making and evaluation involving assessment of three key tiers; Tier 1, understanding the extent and condition of natural capital; Tier 2, understanding the ecosystem functions that play a critical role (often alongside human, social, manufactured and other capital) to transform natural capital in a given state into benefits to humans; Tier 3, understanding the economic value of the flows of ecosystem services derived from natural capital.

Ecosystem functions (EF): Sometimes also referred to as environmental or ecosystem processes or intermediate ecosystem services these functions are those which contribute to the transformation of NC into ecosystem services (e.g. carbon cycling and storage, nutrient retention and cycling, storage and degradation of pollutants, pollination, natural pest control).

Ecosystem services (ES): The direct benefits that people derive from ecosystems (e.g. food, fibre, clean drinking water, climate regulation, recreational enjoyment).

Intermediate ecosystem goods and services: Intermediate goods or processes that facilitate the creation/maintenance of a good or service that supports the creation of a final product (e.g. habitat generation and maintenance to sustain crop growth).

Final ecosystem goods and services: A good or service that has a direct benefit to humans/society (e.g. crops for food consumption).

Economic value: Defined here as a monetary measure of an individual/household/firm's welfare gain. Value in the context of economics is a measure of welfare and is based on an individual's understanding of the benefit of accessing a good or service. An understanding of how much someone values something can be derived from their interest in exchanging something for it. As money is typically used as a currency in the exchange of goods and services it is commonly applied as a proxy for economic value.

Producer surplus: A measure of producer benefit from the production of goods (e.g. crops). Calculated as revenue from product sales minus variable production costs.

Farm scale: Used throughout the thesis to refer to a range of agricultural business structures from the small farm to the large estate with multiple farm tenants. The term is used to cover organisational units (i.e. a farm business or estate business) that typically make land management decisions. These decisions might require information at a range of resolutions. Depending on the nature of the decision this could include understanding and detecting differences in NC, EF and/or ES at the following resolutions: within field, between different fields, between groups of fields (e.g. at stages of the rotation), between farm habitats or between whole farms.

Sustainable agriculture: Sustainable agriculture or the sustainability of agricultural practices is defined following Tilman *et al.,* (2002) as practices which meet current and future societal needs for food and fibre, ecosystem services and for healthy lives and do so by maximising the net benefit to society, when all economic, environmental and social costs and benefits of the practices are considered.

Natural science terms:

Bulk density (BD): The mass of a unit volume of dry soil, usually expressed as g soil cm^{-3}.

Soil organic carbon (SOC): Percentage of organic carbon contained within the soil

Soil organic matter (SOM): Percentage of organic matter contained within soil (e.g. as plant residues, root exudates and microbial biomass). Typically measured through loss on ignition (LOI).

Total carbon (TC): Percentage of total carbon (including organic and inorganic carbon fractions) contained within soil

Total nitrogen (TN): Percentage of total nitrogen (including organic and inorganic nitrogen) contained within soil

Bioavailable phosphorus (P): The quantity of bioavailable phosphorus as phosphate contained within soil, typically measured through the Olsen-P extraction process. P is a very important plant nutrient.

Bioavailable magnesium (Mg): The quantity of bioavailable magnesium contained within soil. Mg is an important plant nutrient.

Bioavailable potassium (K): The quantity of bioavailable potassium contained within soil. K is an important plant nutrient.

Soil pH: A measure of soil acidity or alkalinity, important in influencing soil chemistry and the availability of soil nutrients.

N-potential: Used to refer to the ratio of clay to soil organic carbon (clay:SOC) with categorisation based on the capacity for a soil to store more carbon and soil stability (i.e. its resistance to soil erosion), as described in Merante *et al.,* (2017).

Chapter 1: Introduction, objectives and thesis structure

1.1 Introduction

Agricultural intensification through the 20th century, focused almost single-mindedly on increased crop production, has had a significant impact on natural capital (NC) (e.g. soil, water, ecosystems) and the flow of ecosystem services (ES) (e.g. clean air and water, climate regulation, recreational enjoyment) that NC provides. In recognition of these issues, the UK government have set out ambitious targets through their 25 Year Environment Plan (HM Government, 2018) to deliver net improvements in England's environment within a generation. The importance of protecting NC is at the heart of the plan, which identifies the benefits NC delivers to the economy and to human welfare. Integral to the delivery of their ambitions is the application of the NC approach to decision-making. Indeed, the government state their intention to "set gold standards in protecting and growing natural capital – leading the world in using this approach as a tool in decision-making" (HM Government, 2018, pg. 9).

The NC approach presents a structured framework for understanding the impact of decisions on NC and the subsequent flows of ES. Three stages or tiers are commonly considered in frameworks relating to the approach: Tier 1) an assessment of NC extent and condition (e.g. soil or water quality): Tier 2) measuring the response in ecosystem function (EF) (sometimes called environmental processes) (e.g. carbon sequestration, crop growth, nutrient filtering); and Tier 3) the valuation of ES (e.g. climate regulation, crop and drinking water provision) (Haines-young and Potschin, 2008; Dominati, Patterson and Mackay, 2010; Faccioli *et al.*, 2020). The approach supports the principles that decisions should not be single-focused (say on profits from food production) but should consider the full range of benefits derived from the natural environment. It provides a framework for decision makers to account for multiple market (e.g. crops, timber and water) and non-market (e.g. climate regulation and flood alleviation) goods and services in their decisions (Hanley *et al.*, 2015; Bateman and Mace, 2020; Ovando, 2021). Economists have long recognised the NC approach as a useful model by which to account for sustainability (Maseyk *et al.*, 2017) but interest in its application to decision-making has grown in recent years.

Whilst the NC approach has initially been applied at relatively large spatial scales, there is growing advocacy for its incorporation into local-scale land management decision-making and evaluation (Faccioli *et al.*, 2020). Organisations such as the Natural Capital Committee and Natural Capital Coalition have actively encouraged its adoption at local and organisational scales in recent years (e.g. for individual farm or estate businesses) (Natural Capital Coalition, 2017; Natural Capital Committee, 2017). This is borne out of a recognition that most NC assets are owned and/or managed by organisations working at much smaller scales (Faccioli *et al.*, 2020). In the UK, 70% of land is managed as farmland (Connors, 2016) and in England, around 92% is privately owned (Shrubsole, 2019). Meaningful change in the condition of the UK's NC will only be realised, therefore, by facilitating changes in decision-making by private landowners, likely to be operating at the farm or estate scale.

The promotion of the application of the NC approach at the farm or estate scale has not been missed by the farming community. There has been a rush of recent articles in the farming media on the potential of incorporating the approach into farm decision-making (EFTEC, 2019; Beedell, 2021; CLA, 2021; Harris, 2021; Norton, 2021) – recognising its potential to identify private and public benefits that arise from NC, support sustainable decision-making and access future funding steams. However, despite positivity around the approach, its successful implementation at the farm scale remains limited. Studies that have attempted to apply the NC approach (see section 2.1.5) at the farm or estate scale have often been partial (i.e. they have not completed all tiers of the approach). Most local scale studies have either failed to adequately measure NC condition (only accounting for asset extent) (e.g. Faccioli *et al.*, 2020), omitted the monitoring of EF or have stopped short of valuing ES (e.g. Ovando, 2020). Furthermore, they have nearly always relied on existing readily available data (irrespective of whether it is fit-for-purpose), modelled data or data derived from other studies (i.e. using value transfer methods) (EFTEC, 2018; Kieboom, Silcock and Russ, 2018; Silcock and Russ, 2018; Faccioli *et al.*, 2020). Whilst conducting this work is no doubt challenging, failure to conduct the complete NC approach and/or omitting to underpin it with site specific, high resolution data brings into question its usefulness in decision-making and evaluation at management appropriate scales (Faccioli *et al.*, 2020).

There therefore remains a significant need to build understanding on the application of the NC approach at the relevant decision-making scale, that of the land manager. In particular, it is important to understand how primary, site-specific data on NC condition, EF and ES values can be used to improve the method's usefulness at management-appropriate scales. This undertaking is the focus of theme one of this PhD which applies stages of the NC approach to the evaluation of land management decisions at Clinton Devon Estate, South West England. The PhD builds towards the complete application of the approach, combining information on NC condition and EF to facilitate the valuation of ES. In doing so, it aims to contribute learning on the practical application of the approach at the farm scale, highlighting the challenges and opportunities for its future application.

In building towards this aim, the PhD contributes empirical insights on a second theme, the impact of specific land management practices on NC, EF and ES. The primary focus was on determining the capacity for organic agriculture to enhance NC and the flow of ES. A secondary line of investigation considered the implications of farm management intensity (and associated management practices) on NC at Clinton Devon Estate. The findings from both have important policy implications at a time when the UK Government are reassessing the manner and level of reward that farmers should receive for the delivery of public goods through their new environmental land management scheme (ELMS).

The overarching research objectives are expanded on below followed by a description of the thesis structure.

1.2 Research aims and objectives

The research was conducted in collaboration with two main industry partners, Clinton Devon Estate and Westcountry Rivers Trust. It also required engagement and support from other important stakeholders – notably, farm tenants and South West Water. The main aims of the PhD were to advance understanding on the application of the NC approach at the farm scale[1] and to contribute to scientific understanding on the impacts of different land management practices on NC, EF

[1] The term 'farm scale' is used through the PhD to refer to a range of agricultural business structures from a small farm to a large estate with multiple farm tenants. Farm scale decisions require information at a range of resolutions. Depending on the nature of the decision this could include understanding NC, EF and ES at the resolution of within field, field, group of fields (e.g. at stages of the rotation), habitat block or across the whole farm(s).

and ES. In doing so, the PhD sought to collect high-resolution data that could be used by Clinton Devon Estate to inform future farm and estate management decisions. Information that was of particular interest to the estate included: 1) how baseline NC conditions should be established and monitored over time; 2) the impact of different land management intensity on NC condition across the tenanted and in-house farms; and 3) the role that organic conversion of the Home Farm (farmed in-hand by Clinton Devon Estate) in 2007 has had on NC and the flow of ESs. There was a keen interest in the research being interdisciplinary, focusing on aspects of soil, water and biodiversity NC and the connected flows of ES.

Through a process of co-creation with the project partners and underpinned by the literature (Chapter 2) four overarching objectives were established. These objectives are outlined below and are addressed to differing degrees across four primary research chapters (chapters 4 to 7). Here the relative contribution each chapter makes to tackle the objective is summarised while each chapter is introduced in more detail in the following section (Section 1.3).

Objective one: To establish baseline natural capital conditions for soil, water and biodiversity natural capital at the farm scale

This is addressed across chapters 4, 6 and 7. Chapter 4 focuses on establishing baseline conditions for soil NC, Chapter 6 for pollinator NC (a measure of functional biodiversity) and Chapter 7 considers those two natural capitals alongside an assessment of baseline groundwater NC condition.

Objective two: To build understanding on how land management practices and intensity impact on natural capital condition and productive output

This is addressed in Chapter 4 which uses measurements of soil NC to understand the impacts of farm management intensity and soil management practices on soil NC condition. It also investigates the relationship between soil NC condition and crop production to evaluate the private implications of degrading soil NC.

Objective three: To explore the capacity for organic agriculture to balance food production, producer welfare and the enhancement of natural capital and ecosystem service delivery

This is addressed in Chapter 5, 6 and 7. Chapter 5 evaluates the detailed impacts of organic conversion at the estate in 2007 on soil NC condition and soil function. Chapter 6 investigates the impacts of organic conversion on pollinator stocks alongside trade-offs in yield and consequences for farm profitability. Finally, Chapter 7 brings in information on groundwater NC and drinking water provision, evaluating the economic costs and benefits associated with conversion to organic agriculture.

Objective four: To undertake a complete application of the natural capital approach (from measurement of natural capital condition through to economic valuation of ecosystem services) using field-based data and, by so doing, build understanding as to how the approach might be implemented at the farm scale and assess whether it is suitable for routine land management decision-making at that scale.

This objective is the focus of Chapter 7, which builds upon data reported in chapters 5 and 6. It addresses the objective in the context of a decision to adopt organic farming while taking account of four ecosystem service pathways impacted by that change: soil carbon storage and climate regulation, crop growth and producer benefits, nitrate leaching and drinking water provision and pollinator stocks and pollinator services.

1.3 Thesis structure

This thesis contains a review of the literature (Chapter 2), a site description and introduction to methods (Chapter 3), four primary research chapters (chapters 4 – 7) and a final synthesis chapter (Chapter 8).

The literature review in Chapter 2 places the research in context and acted as the foundational base for developing the primary research chapters. Firstly, it introduces the NC approach and examines current applications of the natural approach at the farm scale. It then explores the important forms of NC, EF and ES in agricultural environments, identifying how they are typically measured. Finally, the review introduces the issues associated with intensive agriculture, the changing state of agricultural policy in the UK and the challenge and opportunities for agricultural systems (such as organic agriculture) to balance food production and the delivery of other ES.

Chapter 3 provides a more detailed introduction to the case study site Clinton Devon Estate, the important forms of NC on that estate and the associated flows of ES. It introduces the measurements of NC, EF and ES applied in this study and places them within the NC approach framework, highlighting the wider relevance of each based on the literature. The chapter finishes by providing an overview of some of the methods that are frequently applied in two or more of the primary research chapters and require further detailed explanation. This information reduces repetition in the primary research chapters.

The following four research chapters are presented in the format of self-contained papers to facilitate on-going publication. As such, there is some repetition across the chapters. Attempts have been made to minimise repetition and where there is overlap in methods, the reader is referred to the relevant section in Chapter 3. A brief overview of each chapter follows.

Chapter 4: Utilising existing land management records to explore the drivers of soil natural capital condition and productive output across different management intensities

This chapter contributes to objective one and addresses objective two. The chapter reports on research activities that established baseline soil NC conditions and goes on to demonstrate how this data can be used alongside farm management records to generate useful information that can inform farm and estate management decisions. The chapter shows that baseline NC data can be used alongside field management records to investigate the impact of land management intensity and identify potential management practices (e.g. tillage, nutrient inputs or rotation) that might be driving soil NC condition. Furthermore, measurements of soil NC condition and farm records can be combined with crop yield data, to determine the impact that NC condition can have on productive output.

The chapter tackles the following research questions:

1. How does farm intensity impact on soil natural capital condition and productive output?
2. What are the likely drivers of differences in soil natural capital condition across the study farms?
3. Does degradation of soil natural capital impact on productive output?

Chapter 5: Does conversion to organic farming improve soil natural capital condition and soil function?

This chapter contributes to address objective three. The aim of the chapter is to evaluate the role that organic agriculture could play in enhancing soil NC condition and, importantly, soil function. In addition, the study seeks to identify, which, if any, of the diversity of soil condition indicators appraised in the research meaningfully, inform the quantification of final soil-based ES. It tackles the following two research questions:

1. Do organic field sites have better NC condition than conventional sites and what practices might explain any differences?
2. Do organic field sites have enhanced soil function compared to conventional sites and what practices might explain any differences?

Chapter 6: Organic conversion and long-term pollinator stocks: A landscape-scale analysis using the BEE-STEWARD software

This chapter contributes to objective three. The aim of the study was to build understanding on the impact of organic conversion and/or habitat interventions on bumblebee populations (as an indicator of insect pollinators), evaluating the potential trade-offs or 'win-wins' between pollinator NC stocks and provisioning services (crop production and producer welfare). The study had three key objectives:

1. To apply the BEE-STEWARD modelling software to answer the following questions:
   A. Does a landscape-scale shift from conventional to organic agriculture enhance floral resources available to insect pollinators?
   B. Does the addition of pollen and nectar habitat interventions enhance floral resources within conventional and organic dominated landscapes?
   C. Does a landscape-scale shift from conventional to organic agriculture enhance long-term bumblebee populations?
   D. Do pollen and nectar habitat interventions enhance long-term bumblebee populations at a landscape scale?
2. To quantify the trade-offs in yield and the returns to farming associated with:

A. A landscape scale shift from conventional to organic agriculture
B. The addition of pollen and nectar habitat interventions in the farmed landscape

3. To estimate the cost-effectiveness of different land management changes in enhancing pollinator stocks, tackling the question: What is the most cost-effective strategy to increase bumblebee populations?

Chapter 7: A systematic application of the natural capital approach at the farm scale: Is it a practical tool for routine land management decision-making?

This chapter contributes to objective three, whilst addressing objective four. It seeks to deliver a complete application of the NC approach combining natural sciences data from chapters 5 and 6 with the economic valuation of a suite of ES. The chapter had two aims: 1) to build understanding around the challenges and opportunities of applying the NC approach at farm management scales; and 2) To evaluate whether organic agriculture delivers greater benefits to society (i.e. higher ecosystem service value) than conventional agriculture. The chapter tackles the following research questions:

1. What are the data and science requirements of the NC approach when applied at the farm scale? Do these requirements make it practical for routine use in farm-management decision-making?
2. Given the costs and complexities of the full NC approach, can we rely simply on biophysical measurements of NC and EF to assess the likely scale of ES values delivered by farm management decisions?
3. When applying the NC approach, can conversion to organic agriculture deliver greater benefits to human society (ecosystem service value) than conventional agriculture?

In Chapter 7 there is some repetition from earlier chapters. This is inevitable as it brings together the research from the previous chapters but has been written so that it can be reproduced as a standalone paper.

To bring the thesis to conclusion, Chapter 8 provides a synthesis of the findings from the four primary research chapters. It goes on to provide some key conclusions and identify future research to build upon the work conducted here.

Chapter 2: Literature review

2.1 The natural capital approach to agricultural decision-making

2.1.1 Defining natural capital and ecosystem services

The Natural Capital Coalition define NC as “the stock of renewable and non-renewable resources on earth (e.g. plants, animals, air, water, soils and minerals) that combine to yield a flow of benefits or “services” to people” (Natural Capital Coalition, 2021). It is typical in the literature for NC and ES to be used alongside each other, with ES considered to flow from NC, as in the definition above. ES are defined in the Millennium Ecosystem Assessment (MEA, 2005) as the “benefits people derive from ecosystems” and Costanza *et al.,* (1997) link the two through the description of ES as the “flows of materials, energy, and information from natural capital stocks which combine with manufactured and human capital services to produce human welfare” (Pg. 254).

Different distinctions have been made between the different types of ES considered to flow from NC. The MEA (2005) assessment identified four types of ES:

- Provisioning services: e.g. provision of food, fibre and raw materials
- Regulating services: e.g. flood alleviation or climate regulation
- Cultural services: e.g. spiritual or recreational landscapes
- Supporting services: e.g. soil formation, nutrient cycling or erosion control).

Others have considered two distinctions in ES, describing “final ES”, providing a direct service or good to society (e.g. fish for consumption) and “intermediate ES”, facilitating the creation/maintenance of a good or service that supports the creation of a final product (e.g. habitat generation and maintenance to sustain fisheries) (Guerry *et al.,* 2015). Some workers have deliberately excluded “supporting” or “intermediate” services in their frameworks for assessing ES, considering them to be ecosystem processes or functions and not ES which deliver a direct and quantifiable benefit to humans (Dominati *et al.,* 2010; Maseyk *et al.,* 2017). The same approach is applied in this study and reference to ES herein relates to final ecosystem goods and services that deliver a direct benefit to humans.

There are different beneficiaries from different ES and ES can be distinguished as delivering private and/or public benefits. Bateman and Balmford (2018) define a public good as being both “non-excludable (i.e. non-paying consumers cannot be prevented from accessing it) and non-rivalrous (i.e. use by one individual does not reduce availability to others)” (e.g. the air we breathe) (pp. 295). In contrast a private good is defined as being both “excludable (i.e. consumers have to pay to access it) and rivalrous (i.e. use by one individual precludes its use by another)” (e.g. food produced by farmers) (Bateman and Balmford, 2018; pp. 295). In the production of private goods there are often externalities that arise which do not directly impact the producer but have impacts on other members of society (e.g. agricultural production leading to the pollution of drinking water supplies downstream). These externalities can be defined as any action that impact the welfare of or opportunities available to an individual or group without direct payment or compensation (Pretty *et al.*, 2003). Some externalities can also be positive. For example, progressive changes in agricultural land management could lead to positive external benefits such as reducing water resource pollution and reducing costs of drinking water treatment (a benefit to the water company) or improved carbon sequestration (a benefit to society due to climate regulation).

2.1.2 What is the natural capital approach?

The NC approach presents a systematic means of looking at the environment from an economic perspective focusing on the value that NC stocks provide to society through flows of ES (CCI, 2016). It involves understanding, measuring and assigning values to these relationships and by doing so presents a method to integrate values derived from nature into decision-making (CCI, 2016).

The NC approach arises from the premise that stocks of NC can be replenished or degraded. The quality and quantity of these stocks (NC condition), along with various anthropogenic drivers and environmental processes (defined here as ecosystem functions - EF), underpins the delivery of goods, services and disservices (ES) that directly impact on the welfare of individuals in society. There are a number of different stages in the NC approach (referred herein as tiers). Whilst the definition of these tiers might vary slightly between studies there is broad agreement that the NC approach involves building a systematic understanding of (Faccioli *et al.*, 2020):

- The extent and condition of NC assets or stocks (Tier 1)

- The environmental pathways (EF) through which changes in NC result in changes in ES (Tier 2)
- The economic value of the flows of ES (Tier 3)

Compiling this information enables the benefits derived from NC to be incorporated into decision-making. It allows an assessment of how changes to NC (extent and quality) following a change in asset management are likely to cascade into changes in ES value (Faccioli *et al.*, 2020).

A number of similar cascade frameworks have been presented that apply this thinking, conceptually linking NC condition to EF and the delivery of ES (Haines-young and Potschin, 2008; Dominati, Patterson and Mackay, 2010; Maseyk *et al.*, 2017). As explained above the components of these frameworks (Figure 2.1) comprise of assessing NC stocks (Tier 1), EF (Tier 2) (sometimes called processes) and ES (Tier 3). Ultimately the complete application of the NC approach aims to compile the necessary data on NC and EF to facilitate the valuation of ES. The goal is to conduct a holistic appraisal of the multiple ES benefits and trade-offs that arise from a land management decision, including both private goods (e.g. food and water) and public goods and services (e.g. climate regulation and recreation).

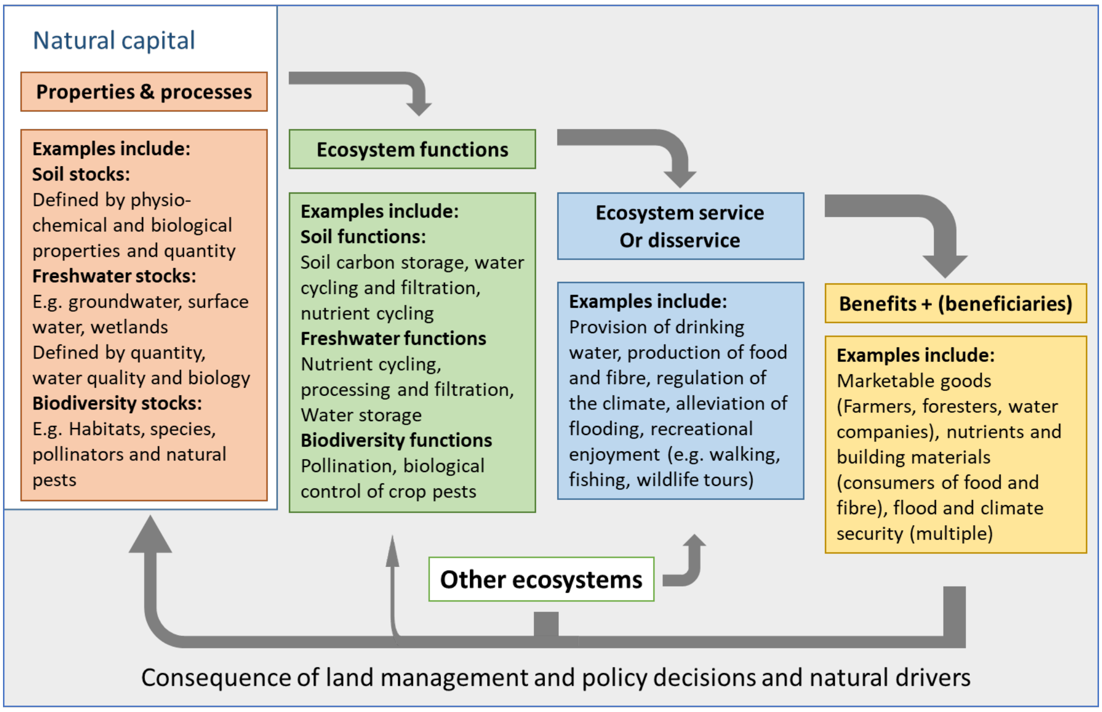


Figure 2.1: Showing the cascade framework applied to understand the connecting tiers in the NC approach. Adapted from Haines-young and Potschin (2008). The cascade framework shows the three tiers and how they connect to delivering benefits to people in society. Tier One shows examples of NC, Tier two shows EFs important in transforming NC into ES and Tier Three shows examples of final ES. The different benefits and beneficiaries are also shown to highlight who stands to benefit from ecosystem service flows.

The benefits and trade-offs associated with ES are commonly expressed in monetary terms (ES values) (Faccioli *et al.*, 2020). Using monetary terms facilitates the application of cost-benefit analysis as a decision-support tool. That is, ES values (e.g. climate regulation or clean drinking water) can be considered in the same units as the financial costs (e.g. agri-environment scheme payments) and the economic returns (e.g. farm profits from crop production) of different land management scenarios. In doing so, supporters of the NC approach suggest that it can be used to select land management practices that are both cost-effective and maximise the output of benefits to humans (Maseyk *et al.*, 2017; EFTEC, 2019; Bateman and Mace, 2020; Defra, 2020a).

The NC approach builds on the earlier ES approach, in that it focuses on understanding NC stocks (NC condition, quantity and sustainability) as well as the flows of ES and the social and economic benefits they provide (Natural Capital Coalition, 2020; Judd and Lonsdale, 2021). The ecosystem services approach or ecosystem approach is variably defined (Judd and Lonsdale, 2021) but in most simple terms it focuses on understanding the flows of goods and services and how they are impacted by a decision (Natural Capital Coalition, 2020). Advocates of the NC approach highlight that existing ecosystem approaches have often failed to adequately link the ES identified to the NC that underpin them (Maseyk *et al.*, 2017). This can be important as ES are not directly influenced by management (as shown in Figure 2.1) but are the consequence of change to NC condition and EF. Understanding the changes in NC condition and/or EF in response to land management change are therefore important in understanding the consequences to ES. Maseyk *et al.,* (2017) suggest that "by focusing on changes to natural capital stocks, consequences for ecosystem services can be inferred and effectiveness of policies assessed". Furthermore, they argue that implementation of an ES approach without "explicit reference" to NC stocks is unlikely to be successful. Similar views have been expressed by other advocates of the NC approach (Robinson *et al.,* 2009; Dominati *et al.,* 2010). The NC approach and ES approach therefore are not identical but interact (Maseyk *et al.*, 2017; Judd and Lonsdale, 2021) and both have been incorporated into decision-making frameworks.

2.1.3 How have land management decisions been evaluated in the past?

The NC approach is a novel way of evaluating the different land management decisions in that it combines both data from the natural sciences and from economics (Faccioli *et al.*, 2020). In doing so it could be a valuable tool in assessing the biophysical, social and economic dimensions of sustainability in agriculture (Tisdell, 1996).

In the past natural scientists have typically used measurements of properties of NC extent or condition (i.e. quantifying Tier 1 properties) or have sought to quantify how ecosystems operate through measuring EF or processes (i.e. quantifying Tier 2 processes) to assess the impacts of land management change. Metrics such as species richness have been used to evaluate changes in biodiversity (Kremen and Miles, 2012; Duncan, Thompson and Pettorelli, 2015), nutrient pollutant status (e.g. phosphorus or nitrate) to assess water quality (Keeler *et al.*, 2012; Peukert *et al.*, 2014) and soil structure or nutrient levels (N, P, K) to evaluate soil condition (Peukert *et al.*, 2012; Rickson *et al.*, 2012; Glendell *et al.*, 2014; Greiner *et al.*, 2017). EFs measurements have included changes in pollination (Hardman *et al.*, 2016a), carbon sequestration (Poulton *et al.*, 2018) or nitrate leaching (Stopes *et al.*, 2002; Benoit *et al.*, 2015). A variety of other metrics have also been used across the natural sciences as quantitative evidence of the impact of different land management decisions on the environment (expanded on in Section 2.3).

In contrast a number of different economic indicators have often been used to evaluate land management decisions and agricultural performance. These include crop yields and revenue from crop sales, farm gross margins (Fezzi *et al.*, 2015), farm income, farm profitability, profitability per ha (Scott, 2020) and the cost-benefit ratio of production (Tisdell, 1996). These indicators are mostly expressed in monetary terms and calculating them is relatively straightforward as crops and the other forms of capital used to produce them are traded on established markets.

A key challenge that arises in conducting the NC approach is trying to align measurements/metrics from the natural sciences with measurements/metrics from economics. The metrics used in the past by natural scientists do not necessarily provide the data required to value ES using monetary terms and quantify the direct impacts to human well-being (Keeler *et al.*, 2012; Duncan, Thompson and Pettorelli, 2015; Smith *et al.*, 2017). That is that metrics used to measure NC condition (Tier 1) (e.g. species abundance) or EF (Tier 2) (e.g. nitrate leaching) are often not appropriate to quantify the value of ES identified in Tier 3 of the framework. Ideally measurements of the natural environment need to align in some way with measurements used in economics. Understanding these ES values are important as they are core to current government thinking and policy making. They are, however, lacking for a range of ES, particularly those derived from biodiversity and the enjoyment of the natural environment (CCI, 2016; Faccioli *et al.*, 2019). The challenge of valuing ES is exacerbated by the fact that the role that EF play in delivering ES can be highly spatially specific (Bateman *et al.*, 2011). This can prevent the validity of using value transfer methods (e.g. from other studies or sites) typical in broad scale environmental economics studies (e.g. Costanza *et al.*, 1997) at the farm scale.

2.1.4 The methods, tools and resources available to undertake the natural capital approach

Whilst the principles behind the NC approach are consistent across different studies, there are two slightly contrasting methods that have been applied in recent farm-scale studies. The Natural Capital Protocol (Natural Capital Coalition, 2021) and Natural Capital Accounting (EFTEC, 2019; Faccioli *et al.*, 2020).

The Natural Capital Protocol (NCP) is a methodological framework provided for use by businesses to include considerations of NC within their decision-making processes. It focuses on businesses building an understanding of their impacts and dependencies on NC (Tier 1) and the likely implications for flows of ES benefits (Tier 2) (Natural Capital Coalition, 2021). The intention is for the NCP to be fairly rapid and to date applications have typically not included field collected data or holistic quantification of ES values (Silcock *et al.*, 2018; Ovando, 2020).

Natural Capital Accounting (NCA) was first developed at the national scale (e.g. Connors, (2016)) and has since been applied at more local scale decision-making and evaluation (EFTEC, 2018; Faccioli *et al.*, 2020). The principles are very similar to the structure presented in Section 2.1.2, focusing on an interconnected series of accounts providing information on NC stocks (an asset register), service flows and economic values. The accounting framework involves both quantifying stock accounts (the extent and condition of NC assets) and service accounts (providing information on the flow of ES) (Faccioli *et al.*, 2020). These can be presented in physical terms (e.g. the amount of carbon sequestered) but the ultimate ambition is to present them in monetary terms (ES values). The suitability of NC accounting at local management scales using the established methods (typically applied at larger spatial scales) has recently been scrutinised (Faccioli *et al.*, 2020).

To complement these frameworks a number of guidance documents have been provided to assist in the application of NC approaches. These include the Natural Capital Protocol handbook (Natural Capital Coalition, 2021), the “How to Guide” developed by the Natural Capital Committee (Natural Capital Committee, 2017) and more recently the “Enabling a natural capital approach” guidance developed by the UK Government (Defra, 2020a).

The Natural Capital Committee ‘How to guide’ (Natural Capital Committee, 2017) and the “Enabling a Natural Capital Approach” guidance documents (Defra, 2020a) both list data sources and tools for gathering/modelling data on NC condition (e.g. biodiversity, soil conditions and pollinators) and ES values. However, these data generally either only include quantitative not qualitative data on NC condition (e.g. areas of farmland or crops) or are at best at a resolution of 1km^2 (e.g. Henrys *et al.*, 2012). It has been identified by other studies that the existing data is frequently not at the appropriate scale to be meaningful to local level applications of the NC approach (Smith *et al.*, 2017; Faccioli *et al.*, 2020). Furthermore, it has been widely acknowledged that there is currently a lack of tools to apply the NC approach to land management decisions at management appropriate scales (Guerry *et al.*, 2015; Maseyk *et al.*, 2017; Smith *et al.*, 2017). Indeed, Howard et al., (2016), in a review of the literature on the tools available to support the practical assessment of NC in land-use decision-making, identify that there is "an apparent absence" of tools for farmers. This is acknowledged in the "Enabling a Natural Capital Approach" guidance which states that applications using the available data and tools may be too "broad-brush" to inform spatially-specific land management decisions (Defra, 2020a). They highlight the need for more detailed appraisal to apply the NC approach at these scales (Defra, 2020a). To meet the needs of this "detailed appraisal" it is likely that empirical data will be required at appropriate spatial scales, requiring input from a range of specialists and stakeholders. This presents a fundamental challenge for those attempting to apply the NC approach as it is likely to require significant resources and specialist support incorporating information from the natural sciences and economics.

2.1.5 Applications of the natural capital approach at local or farm scales

Given the challenges associated with applying the NC approach at small spatial scales it is perhaps unsurprising that Defra (2020a) identify that application of the NC approach at the local level is in its infancy. Table 2.1 provides an overview of recent practical studies that have applied the NC approach at local scales[2].

Table 2.1 highlights that nearly always local or farm scale studies have been informed by modelled data or data derived from other studies (value transfer method) and have rarely collected empirical field data. This has restricted the capacity to build in-depth understanding on the local conditions of NC condition, specific service flows under these NC conditions and harder to measure ES values. Those studies conducting NCA have focused primarily on the quantity of

[2] EFTEC have also completed or are in the process of completing over 12 farm natural capital accounts for farming estates. Unfortunately these are almost always private accounts and so cannot be published or shared (Royle pers. comm., 2021). They follow the same format as in (EFTEC, 2018) with methodological guidance presented in (EFTEC, 2019).

NC assets (i.e. conducting an asset register) and whilst they acknowledge the importance of NC condition in determining ES delivery (Faccioli *et al.*, 2020) they have not quantified such metrics. In contrast, studies applying the NCP, which focuses on identifying the connections, impacts and dependencies of businesses on NC, have typically only conducted a very partial assessment of ecosystem value (e.g. only focusing on carbon sequestration benefits in response to peat restoration) (Kieboom, Silcock and Russ, 2018; Silcock and Russ, 2018; Ovando, 2020). These studies instead tended to report on the "extent" and "condition" and the direction of change of a suite of different NC assets.

More detailed ES valuation of land management change has been attempted in a few academic studies which have combined field measurements of NC condition, some EF and ES values in evaluating different land management practices (Sandhu *et al.*, 2008; Porter *et al.*, 2009; Ghaley *et al.*, 2014b; Sandhu, Wratten, Costanza, Pretty, *et al.*, 2015; Fan *et al.*, 2016). While these field or plot scale studies at sites in Denmark and New Zealand have showcased the start-to-end application of the NC approach (i.e. they have collected/used data from all tiers of the framework) they have relied on a number of relatively coarse assumptions in order to link NC condition to EF (e.g. earthworm abundance to soil formation) and used replacement cost methods as hard-to-justify proxies for the actual ES values (e.g. the price of top soil if no earthworms were present) (Sandhu *et al.*, 2008). These studies have also primarily focused on using ES frameworks to compare and contrast different land management techniques to inform the academic literature rather than refining the approach for land managers. As a result, there is a requirement to build understanding on what data and methods are needed to apply the NC approach at farm scales to ensure it is valuable tool to land managers in making sustainable management decisions. Faccioli *et al.*, (2020) provide a number of recommendations for such future applications of the NC approach, importantly the collection of fit-for-purpose, spatially specific data on NC condition that can inform an understanding of EF and ES provision.

Table 2.1: An overview of recent literature linked to the application of NC approaches at local scales including farm, estate and national park examples. The key to data types used is: F – field data (primary collected data), M – modelled output data and VT – value transfer (data taken from other studies an applied to estimate EF and/or ES).

Study	**Scale**	**Decision or comparison**	**Indicators/metrics used:**			**Data types used:**		
	(Methodological approach)		**Natural capital**	**Ecosystem function**	**Ecosystem service values**	**F**	**M**	**VT**
Eftec (2018)	Cholderton Estate (Natural capital account)	Organic vs conventional agriculture	Quantity of stocks: Grassland, farmland, woodland	Crop and livestock biomass production, nitrate contamination of groundwater, carbon sequestration and avoided emissions	Climate regulation, drinking water provision (water treatment savings), crop and livestock production	N	N	Y
Faccioli *et al.*, (2020)	Exmoor and Dartmoor NP (Natural capital account)	NA	Quantity of stocks: Woodland, open water, grassland, arable etc.	Carbon storage, PM10 absorbed (air quality), crop and timber biomass production	Recreational, climate regulation, air quality, crop production, timber production	N	Y	Y
Silcock and Russ (2018)	Glenlivet Estate (Natural capital protocol)	Peatland restoration	Quantity and quality: Habitats, farmland, woodland, freshwaters	Carbon storage and sequestration	Partial: Carbon emission reductions based on peatland restoration	N	N	Y
Kieboom, Silcock and Russ (2018)	Den Farm (Natural capital protocol)	Improving soil (addressing degradation)	Quantity and quality: Soil, farmland, hedgerows, field margins, freshwaters	Crop biomass production (cereal yields)	Partial: Increases in crop returns measured and estimated (attributed to soil improvements)	Y*	N	Y

(Ovando, 2020)	Glensaugh Farm (Natural capital protocol)	Woodland expansion	Quantity and quality: Habitats, farmland, woodland, freshwaters	Carbon storage and sequestration, crop, livestock forage and timber biomass production	Not conducted	Y	Y	Y

* Partial field data on some soil conditions and mean crop yields

2.2 Natural capital and ecosystem services in the agri-environment

As previously eluded to there are a number of different types of NC in the agri-environment and a summary of these are presented in Table 2.2 including components of soil, water and biodiversity NC. The table is not presented as an exhaustive list and researchers recognise multiple different NC components that contribute to ES (Smith *et al.,* 2017).

The aim of this PhD is to take a multi-disciplinary approach to assessing the impact of different land management practices on NC, EF and ES, as recommended in Kremen and Miles (2012) and Bommarco, Kleijn and Potts (2013). Whilst some studies have adopted a similar approach, comparing multiple ES (Sandhu *et al.*, 2008; Smukler *et al.*, 2010; Ghaley *et al.*, 2014a; Fan *et al.*, 2016), others have focused on specific components of NC or ES (Dominati *et al.*, 2014; Calzolari *et al.*, 2016). Broadly these distinctions are: soil, water and biodiversity and this categorisation is used throughout the review. The following sections introduce each of the three separately.

Table 2.2: An overview of the different types of natural capital that arise in the agri-environment and the component parts, including soil, water and biodiversity.

Natural Capital	Components:	Sources:
Soil	Soil physio-chemical properties Soil water Soil depth (i.e. quantity) Soil biota Soil temperature Soil function and processes	Robinson *et al.,* (2009; Dominati, *et al.,* (2010); Dominati *et al.,* (2014), (2016); Hewitt *et al.,* (2015); Brady *et al.*, (2015); Calzolari *et al.*, (2016)
Water resources	Groundwater	Fenichel *et al.,* (2016); Bergkamp and Cross (2006)
	Surface waters	Khan and Din (2015)
	Wetlands	Khan and Din (2015); Dickie *et al.,* (2015)
Biodiversity	Natural and semi-natural habitats	Dickie *et al.*, 2015; Smith *et al.,* (2017);
	Species (abundance and richness)	Smith *et al.,* (2017)
	Pollinator capital	Hanley *et al.,* (2013); Hanley *et al.,* (2015)
	Earthworms as NC	Keith *et al.*, (2012)

Other forms of NC such as air and oceans have not been focused on in this review or more widely throughout the PhD due to resource limitations. They do, however, represent important NC components that contribute to a range of ES benefits to humans (Smith *et al.,* 2017) and can be significantly degraded by agricultural practices (Pretty *et al.,* 2000; Pretty *et al.,* 2003).

2.2.1 Soil natural capital and soil-based ecosystem services

Soil as NC has received increasing attention in recent years and is now recognised as an important NC asset underpinning the delivery of multiple ES (Robinson *et al.,* 2009; Dominati *et al.,* 2010; Hewitt *et al.,* 2015; Baveye *et al.,* 2016; Dominati *et al.,* 2016). The UK National Ecosystem Assessment (Smith *et al.,* 2011) identified that "Soil quality is linked to almost all other regulating services (e.g. nutrient cycling, biomass production, water quality, climate regulation, pollination, etc.) through the soil's capacity to buffer, filter and transform" (pg. 538).

Since 2009 there has been the development of conceptual frameworks for defining and classifying soil NC (Robinson, Lebron and Vereecken, 2009; Dominati, Patterson and Mackay, 2010) and more recently studies have emerged that quantify the soil ES that flow from soil NC (Dominati *et al.,* 2014; Hewitt *et al.,* 2015; Calzolari *et al.,* 2016). Dominati, Patterson and Mackay (2010) present a conceptual framework linking soil NC, degradation drivers, ES and human needs (Figure 2.2), identifying the importance of the status of 'inherent' and the condition of 'manageable' soil properties as dictating the function and ultimate delivery of ES.

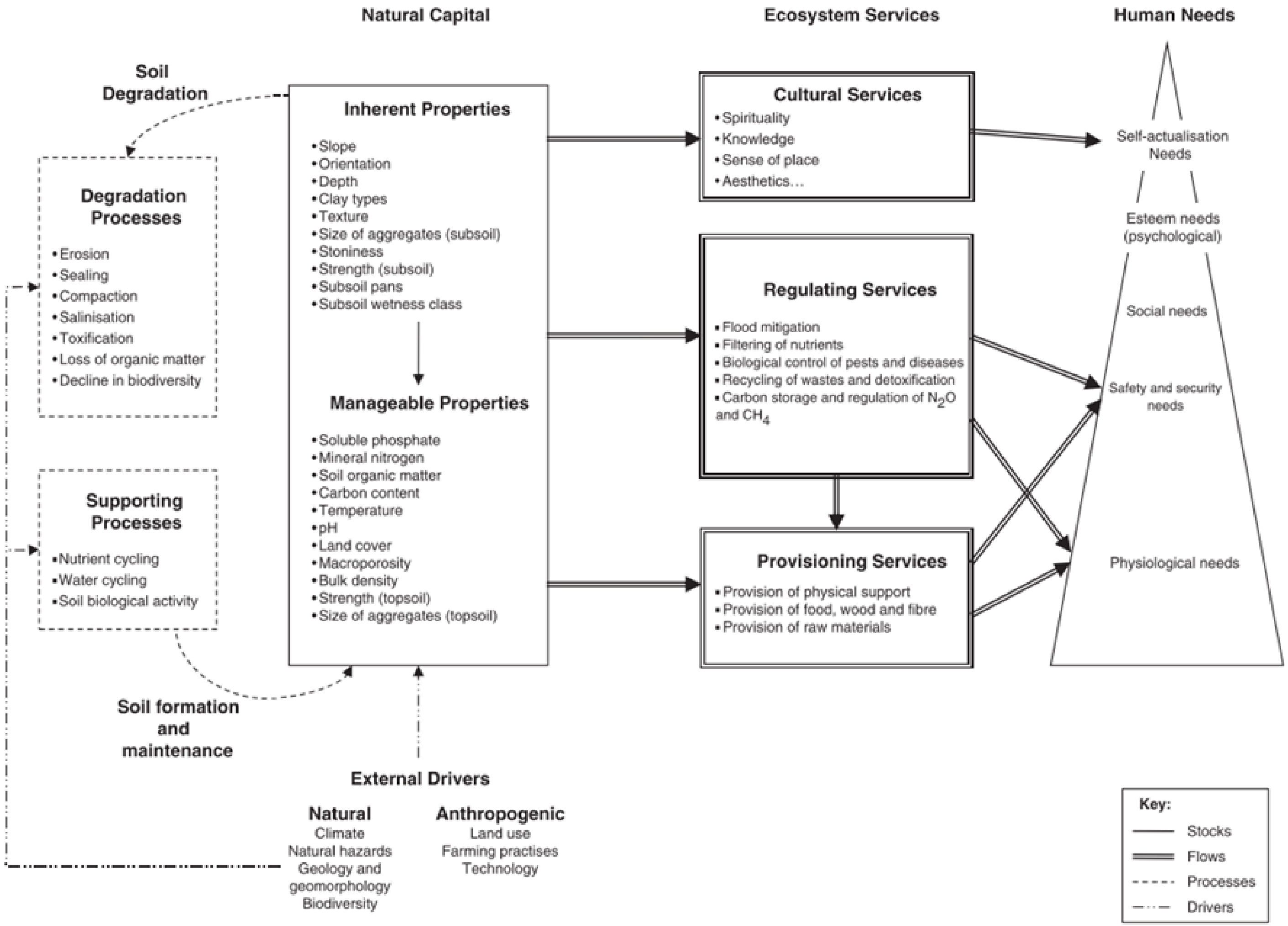


Figure 2.2: Conceptual framework showing the relationship between soil NC, soil processes and soil ES (from Dominati et al., 2010). The framework shows the soil properties and soil functions (referred to in the diagram as processes) important in driving a suite of ES. It also makes clear that there are both natural and anthropogenic drivers of soil NC condition and therefore the flow of ES.

The frameworks established by Robinson, Lebron and Vereecken (2009) and Dominati, Patterson and Mackay (2010) make the case that soil NC should be the starting point to quantifying the value of soil, based on the premise that flows can be inferred from the stocks and that changes to soil NC (i.e. changes to soil properties), whether they be positive or negative, will impact on a soils capacity to deliver ES services. There are, however, still significant gaps in understanding on exactly how different soil properties in a given state impact on soil function and final ES (Baveye, Baveye and Gowdy, 2016). This is discussed further in Section 2.3.1.

2.2.2. Freshwater natural capital and ecosystem services

There is a rapidly growing number of studies on freshwater resources as an ES (Hackbart *et al.,* 2017) but there has been less focus (compared to soil NC) on categorising the NC base from which freshwater ES flow. The provision of clean drinking water, the attenuation of water to reduce flood risk and its value for recreation are examples of some of the key ES derived from the freshwater environment (Brauman *et al.,* 2007; Keeler *et al.,* 2012; Grizzetti *et al.,* 2016). A summary of other ES is presented in Table 2.3.

Brauman *et al.,* (2007) do consider water ES in the context of NC, placing importance on the ecosystem as influencing the quality of water ES, suggesting that the climate, slope, soil and vegetation type, age and management all play a governing role in water use and behaviour. Whilst they do not specifically refer to these ecosystem attributes as NC assets, soil and vegetation type, such as natural or semi-natural habitats have since been considered as such (Robinson *et al.,* 2009; Dominati *et al.,* 2010; Hewitt *et al.,* 2015; Calzolari *et al.,* 2016; Dickie *et al.,* 2015). Groundwater, open water, wetland habitats and flood plains have also been considered as NC assets that play a role in water ES delivery (Khan and Din, 2015; Bergkamp and Cross, 2006; Fenichel *et al.,* 2016). Khan and Din (2015) considered the latter three in the UK's first set of Freshwater Ecosystem Assets and Services Accounts, valuing UK freshwaters at £39.5 million in 2012. The accounts fail to value groundwater, something which Bergkamp and Cross (2006) suggest is often overlooked in the consideration of water ES.

Groundwater is a particularly important asset within the landscape at Clinton Devon Estate (see Chapter 3 for further details). It is recognised that ES delivery from groundwater NC will depend on its quality (i.e. its suitability for drinking water) and quantity (i.e. the amount of water it holds) (Bergkamp and Cross, 2006; Fenichel *et al.*, 2016). Both attributes have been shown to be impacted by agriculture which can have detrimental impacts due to over abstraction for irrigation (Fenichel *et al.,* 2016) or pollution, for example through application and leaching of nitrogenous fertilisers and other agri-chemicals (Bergkamp and Cross, 2006; Wang *et al.*, 2011, 2016). Nitrate pollution of groundwater is a significant issue in the UK and can have a long-term environmental and economic impact due to the high residence times and slow biodegradation in subsurface waters (Bergkamp and Cross, 2006; Stuart and Lapworth, 2016). Whilst Bergkamp and Cross (2006) advocate the consideration of groundwater as “critical natural capital” from which multiple services flow, there appears to be limited literature on groundwater as NC. Furthermore, no known published studies have considered both the environmental and economic impact of land management practices on local ground water NC in the UK.

Multiple studies have investigated the environmental impact of agricultural management on water quality (e.g. monitoring changes in nitrate, phosphate or sediment loads) (Bilotta *et al.*, 2008; Glendell *et al.*, 2014; Peukert *et al.*, 2014; Cooper *et al.*, 2017) but these studies have typically failed to link these to measurements of the economic impacts on ES (e.g. the implications for drinking water costs or recreational value). Keeler *et al.,* (2012) identify that often there is a lack of understanding, available modelling tools or data to make it possible to make meaningful links between the changes in water quality to changes in ES. There is therefore considerable scope to develop understanding on how land management impacts the quality of freshwater NC and the implications this could have on the condition and value of ES.

Table 2.3: Final ES linked to the freshwater water environment and potentially impacted by agricultural land management.

Ecosystem Service	Further details	Source(s)
Clean drinking water		Brauman *et al.*, 2007; Elsin, Kramer and Jenkins, 2010; Keeler *et al.*, 2012; Grizzetti *et al.*, 2016
Water for agriculture	Irrigation, livestock drinking	Brauman et al., (2007); Grizzetti et al., (2016)
Water for commercial or industry use		Brauman et al., (2007); Grizzetti et al., (2016)
Water for thermoelectric uses		Brauman et al., (2007)
Hydropower generation		Brauman et al., (2007); Keeler et al., (2012)
Transportation/navigation		Brauman et al., (2007); Keeler et al., (2012)
Freshwater fish production and commercial fisheries	Freshwater, bays, estuaries, coasts	Brauman et al., (2007); Keeler et al., (2012); Grizzetti et al., (2016)
Flood alleviation		Brauman et al., (2007); Grizzetti et al., (2016)
Carbon Sequestration and climate regulation	Carbon accumulation in sediments	Grizzetti et al., (2016)
Coastal and freshwater recreation	Bathing, boating, nature viewing	Brauman et al., (2007); Keeler et al., (2012); Grizzetti et al., (2016)
Recreational angling		Keeler et al., (2012); Grizzetti et al.,(2016)
Aesthetic appreciation		Brauman et al., (2007); Grizzetti et al., (2016)
Tourism		Brauman et al., (2007)

2.2.3 Biodiversity natural capital and ecosystem services

Biodiversity (comprising of species and ecosystems) is considered to play a central role in EF and the delivery of ES (Maseyk *et al.,* 2017) and Perrings *et al.,* (2006) highlight that it is an important form of NC. Many of the functions delivered by species and ecosystems contribute to final ES including recreational enjoyment, crop production and climate regulation (Perrings *et al.*, 2006; Kremen and Miles 2012). A non-exhaustive overview of the ES linked to biodiversity is presented in Table 2.4. It is beyond the scope of this review to consider all links between biodiversity NC and ES, instead the focus is directed at pollinator NC stocks and ES given the interest in these important insects at Clinton Devon Estate.

Table 2.4: Showing an overview of biodiversity NC, EF and ES in agri-environments.

Biodiversity natural capital	Ecosystem function	Ecosystem service	Source
Stock of natural pests (e.g. ground beetles)	Crop pest predation	Enhanced provision of crops	Sandhu *et al.*, 2008; Pywell *et al.*, 2015; Fan, Henriksen and Porter, 2016; Van Vooren *et al.*, 2017; Mansion-Vaquié *et al.*, 2017
Pollinator stocks	Crop pollination	Enhanced provision of crops	Pywell *et al.*, 2015; Knapp *et al.*, 2019
		Enjoyment of pollinators, wild flowers and other dependent wildlife	Mwebaze *et al.*, 2018
Stock of decomposing organisms (e.g. dung beetles)	Decomposition of dung: nutrient cycling	Enhanced provision of crops/pasture	Beynon, Wainwright and Christie, 2015; Manning *et al.*, 2016
Natural biomass: vegetation, trees etc.	Carbon storage	Climate regulation	Steinbeiss *et al.*, 2008; Van Vooren *et al.*, 2017
Wildlife and ecosystems		Amenity and recreational use	Andersson *et al.*, 2015; Connors, 2016

Pollinators are an important NC stock and The annual value of pollination services to UK agriculture have been estimated at £430 - £603 million (Smith *et al.*, 2011; Hanley, Ellis and Breeze, 2013) and globally from US$195 to US$387 billion (Porto *et al.*, 2020). Despite recognition of their importance they are considered to be nationally and globally declining (Smith *et al.,* 2011; Garratt *et al.,* 2014; Holland *et al.,* 2015). In the UK, as for other temperate regions, the primary insect pollinators are honeybees, bumblebees, solitary bees, wasps and hoverflies (Hanley *et al.,* 2015). Whilst the majority of UK crops are wind pollinated (e.g. maize and wheat) they are particularly important in the pollination of top and soft fruit, some vegetables, oilseed rape (*Brassica napus*) and field beans (*Vicia faba*). (Hoehn *et al.,* 2008). Specific benefits of insect visits to field beans have long been recognised and include associated increases in pod set, beans per pod and pod weight (Garratt *et al.,* 2014). Pywell *et al.,* (2015) reported

25% and 35% increases in field bean yields in Buckinghamshire, UK following the creation of basic and enhanced on-farm habitat, something they attribute to increasing pollinator abundance and diversity.

Furthermore, pollinators have been identified as being important in pollinating wild plants such as hedgerow fruiting plants which are critical in supporting farmland birds (Jacobs *et al.*, 2009). These wild plants and farmland birds, along with observations of the bumblebees, butterflies and more, are all likely to be enjoyed by a large number of people who access farmland. A recent study looking at public support for pollinators show that individuals do value pollinators irrespective of their role in crop production. The study was based on willingness to pay for theoretical bee protection regulations in the UK and shows that households were prepared to pay £43 per year, which based on 30.6 million taxpayers is equivalent to £842 million per year (Mwebaze *et al.*, 2018).

It is argued that valuation of the ES that flow from the pollinator NC stock should be incorporated in decision-making (Hanley *et al.,* 2015). However, whilst pollinator stocks have been valued at national or regional levels there are no known studies that assess, monitor or place a value on pollinator NC at the farm or estate management scale.

2.3 Measuring natural capital and ecosystem services

There is no established suite of metrics for the application of the NC approach at the farm scale and this represents a challenge for its practical application. It is currently necessary for any study applying the NC approach (or similar) to select their own suite of metrics. A review of those used in the literature for soil, water and biodiversity is expanded on below before providing an overview of the common techniques used to value ES. The focus is directed here on quantifying NC relevant to the study site, justification for which is expanded upon in Chapter 4. This includes soil-based ES, groundwater quality and pollinator NC and ES.

2.3.1 Measuring soil natural capital, ecosystem function and soil-based ecosystem services

There is a large amount of literature proposing methods regarding how best to assess the quality of agricultural soils, with many authors suggesting different soil quality indexes (SQI) (Askari *et al.*, 2015; Obade and Lal, 2016) or different indicators (Barrios, 2007; Brazier et al., 2011). Determining what metrics to use in order to assess soil condition is recognised as being important in enabling decision-making (Obade and Lal, 2014) and assessing soil function and quantifying or signalling the delivery of soil-based ES (Williams and Hedlund, 2013; Greiner *et al.*, 2017). Despite the recognition of metric selection being important there remain fundamental gaps in understanding on how some of the established measurements of soil condition actually link to soil function and the delivery of final soil-based ES (Baveye, Baveye and Gowdy, 2016).

An overview of frequently used indicators of soil function used in soil-based ES studies is presented in Table 2.5. Greiner *et al.,* (2017) in their review of the literature (n = 181) of soil assessment methods for mapping ES, identify the most frequent soil properties used as indicators to assess soil function and infer ES delivery as: soil organic carbon (SOC), available water capacity, clay and silt contents (texture), soil type, depth and bulk density (BD). Other parameters such as physical properties, like macroaggregate stability, P, N, pH, Cation Exchange Capacity (CEC), air capacity and C:N ratio were less frequently included (Greiner *et al.*, 2017). The Natural Capital Committee (2019) identify a similar suite of frequently cited soil metrics; BD, pH, SOC, soil N and soil P. At a national scale the Countryside Survey for Great Britain, conducted since 1978 as an 'audit' of the natural resources of the UK, assesses change in soil BD, carbon, pH, nitrogen and mineralisable-N, Olsen-P, metals and soil invertebrates (Emmett *et al.,* 2010). Measurements of carbon stocks taken from Emmett et al., (2010) have since been used in the first national NC accounts (Connors, 2016).

Table 2.5: Soil properties and functions used in the literature to quantify soil function and signal the potential delivery of soil-based ES. Adapted from Greiner et al., (2017).

Soil Function	**Eco-system service**	**Soil property data suggested or applied in the literature to quantify soil function**														**Source(s)**	
		SOC	Texture	pH	Bulk Density	Soil depth	Stone content	Soil water properties	Hydraulic conductivity and infiltration	Cation Exchange Capacity	Porosity	Structure and aggregation	Microbial activity	Earthworm diversity and/or abundance	Bait lamina	Review texts	Field/lab studies
Water cycling and storage:	Flood alleviation	x	x		x	x	x	x	x		x	x				Adhikari and Hartemink, 2016; Greiner *et al.*, 2017; Rabot *et al.*, 2018	Archer *et al.*, (2013)
Infiltration	Provision of water																
Water storage for plants	Crop production																
Groundwater recharge																	
Water filtration:	Provision of clean water	x	x	x	x	x	x	x	x	x			x			Adhikari and Hartemink, 2016; Greiner et al., 2017; Rabot et al., 2018	
Nutrient cycling:	Crop production													x	x	Adhikari and Hartemink, 2016; Greiner et al., 2017	Sandhu *et al.*, 2008; Williams and Hedlund 2013; Calzolari *et al.*, 2016
Nutrient availability to plants		x	x	x	x	x	x			x		x	x				

Nutrient retention:	Crop production Provision of clean water	x	x	x	x	x			x	x		x	x	x	Adhikari and Hartemink, 2016; Greiner et al., 2017	Williams and Hedlund, 2013
Storage, filtration and degradation of pollutants	Provision of clean water Crop production	x	x	x	x	x		x			x	x			Greiner et al., 2017	Dominati *et al.*, 2016
Carbon cycling and storage	GHG regulation	x	x		x	x	x						x		Adhikari and Hartemink, 2016; Greiner et al., 2017	Williams and Hedlund, 2013
Supporting plant growth: Habitat for plants Supporting long-term crop yield potential	Crop production Cultural services	x	x	x	x	x	x	x	x	x		x	x		Adhikari and Hartemink, 2016; Greiner et al., 2017	
Habitat for pests, diseases and their natural enemies	Natural pest managem ent Crop production	x		x				x					x		Adhikari and Hertemink 2016	Dominati et al., 2016
Physical support for infrastructure			x		x			x	x		x	x			Rabot et al., 2018	Calzolari *et al.*, 2016; Dominati *et al.*, 2016

Most studies to date have used terminology around quantifying or using indicators of soil function and/or soil-based ES, rather than explicitly assessing soil NC condition. Studies have applied a range of approaches to quantify or signal change in soil function and ES, using empirical data collected from agricultural trials (Sandhu *et al.*, 2008, 2015; Williams and Hedlund 2013; Ghaley *et al.*, 2014a), existing regional and national soil data (Calzolari *et al.*, 2016) or modelled soil functions (Dominati *et al.*, 2014). Some workers have stopped at quantifying soil indicators of soil function (Williams and Hedlund, 2013). Others have used scored indexes based on soil functional capacity (Calzolari *et al.*, 2016) and some have sought to place a monetary value on soil-based ES (Sandhu *et al.*, 2008; Dominati *et al.*, 2014; Sandhu, Wratten, Costanza, Pretty, *et al.*, 2015).

Soil-based ES studies to date have often used either modelled, regional or national data (Dominati *et al.*, 2014; O'Sullivan *et al.*, 2015; Calzolari *et al.*, 2016), based on relatively few physical and chemical, and typically no biological, soil property measurements and taken over large spatial scales, to infer soil condition and function. Whilst considerable spatial variation of soil properties has been recognised as small as at the field scale (Peukert *et al.,* 2012), there is less evidence of field or farm scale assessments of soil NC, informed through field measurements and assessed alongside soil function. Whilst some workers have found partial agreement between some soil properties in national soil dataset data and field data (Glendell *et al.*, 2014), there is a need to understand the implications of using such coarse data to determine soil NC at the farm scale. There is therefore considerable scope to build upon the examples presented and using existing soil NC frameworks refine its practical application at management appropriate scales.

2.3.2 Measuring freshwater natural capital, ecosystem function and ecosystem services

The concept of valuing water ES appears to have received significantly more attention than considering water resources within the context of a NC approach. However, despite the development of frameworks to quantify water ES there is no consensus on a standard methodology and few studies have measured the actual response of water ES linked to changes in land management (Hackbart *et al.,* 2017).

Grizzetti *et al.*, (2016) consider the connections between pressures on the water environment, ecosystem status and ES presenting a framework which has synergies with that for the NC approach. A summary of the potential biophysical measurements of ecosystem status (re-phased here as NC components) and the connection with ES delivery is presented in Table 2.6. The study advocates the use of indicators of water-based ES recognising the need for biophysical data (to quantity the ecosystem process) and social-economic data to quantify ES value. They use the example for groundwater ES, highlighting the need for measurements of available renewable water, understanding of water abstractions and nitrogen contamination and an understanding of the costs to the water sector (including water treatment costs) (Grizzetti *et al.*, 2016).

Table 2.6: An overview of components of freshwater NC, potential important properties of these components of NC and the ES that could be impacted by them. Adapted from Grizzetti *et al.*, (2016) *framework analysing the links between pressures, ecosystem status (described here as NC) and ES.*

Components of NC	**Indicators of NC and/or EF**	**Ecosystem services**
Water quantity	E.g. Water flow, groundwater recharge, E-flow (flow regime that mimics natural pattern of river flows)	Water provision for drinking Water provision for non-drinking (e.g. industry, irrigation, navigation) Flood alleviation
Water quality	E.g. Nitrogen (N), phosphorus (P), sediments, pesticides, metals	Water provision for drinking Fisheries (food and recreational)
Biological elements	E.g. Chlorophyll, algal blooms, fish biomass, macroinvertebrates	Climate regulation (carbon sequestration) Recreational (e.g. swimming, boating, nature viewing, fishing)
Hydromorphological structure	E.g. Nursery habitat, natural habitat, natural flood plains, riparian areas	Flood alleviation Fisheries (food and recreational) Water provision (through water purification)

2.3.3 Measuring biodiversity natural capital, ecosystem function and ecosystem services

A summary of the use of different indicators to infer specific biodiversity linked ES in farm scale studies has been provided in Table 2.7 below. It is evident that some studies have used different metrics to assess the same ES, however there are consistent overlaps. Whilst studies have measured a number of important EF often they have stopped short of valuing ES (although see Sandhu *et al.*, 2008; Fan, Henriksen and Porter, 2016). Where they have quantified the value of ES there have been instances of double counting (e.g. based on soil formation; see Fan, Henriksen and Porter, (2016)) and frequent use of replacement cost method (e.g. cost to replace natural predation or decomposition and N-mineralisation with pesticides or fertilisers respectively). The issues with the replacement cost approach is covered in the following section.

Assessing the impact of land management changes on pollinators has been conducted through a number of methods and commonly pollinators (e.g. bumblebees, honeybees, butterflies) are measured based on abundance and species diversity (Pywell *et al.*, 2006, 2015; Gabriel *et al.*, 2013; Carvell *et al.*, 2016). Measuring the EF of pollination has also been conducted by using plant phytometers placed within farmland (Hardman *et al.*, 2016b). Whilst these do not reflect actual crop pollination they provide an insight into whether different land management is increasing pollination of plants in the vicinity (Hardman *et al.*, 2016b).

Wood *et al.,* (2015) provide one of the few examples that have considered population-level change in pollinators (focusing on common bumblebee species) as a response to land management change (i.e. adding pollinator habitats across a farm). They highlight that a problem with earlier studies has been that it is unclear whether increased bumblebee abundance in and around sown flower habitats is simply due to attracting workers to that location or corresponds with a genuine population increase. Population level responses are important to understand as they really reflect changes in pollinator stocks (increases in NC quantity). The quality of the stock can then be considered on the basis of functional biodiversity (i.e. the assemblage of pollinator groups and species). Bumblebees are identified as being of particular importance as a commercial crop pollinator and they are identified as a keystone species (Goulson *et al.*, 2011). Visitation rates by bumblebees have been used by one study to directly link pollination with increased productivity of courgettes in Cornwall (Knapp *and Osbourne* 2017). Measuring bumblebee populations is not without difficulties, however, and is resource intensive (Wood, Holland and Goulson, 2015). Advances in bumblebee models present a resource-efficient means to address this challenge (Becher *et al.*, 2018; Twiston-Davies, Becher and Osborne, 2021). For example, the recently published BEE-STEWARD software offers the potential for land managers, academics and other stakeholders to obtain a fairly rapid understanding of the implications of different land management changes on bumblebee populations across a range of scenarios (Twiston-Davies, Becher and Osborne, 2021).

Market valuation of pollinator services has typically been conducted by either estimating the proportion of yield loss without pollination (based on assumed Dependence Ratio – DR, of particular crops) or through replacement cost methods (Hanley *et al.,* 2013). DRs can be drawn from the literature, for example field beans could be considered to have a moderate DR of 25% and then this percentage removed from the known yield, either at a national or local level (Hanley *et al.,* 2013). This method was used by Smith *et al.,* (2011) and Hanley *et al.,* (2013) to come up with the estimates used to value UK pollination services. It is recognised, however, that this approach contains inherent weakness and is often based on DRs taken from studies in different countries with different conditions (Hanley *et al.,* 2013). Replacement cost methods, such as estimating the costs of replacing insect pollinators by paying for hand pollination, are also not without their issues as the value does not reflect the fact that often it is too costly to be viable and is not effective for some crops. This means the figures generated are unlikely to ever be paid by a farmer, who is more likely to switch to a less pollinator reliant crop (Hanley *et al.,* 2013). These methods whilst acknowledged as offering a simplified measure of pollinator value, do present a way in which to consider the value of pollinator NC in landscapes dominated by insect pollinated crops.

Table 2.7: An overview of the metrics used in the literature to either assess different ecosystem functions or services derived from biodiversity in agricultural landscapes or correlating with the delivery of specific ES. Study types (ST), values used are from: FM (field measurements), ED (existing data), L (literature) or Q (questionnaire outputs). Other acronyms used are for hedgerows (HR), grassland (GS), abundance (abund.) and species diversity (SD)

			Metrics used for the assessment of specific ecosystem functions or services												
Ecosystem function	**Study**	**ST**	Aphid predation rate	Predation of artificial prey	Abund. of functi. insect groups	Abund. and diversity of coleoptera	Plant SD	Dung beetle abund. functional	Bumblebee and honeybee abund. & SD	Bee visitation	Bait lamina	Soil carbon	Semi-natural habitat presence and abund.	Bird diversity	Recreational visits
Natural Pest control	Sandhu *et al.*, (2008)	FM	x												
	Fan, Henriksen and Porter (2016)	FM	x												
	Mansion-Vaquié *et al.*, (2017)	FM	x	x											
	Pywell *et al.*, (2015)	FM				x							x		
Pollination of crops	Pywell *et al.,* (2015)	FM							x				x		
	Knapp and Osborne (2017)	FM								x					
Decomposition of dung; reducing pasture fouling	Manning *et al.*, (2016)	FM						x			x				
	Beynon et al., (2015)	FM & L						x							
Decomposition of dung: nutrient cycling	Beynon et al., (2015)	FM & L						x							
Carbon storage	Steinbeiss *et al.*, (2008)	FM					x					x			
	Van Vooren *et al.*, (2017)	ED & L										x	x (HR and GS)		
Amenity/recreation use:	Andersson *et al.*, (2015)	ED & Q					x							x	
	Connors (2016)	ED													x

2.3.4 The valuation of natural capital and ecosystem services

In economics the concept of value relates to welfare or utility. That is the enjoyment we gain from an object or activity (CCI, 2016). For example undertaking a recreational activity in a beautiful environment contributes to economic value as it has the potential to increase our welfare (CCI, 2016). Other services such as food and water provide nourishment and hydration also contributing to our welfare. Economic values differ from financial values, which are derived from prices, two very different things in economics. Prices might signal how much someone is willing to pay to secure a product but it will not necessarily reflect its 'value' (CCI, 2016). Indeed, in the context of ES the prices paid for services such as clean air might be zero but it does not mean that clean air has no value (CCI, 2016). Some welfare values can be considered in monetary terms using a suite of methods (Table 2.8) and by doing so can be incorporated alongside other costs and benefits in decision-making (Bateman *et al.*, 2011; CCI, 2016).

The economic value of a NC stock at a point in time is defined by the present value of the future stream of market and non-market valued benefits (Hanley, Ellis and Breeze, 2013). Whilst valuation of NC and ES is identified as important in undertaking the complete NC approach to decision-making, research is still required to refine valuation methods for non-market services or dis-services. Brady *et al.*, (2015) identify that a "lack of practical methods to value the long-term effects of current farming practices results, inevitably, in short-sighted management decisions" (pg, 1809).

Table 2.8: Showing an overview of the different methods applied in the economic valuation of ES that flow from NC.

Type	**Description**	**Sources:**
Market price or adjusted market price method	Use of established market prices from commercial markets. Adjustment can be made for distortions such as taxes, subsidies or non-competitive practices.	(Power, 2010; Bateman, Mace, *et al.*, 2011; FAO, 2015)
Production Function method	Use modelled or observed understanding of scale of contribution a specific ES has on the delivery of a good traded on a commercial market. E.g. % contribution pollination has on crop yield.	(Brown *et al.,* 2007; Bateman *et al.,* 2011; FAO, 2015)
Replacement cost method	Estimation of cost required to replace the ES if it is lost. This could include replacement with human, built or NC. Note: Bateman et al., (2011) suggest that replacement methods should be used with caution and that often restoration or replacement costs are unlikely to even closely resemble the values they attempt to approximate.	(Brown *et al.,* 2007; Power, 2010)
Damage cost avoided method	Calculation of costs avoided by not allowing NC and ES to degrade.	(Bateman, Mace, *et al.*, 2011)
Revealed Preference methods	Travel cost method: Observed expenditure and time spent to visit recreational areas Hedonic pricing: Examines premium people are willing to pay to purchase property in area of higher environmental quality Averting behaviour: Observed expenditure of individuals to avoid environmentally degraded areas that could pose risk to human health	(Brown *et al.,* 2007; Bateman *et al.,* 2011; FAO, 2015)
Stated Preference methods	Contingent valuation: Survey technique asking respondents to state their willingness to pay to assess or maintain an environmental feature or service Discrete choice experiments: Presents choice of sets of services or environmental features to select preferences	(Brown *et al.,* 2007; Bateman *et al.,* 2011; FAO, 2015)
Abatement costs are not true economic values but they can be used to apply a monetary figure to a negative externality arising due to agricultural management. These costs include the cost of treating or removing a negative externality of production through reducing emissions or reducing impacts. For example using the reduced costs of water treatment (FAO, 2015) to place a 'value' on mitigating practices that reduce water contamination.		

2.4 Agriculture at a crossroads

Agricultural intensification through the 20th century and the pursuit of increased crop production has had a significant impact on natural and semi-natural environments (Pretty *et al.*, 2000, 2003; Tilman *et al.*, 2002; Stoate *et al.*, 2009; Graves *et al.*, 2015; Hayhow *et al.*, 2016). It is widely accepted that progressive land management solutions are needed to address these issues whilst sustaining food production (Bommarco et al., 2013; Elliott et al., 2013; Firbank et al., 2013; Petersen and Snapp, 2015; Pearce, 2016). This is likely to be reflected in a future UK agricultural policy as the UK exits the European Union where there are strong signals that future agricultural subsidies will be targeted at farmers delivering public ecosystem goods and services (Bateman and Balmford, 2018). The following section provides an introduction to the impacts of agricultural intensification as context to the challenges faced before expanding on the changing state of UK agricultural policy and the consequences for the future. It finishes by providing an overview of land management practices that could tackle the problems associated with intensive agriculture and the deliver wider ES. There is a wide literature on this final topic and in the interest of brevity the section focuses on management systems and techniques that relate to the case-study site.

2.4.1 The impact of agricultural intensification

Agricultural yields in the UK and globally have significantly increased during the 20th and 21st century (Pretty *et al.,* 2000). Whilst this has significantly increased food production it is widely recognised that this has been to the detriment of the environment, impacting other important ES such as clean drinking water supply, flood regulation and biodiversity (Tilman *et al.,* 2002). Despite the recognition of the value of soil, water and biodiversity NC the condition of all is generally considering to be in decline. Biodiversity is declining at global and national scales (Stoate *et al.*, 2009; Hayhow *et al.*, 2016), soil has been significantly degraded in the UK over the past 50 years (Smith et al., 2011; Brazier *et al.*, 2011) and 65% of UK surface waters are failing to achieve Good Ecological Status under the Water Framework Directive (WFD) (JNCC, 2017). The impacts have been significant and Pretty *et al.,* (2000) estimate the total cost of the externalities associated with UK agriculture in 1996 at £2.3 billion. Whilst the negative impacts associated with agriculture can impact on the farm business, the majority of the impacts appear to occur off the farm which offers little incentive for the farmer to address them (Stoate *et al.,* 2001; Graves *et al.,* 2015). Graves *et al.,* (2015) suggest that whilst 20% of the annual quantified costs of soil degradation are associated with the loss of provisioning services (predominantly food production), 80% are linked to the loss of important regulating services.

Table 2.9 provides an overview of some of the main impacts of agriculture in the UK and, where available, the estimated costs (private and public) associated with these. It is evident that modern-day intensive agriculture has a poor track record with regards to sustaining soil, freshwater and biodiversity NC condition. This NC degradation is having a significant impact on the ES that flow from it, with likely implications on its present and future value.

Table 2.9: A non-exhaustive summary of the impacts of intensive agriculture on the environment. Including the area of impact, the specific impact, the main agricultural management drivers and the estimated costs of the impact (where available). Sources include review and empirical studies.

Area of impact	Environmental Impact	Main agricultural practices responsible	Est. cost of issue (if estimated) (£'million per yr)	Source(s)
Degradation of soil	Soil erosion and loss of productive top soil	Over stocking, winter cropping (bare fields), poorly timed grazing and crop cultivations or harvesting	£165**	(Pretty *et al*., 2000; Bilotta *et al*., 2008; Pilgrim *et al*., 2010; Graves *et al*., 2011**; Glendell and Brazier, 2014)
	Decrease in soil fertility			
	Damage to soil structure; capping, compaction	Over stocking, winter cropping (bare fields), poorly timed grazing and crop cultivations or harvesting	£393**	(Graves *et al.,* 2011** (compaction only); Bilotta, Brazier and Haygarth, 2007)
	Reduction in soil carbon and increased CO_2 emissions	Ploughing up of pasture, intensive cultivation	£82.3* - £558**	(Pretty *et al.,* 2000*)
	Damage to soil biodiversity			(Graves *et al.,* 2011)
Pollution of fresh, estuarine and coastal waters; reduced water quality	Increased incidence of eutrophication	N & P pollution from fertiliser, livestock wastes, soil erosion		(Pretty *et al*., 2003; Mu *et al*., 2006; Bilotta *et al*., 2008; Glendell and Brazier, 2014; Global Food Security, 2014; Peukert *et al*., 2014; Withers *et al*., 2014)
	Damage to drinking water quality	Nitrate leaching from livestock waste and fertiliser	£16.4*	(Pretty *et al*., 2000*; Howden *et al*., 2013; Stuart and Lapworth, 2016)
	Damage to drinking water quality; pesticide pollution	Herbicide, insecticide and fungicide application to treat crop diseases	£124.9*	(Pretty *et al.,* 2000*)
	Damage to drinking water quality; phosphate and soil	Soil erosion (see above) and increase fertiliser application	£52.3*	(Pretty *et al.,* 2000*)
	Increased flood risk	Changes to catchment hydrology: vegetation cover change, soil structural change, land drainage		(Wheater and Evans, 2009; Graves *et al*., 2011; Dadson *et al*., 2017; Rogger *et al*., 2017)
	Damage to aquatic biodiversity	Eutrophication, pesticides, sedimentation		(Kemp *et al*., 2011; JONES *et al*., 2012)

	Reduction in amenity value; pollution of rivers, bathing waters and lakes	Eutrophication, sedimentation, pollution from animal waste (inappropriate slurry or manure spreading)		(Pretty *et al.*, 2003; Ferrini, Schaafsma and Bateman, 2014)
Air quality and pollution	Greenhouse gas emissions; methane (livestock), nitrous oxide (fertiliser), carbon dioxide (farm operations)		£1065*	(Pretty *et al.*, 2000*; van Groenigen *et al.*, 2010; Horrocks *et al.*, 2014)
	Ammonia emissions causing soil, water acidification, terrestrial eutrophication	Livestock waste spreading and storage		
Biodiversity	Physical habitat destruction			(Stoate *et al.*, 2001; Hayhow *et al.*, 2016)
	Reduction in wildlife species			(Pretty *et al.*, 2000; Hayhow *et al.*, 2016)
	Local extinction of species			
**** and * denote which study the costs of impact was taken from**				

2.4.2 The future for agriculture in the United Kingdom

Many of the detrimental impacts of modern intensive agriculture shown in Table 2.9 have come about in the 60 years of the European Unions (EU) Common Agricultural Policy (CAP). Given the scale of environmental degradation, it is perhaps unsurprising that the CAP is now widely recognised as not having been fit for purpose, having had both significant financial and environmental costs (Bateman and Balmford, 2018). As the UK exits the EU it has the capacity to leave the CAP behind and change its domestic agricultural policy which is likely to have a profound impact on agriculture. Whilst the government has committed to keep existing Common Agriculture Policy (CAP) payments in place for now there is clear signalling that major changes in policy are imminent (Bateman and Balmford, 2018). The future of land management support schemes remains unclear but there has been strong messaging from the UK Government that there is likely to be a shift in emphasis away from baseline support for agricultural production towards the targeting of public money towards delivering public goods (Bateman and Balmford, 2018). Such a land management support scheme sits well with the use of the NC approach to land management decision-making which aims to systematically understand the implications on public and private ES goods and services.

Leaving the CAP behind presents a significant opportunity for the UK to address failings of the past 60 years and deliver an agricultural policy that supports agricultural production methods that both produce food and enhance the delivery of other ES. Bateman and Balmford (2018) suggest that “It is the net public benefits to society which farming can generate that should be the focus for, and determine the level of, future public subsidies (pg. 294). They note that there are nuances to this approach and in some cases it will be efficient to support the development and uptake of techniques which increase productivity (output per unit input) whilst lowering environmental inputs. Linking such productivity gains in some places might facilitate ‘land sparing’ approaches (taking land out of production) which could also be necessary in delivering the objectives of the Governments 25 year Environment Plan (Bateman and Balmford, 2018). Critically a focus of any future policy needs to be on supporting approaches that support high environmental improvements alongside profitable farming.

The future for agriculture in the UK is therefore uncertain but there is clear recognition that change is needed. Whilst the majority of farms will still have the profitable production of private goods (food and fibre) as their primary aim, it is likely that public subsidy will be orientated towards the delivery of public goods (Bateman and Balmford, 2018). This raises a challenge for land managers wishing to access public subsidy support and is likely to require many of them to adopt agricultural techniques and systems that are capable of delivering improvements in public goods.

2.4.3 Agricultural solutions that balance food production with the delivery of ecosystem services

In response to the challenge for farming to tackle the detrimental impacts of its past and deliver both private and public goods and services a number of different land management techniques have been promoted. These techniques could be applied to the agricultural approach (e.g. organic, conservation or conventional agriculture), the farming system (e.g. grazing, cropping or cultivation system) or to particular field management (e.g. cover cropping or in-field habitat creation). An overview of key land management practices considered applicable to the study area, alongside potential yield or environmental gains or losses are presented in Table 2.10.

A number of different agricultural approaches have also been proposed, often adopting one or more of the techniques in Table 2.10, that have the common goal of addressing the impacts of intensive practices and improving the environment. Often these approaches will have the aim of achieving this goal whilst maintaining or even increasing productivity. Petersen and Snapp (2015) identify six such approaches that have been raised in the literature and they include organic and conservation agriculture, agroecology, ecological and sustainable intensification and sustainable farming systems. The most widely practiced of these is organic agriculture, which is conducted on around 0.9% of the worlds agricultural land (Ponisio *et al.*, 2014) and is the focus of this review given its relevance at the study location. Table 2.11 provides some examples of field or farm scale studies that have looked at the delivery of multiple ES of different approaches (notably organic management and ecological intensification).

.

Table 2.10: An overview of field and farm management practices and associated potential crop yield and environmental benefits or impacts. The potential for Improvement are shown as ↑, reduction as ↓ and potential for no difference as -.

	Soil Structure	Soil fertility	Soil erosion prevention	Soil organic carbon	Soil moisture	Water quality	Water quantity	Biodiversity	Climate Regulation	Yield	Livestock disease	Pest /weed control	Sources
Crop management practices													
Cover cropping	↑	↑	↑		↑	↑		↑		↑ -		↑ ↓	(Chen, Weil and Hill, 2014; Schipanski *et al.*, 2014; Cooper *et al.*, 2017; Prechsl *et al.*, 2017)
Targeted crop rotations	↑	↑	↑	↑				↑ -	↑	↑ -		↑	(Ball *et al.*, 2005; Kremen and Miles, 2012; Barbieri, Pellerin and Nesme, 2017)
Use of mixed legume leys	↑	↑	↑				↑ ↓	↑				↑	(Honisch, Hellmeier and Weiss, 2002; Döring *et al.*, 2013)
Crop and land-use selection													
Stock management practices													
Holistic plan/mob/Managed Intensive Grazing				↑						↑ -			(Leach *et al.*, 2014; Machmuller *et al.*, 2015)
Cultivation practices													
Reduced tillage regimes	↑		↑	↑	↑		↑		↑	↑ ↓			(Powlson *et al.*, 2012; Büchi *et al.*, 2017; Jarvis and Woolford, 2017)
Sub-soiling or soil aeration	↑	↑								↑			(Bhogal *et al.*, 2011)
Nutrient management practices													
Precision agriculture		↑				↑				↑			(Awan, 2016)
Nutrient management planning													

Targeted organic matter application	↑				↑			(Powlson *et al.*, 2012)
Green manure incorporation	↑							(Sharma *et al.*, 2017)
Pest/disease management practices								
Habitat creation/management to support natural pests				↑		↑ ↓	↑ ↓	(Bianchi, Booij and Tscharntke, 2006; Landis *et al.*, 2008; Bommarco *et al.*, 2012; Tschumi *et al.*, 2016; Hatt *et al.*, 2017)
Precision agriculture for pesticide application							↑	(Petersen and Snapp, 2015)
Habitat management practices								
Habitat creation/management			↑	↑	↑	↑ ↓	↑	(Kremen and Miles, 2012; Marshall *et al.*, 2014; Pywell *et al.*, 2015)
Management/creation legume and herb species rich grassland (temporary or permanent)				↑				
Creation of in-field, riparian or marginal grass/flower strips	↑	↑		↑			↑	(Kremen and Miles, 2012; Pywell *et al.*, 2015; Hatt *et al.*, 2017)

2.4.4 Organic agriculture, a solution to balance profitable farming with ecosystem service delivery?

Tully and Mcaskill (2020) explain that organic agriculture "is a mixture of modern technologies and tools with traditional (sometimes ancient) management practices. Practically defined, organic farmers do not use synthetic fertilizers or pesticides, and instead rely on rotating crops, managing pests naturally, and providing crops with nutrients via compost, manures, and legume residues."

There is growing interest, particularly within the European Union (EU), in the role organic agriculture could play in enhancing the farmed environment as part of their Farm to Fork strategy. Indeed, the EU have recently announced the target of converting 25% of the utilisable agricultural area to organic management by 2030 (Comissão Europeia, 2020). There have been calls for the UK Government to match the target in the UK (ORC, 2021), which already has a percentage cover of organic agriculture far lower than that of most other European nations (Scott, 2020).

Whilst it is recognised that organic farming does not typically deliver the same yields as conventional farming and whilst 'organic' and 'sustainable' farming cannot be used interchangeably, it can perform better than conventional farms in delivering environmental improvements (Kremen and Miles, 2012; Tuomisto *et al.*, 2012; Ponisio *et al.*, 2014; Tuck *et al.*, 2014; Stein-Bachinger *et al.*, 2021). A number of field studies and meta-analyses have shown that organic farming can enhance water retention and soil structure (Lotter, 2003; Gomiero, 2013; Williams *et al.*, 2017), increase soil carbon stocks (Mondelaers, Aertsens and Huylenbroeck, 2009; Gomiero, Pimentel and Paoletti, 2011; Gattinger *et al.*, 2012; Tuomisto *et al.*, 2012), reduce nutrient losses (nitrate, nitrous oxide and ammonia) (Snapp, Gentry and Harwood, 2010; Tuomisto *et al.*, 2012; Benoit *et al.*, 2014; Biernat *et al.*, 2020), improve soil biology and biological processes (e.g. the decomposition of organic matter) (Domínguez *et al.*, 2014; Lori *et al.*, 2017; Martinez-Garcia *et al.*, 2021) and enhance pollinator stocks and pollination services (Feber *et al.*, 1997; Holzschuh, Steffan-Dewenter and Tscharntke, 2008; Andersson, Rundlöf and Smith, 2012; Hardman *et al.*, 2016a). Some authors have been cynical of some of these findings (Leifeld *et al.*, 2013; Kirchmann *et al.*, 2016) and other studies have found no significant differences between the environmental performance of each system (Gosling and Shepherd, 2005;

Williams and Hedlund, 2013). Tuck et al., (2014) therefore identify that there remains a need to understand more about the precise environmental benefits delivered by organic farming.

Due to the lower yields typically associated with organic agriculture there is also a debate over the role that organic agriculture might play in the transition to a more sustainable agricultural system (De Ponti, Rijk and Van Ittersum, 2012; Ponisio *et al.*, 2014; Röös *et al.*, 2018; Smith *et al.*, 2019). Opposition to organic agriculture raise these lower yields as a significant issue, highlighting that this means more land is needed to produce the same amount of food leaving less space for natural or semi-natural habitats. They argue that this results in organic agriculture delivering higher environmental impacts per unit produced (Connor and Mínguez, 2012). In contrast advocates support the wider expansion of organic agriculture, highlighting the positive delivery of environmental benefits and the frequently improved profitability for producers (Crowder and Reganold, 2015; Wilbois and Schmidt, 2019). Muller *et al.*, (2017) and Wilbois and Schmidt (2019) offer a more nuanced point, that consideration on the role of organic agriculture in the global food system should not be whether it can entirely replace conventional agriculture but how it can be utilised, alongside other strategies, to improve the sustainability of farming into the future.

Given the growing interest in the expansion of organic agriculture there is a need to further understand the scale and consistency with which it can deliver environmental benefits, the trade-offs in yield and the implications for farm profitability.

Table 2.11: Showing a summary of literature investigating multiple ES at the field or farm scale in response to different farm approaches (organic management and ecological intensification) or management (organic matter additions). Improvements are shown as ↑, reduction in the service ↓ and no difference between the treatments as -.

Study	**System and Treatment or comparison**	**Yield**	**Key function (as an indicator of ES) studied:**						
			Soil carbon sequest-ration	Nitrogen efficiency or cycling	Reduced Nitrate leaching	Biocont rol by natural pests	Soil water storage	Biodiver sity	Economic valuation
Snapp, Gentry and Harwood, (2010)	Organic vs conventional rotations (12 year study) Treatment: organic mgmt.	↓		↑	↑				Not applied
Sandhu *et al.*, (2008)	Organic vs conventional arable: Treatment: organic mgmt.	Only use market price	↑ (marginal)	↑		↑			Org: $4600 US ha-1 yr-1 Conv: $3680 US ha-1 yr-1
Williams and Hedlund (2013)	Organic vs conventional barley: Treatment: Organic mgmt.	↓ Decline (33%)	-	↑					Not applied
Fan, Henriksen and Porter (2016)	Organic mixed: Increasing scales of organic matter added (Low, Med, High amendments)	↑	↑	↑		↑	↑		Low: $1502 US ha-1 yr-1 Med: $1765 ha-1 yr-1 High: $2210 US ha-1 yr-1
Pywell *et al.*, (2015)	Ecological intensification: Increasing levels of habitat creation around arable fields (control (no habitat), basic, high)	↑						↑	Not applied

2.5 Literature review summary

In summary of the literature reviewed here it is evident that there are number of different forms of NC within agri-environments including soil, biodiversity and freshwater NC. The quantity and quality of these NC assets, alongside other anthropogenic and natural drivers, influences how the EFs and combined with other forms of capital determine the flow of ES benefits to humans. Despite growing recognition of the importance of NC within agri-environments it is widely acknowledged that the intensification of agriculture has had a significant impact on NC extent and condition, severely affecting the flows of ES.

The NC approach presents a framework for evaluating land management decisions in a holistic way, identifying both the public and private benefits and trade-offs that might arise from a change in land management. It offers a solution to, 1.) Identify land management practices that reduce the destructive impacts of intensive agriculture and 2.) Bring land management decisions into a framework that could align well with accessing future agri-environment schemes in the UK (i.e. public money for public goods). However, despite the growing interest in the application of the NC approach to land management decisions it has rarely been conducted at management appropriate scales. This is perhaps unsurprising given the current challenges presented in undertaking the NC approach. One of the largest challenges to overcome appears to be meeting the resource and expertise requirements needed to build an understanding on, 1.) How NC condition and EF are impacted by specific land management practices, 2.) How NC condition in a given state impacts EF and the subsequent flow of ES and 3.) How ES that flow from the agri-environment can be valued.

Studies that have applied the NC approach at the farm scale have typically used existing data (where available) and have inferred changes in ES flows based on other research (i.e. value transfer method). No known farm-scale studies presented in the UK to date have focused on collecting spatially specific primary data to inform the NC approach across a suite of different ES flows. There is therefore a need for further research that builds understanding on the practical application of the NC approach at management appropriate scales. Importantly this research needs to advance knowledge on the metrics, methods and tools that can be useful in evaluating NC baseline condition, measuring EF and valuing ES.

The NC approach can be put to good use in evaluating contested land management practices that could deliver greater benefits for society. As presented in this review there are a number of land management practices that have been associated with delivering ES benefits beyond just food production. Field scale practices such as the incorporation of grass leys, cover crops and legume crops or changes in tillage regimes have been presented alongside systematic changes in agricultural approaches such as the shift to organic agriculture to improve ES delivery. The understanding of the role of these practices in systematically improving NC condition and enhancing the delivery of ES, however, is still not fully understood.

This literature review formed the foundational base for the development of the PhD objectives (presented in Chapter 1) which were then refined in collaboration with Clinton Devon Estate. The review was used to develop the methods introduced in the following chapter and refine the research questions applied in the four primary research chapters. Where necessary further specific literature is bought in to introduce and justify the development of each research chapter.

Chapter 3: Site description, methodological framework and introduction to methods

This chapter first introduces the study site before covering the processes involved in identifying the key measurements of NC, EF and ES investigated in this research and the NC approach framework applied throughout. This is followed by a justification of the importance of each measurement of NC. The chapter finishes by providing a detailed explanation of some of the key field and lab methods used in two or more of the data chapters. The selection process for field sites, chapter specific methods and statistical methods are outlined separately in each of the four research chapters (Chapters 4 – 7).

3.1 Study site

The study was conducted on Clinton Devon Estate (CDE) a lowland farming estate in South West England (Figure 3.1). The 7,558ha estate offers the advantage of covering a large part of the lower River Otter catchment and containing a range of agricultural land management practices. The farmed part of the estate (4,590ha) comprise organic (1600ha) and conventional (2990ha) farms, which extend over the continuum of scales of intensity including: dairy, beef, pigs, arable, mixed and some vegetable agriculture. A large holding on the estate (ca. 900ha), the Home Farm (including Farm 3 and 5 in Figure 3.1), is farmed in-hand and was converted from conventional arable and dairy to organic arable and dairy in 2007. The agriculture before conversion was typical of neighbouring conventional farms, with winter cereals, maize, rye-grass and clover silage leys and improved pasture.

The estate is underlain with Permo-Triassic and Carboniferous sandstone and siltstone geology. The sample sites for the study are under the Bromsgrove soil association (0541b) - a well-drained coarse loamy to sandy soil, which is widespread in the UK (covering 714 km^2 of agricultural land), mostly across the South West and Midlands (Landis, 2018). There are three soil series within the Bromsgrove association (Bromsgrove, Hodnet and Eardiston) which are mostly brown earths and are predominantly classified as eutric chromic endoleptic cambisols within the World Reference Base for Soil Resources (Landis, 2018). The Bromsgrove association soils are good agricultural soils and are easily worked facilitating a range of cropping regimes. The soils should readily accept winter rainfall even on steep slopes but can limit the available water for crops in summer. The soils can therefore be vulnerable to drought and grass growth, in particular, is often limited during summer (Landis, 2018). Mean annual temperature and precipitation for the study area is 10.7°C and 825mm, respectively. The majority of the estate soils drain to the Otter Sandstone or Budleigh Salterton Pebblebed Heaths Formation. These aquifer units represent a major groundwater drinking water resource in South West England (Bearcock and Smedley, 2012).

Figure 3.1 shows the field sites and farms used in this study. Data from the main study fields is utilised primarily in Chapter 4 and Chapter 6. The sub-set study fields are used for more detailed analysis in Chapter 5 and Chapter 7. Further details on site selection criteria and experimental design are described within the respective chapters.

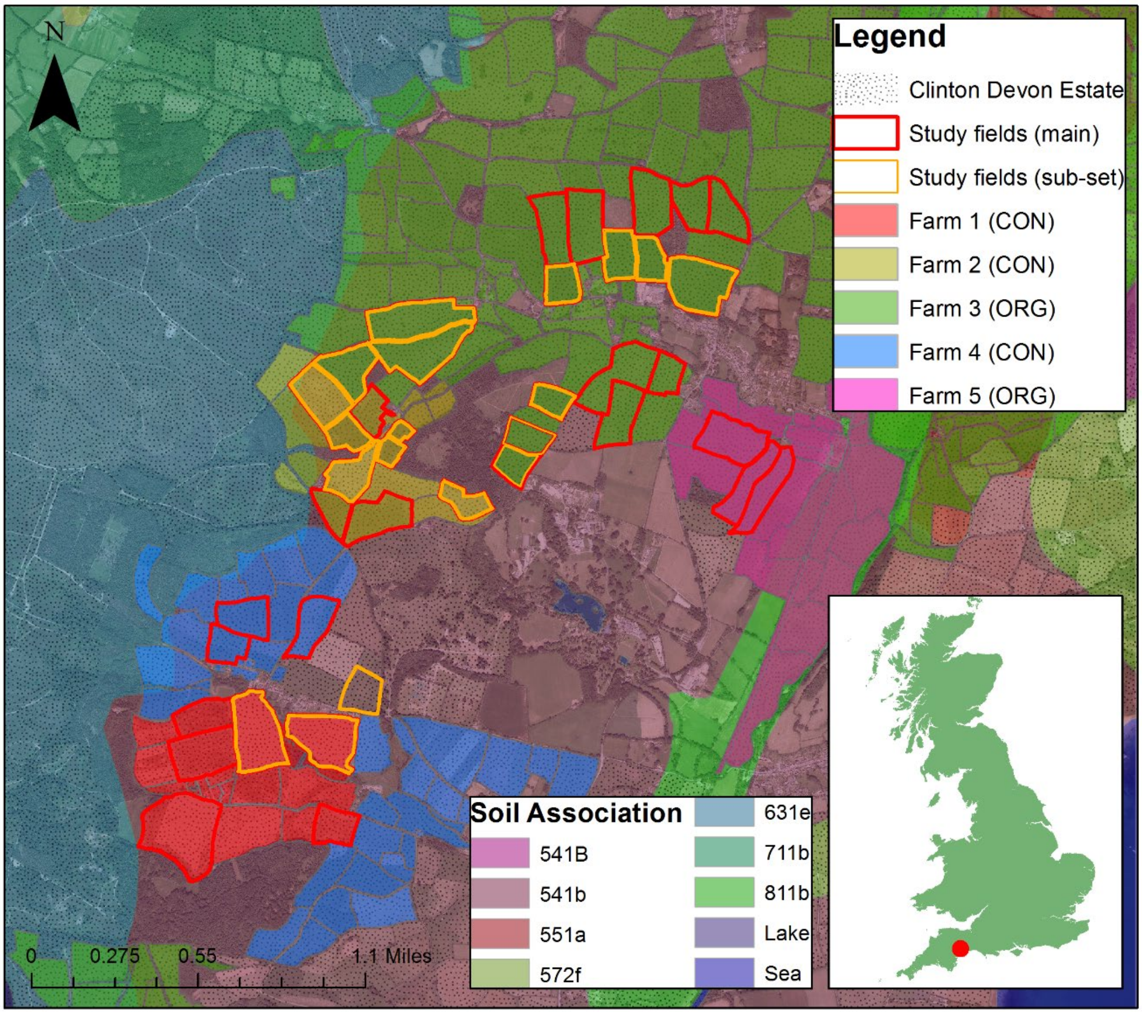


Figure 3.1: Map of the study area in the lower Otter valley, South West England. The map shows the coverage of Clinton Devon Estate, the soil association for the area, farms (farms 1 – 5) and the study fields used in the study. Data from the main study fields are included in Chapter 4 and Chapter 6, whereas data from the sub-set study fields are used for more detailed analysis in Chapter 5 and Chapter 7. Acronyms used: CON (conventional farm) and ORG (organic farm).

3.2 Natural capital and ecosystem services at Clinton Devon Estate

The initial phase of this study involved a process of collaboration with CDE to identify and categorise what forms of NC are present on CDE and the flows of ES from it. This process involved establishing which NC assets were a priority to understand on the estate and which could be practically measured given the available resources. Alongside this process ran a review of the literature to understand the importance of potential NC assets and ES flows within the wider context of agricultural landscapes. Figure 3.2 displays a non-exhaustive list of NC on the estate and how these might link to final goods and services from which an economic value could be derived.

From this process, three key NC components and four important ES emerged that became the focus for the study. Forms of NC were: 1) soil NC; 2) groundwater NC; and 3) functional biodiversity NC (specifically pollinator stocks). The flows of ES selected were: 1) climate regulation (based on carbon storage); 2) drinking water provision (based on nitrate leaching and groundwater contamination): 3) crop production; and 4) pollinator services. All types are described in more detail in the following sections.

It is recognised that these selected measurements or metrics are not a comprehensive suite of all ES or disservices that arise in agricultural landscapes. However, the purpose of the study was to gather high-resolution data to understand in detail the impacts of land management on each tier of the NC approach and the assessments that can be made from this. The intention was not to apply the more broad-brush approaches using existing data to understand a wider suite of NC and ES. Further details on the importance of the metrics selected and the rationale behind each is presented in Section 3.4.

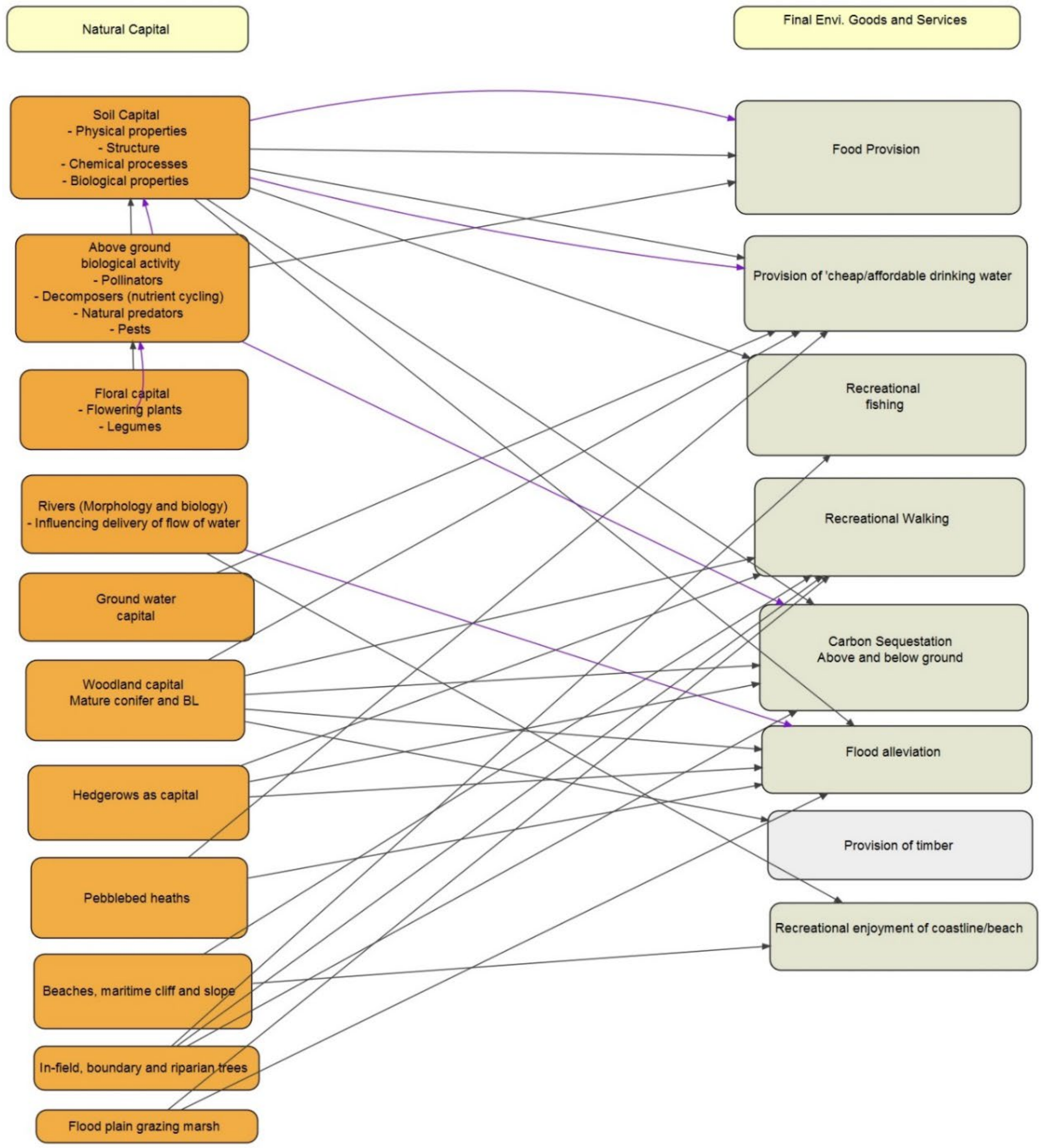


Figure 3.2: The output of the initial brainstorming exercise in collaboration with CDE, reviewing NC on the estate and its connections with final ecosystem goods and services.

3.3 Methodological framework for evaluating natural capital at Clinton Devon Estate

The NC approach framework introduced in Section 2.1 is applied throughout this study to evaluate land management decisions at CDE. Chapters 4 and 6 focus primarily on Tier 1 of the framework, evaluating differences or change in NC stocks, albeit alongside the production of crops. Chapter 5 focuses on Tiers 1 and 2 considering the benefits of incorporating measurements of EF, as well as

Tier 1 NC stocks. Finally, Chapter 7 brings all of these together, applying the complete framework and incorporating measurements from tiers 1, 2 and 3.

The framework used in this study identifies properties of NC, EF and ES that are considered within the literature to be connected. It has been adapted from Haines-young and Potschin (2008) but is very similar to other frameworks linking stocks, functions and services (e.g. Dominati, Patterson and Mackay (2010)). The principles behind the NC approach framework are described in more detail in Chapter 2, Section 2.1.

Figure 3.3 places the NC stock and the flows of ES identified at CDE within the NC approach framework. It provides an insight into the measurements used across the study to quantity NC condition, EF and, where possible, the value of ES. Further detail is provided in the following two sections.

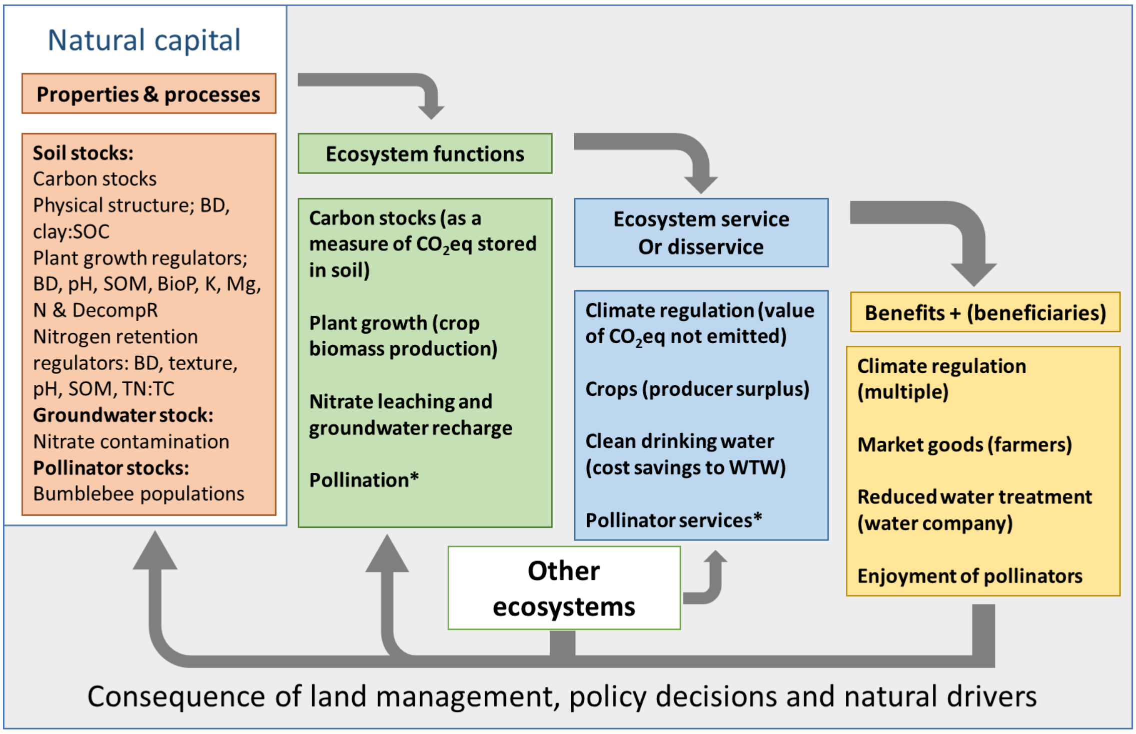


*Figure 3.3: Natural capital approach framework applied showing the component parts and how they theoretically link to deliver benefits to humans. Measurements applied include, 1) Soil carbon stocks relating the function of carbon dioxide storage/sequestration and the service of climate regulation, 2) Soil NC properties that support plant growth relating to the function of crop growth and the provision of food, 3) Soil NC properties and the condition of groundwater NC considering soil nitrogen retention and the provision of clean drinking water, 4) Pollinator stocks and related pollination of crops and wild plants contributing to food production and biodiverse areas. This framework is adapted from Haines-Young and Potschin (2008). Acronyms used: BD (bulk density), SOM (Soil organic matter), BioP (bioavailable phosphorus), K (potassium), Mg (magnesium), N (nitrogen), DecompR (decomposition of organic material) and TN:TC (nitrogen to carbon ratio). *shows measurements that could not be made in this study.*

3.4 Detailed overview of the natural capital and ecosystem services included in this study

The following section introduces the four key pathways that were selected to be tracked through each tier of the NC approach framework: 1) carbon storage and climate mitigation; 2) soil fertility, crop growth and provisioning services; 3) soil nutrient retention, nitrate leaching and the provision of clean drinking water; and 4) pollinator stocks and pollinator services. Building upon the context provided in Chapter 2 the following sub-sections identify why each of the four pathways are important and what measurements or metrics are commonly used to quantify them.

Whilst the four pathways are referred to at different stages of chapters 4, 5 and 6, they are all bought together in Chapter 7 to evaluate the complete application of the NC approach. Chapter 4 and Chapter 6 focus primarily on individual NC components (i.e. soil and pollinators respectively), whereas Chapter 5 incorporates measuring indicators of both soil and water NC and EF. The assessment of soil and water NC and EF were not separated into different chapters due to the multiple connections between the state of water quality and that of soil condition and soil function (e.g. nutrient retention and nitrate leaching to groundwater). This is typical of other studies investigating soil-based ES, which also incorporate measurements of soil functions linked to freshwater NC and ES (e.g. surface and groundwater condition and drinking water supply). See Section 2.2.1 and 2.3.1 for further insight into the overlap between soil and water ES.

3.4.1 Carbon storage and climate mitigation

There is considerable interest in the capacity for soils to sequester carbon and mitigate climate change (Minasny *et al.*, 2017) and it is recognised that land management is critical in determining whether soils act as a net sink or source of greenhouse gasses (FAO, 2017; Paustian *et al.*, 2019a). Carbon sequestration has the potential to draw down significant amounts of atmospheric carbon dioxide (Minasny *et al.*, 2017) which, when stored within soil, can be measured as carbon stocks. Expressing those soil carbon stocks as carbon dioxide equivalents (CO_2eq.) permits valuation as avoided greenhouse gas emissions. That valuation process can progress in a number of ways, adopting different assumptions about carbon costs/prices (Bartkowski *et al.*, 2020). Each approach estimates avoided emissions (tonnes CO_2eq.) and then multiplies the figure by the chosen carbon price (£ t $^{-1}$), in order to arrive at a value for the ES of climate regulation. In the UK, the Department for Business, Energy and Industrial Strategy (BEIS) now publish a list of non-traded carbon prices which are advised in the use of projects evaluating land use change scenarios (Hurst, 2019). In this study, soil carbon stocks were selected to evaluate soil NC, the storage of CO_2eq. as a proxy of soil function and BEIS non-traded carbon prices to value the service of climate regulation to society.

3.4.2 Soil fertility, crop growth and provisioning services

The provision of food and fibre is a critically important ecosystem good and with global population set to exceed 9 billion by 2050 it is important that we understand the capacity for different agricultural systems to provide crops that support human nutrition (Muller *et al.*, 2017).

Given the almost single-minded focus of agricultural systems on the maximisation of crop outputs, it is perhaps unsurprising that soil properties linked to plant growth are frequently measured in the evaluation of soil condition. Nutrients such as nitrogen (N), phosphorus (P) and potassium (K) are all of critical importance for plant growth (Brady and Weil, 2008; Dungait *et al.*, 2012), along with other nutrients such as sulphur (S), calcium (Ca) and magnesium (Mg). Likewise, soil organic matter (SOM) and its decomposition through biological activity facilitates the release and storage of these nutrients (Brady and Weil, 2008) and soil structure is important in determining air and water availability. N, P, K, Mg, SOM and measurements of soil structure can therefore be considered important properties of soil NC condition, contributing to the soil function of plant growth. Vegetation biomass is a common measure of plant growth and productivity and it is used in this study to assess soil function of plant growth and evaluate what conclusions might be made regarding provisioning services using these measurements.

In order to value provisioning services, crop yields are frequently multiplied by crop market prices (e.g. Sandhu *et al.*, 2008; Fan, Henriksen and Porter, 2016; Silcock *et al.*, 2018). Of course, prices only represent the revenue generated by crops and not the net benefits of farming. Rather the actual welfare gain of the producer from food production is given by producer surplus, which is simply revenue net of production costs. This measure is recognised by economists as the theoretically correct measure of the value of productive activity to be used in social cost-benefit analysis. It is used in this study as a measure of the value of the provisioning service of crop production and is calculated as the total revenue made through the sale of goods minus the total costs of production. Whilst gross margins for crops are often reported in the industry handbooks (e.g. Nix, 2018; ORC, 2017) and are often used as proxy values for provisioning services (e.g. Fezzi *et al.*, 2014), these measurements do not include the full field costs of producing the crops: for example, land preparation, crop management and harvest costs. The measure of producer surplus applied in chapters 6 and 7 is essentially a measure of operating profit; that is to say the revenues generated by the sale of the crop minus all field-based management costs incurred in the crops production.

3.4.3 Soil nutrient retention, nitrate leaching and the provision of clean drinking water

The provision of drinking water is an important ES (Keeler *et al.*, 2012; Grizzetti *et al.*, 2016) and yet the sufficient supply of clean drinking water is under threat globally (Boretti and Rosa, 2019). A particular issue is with nitrate. Nitrate contamination has both environmental and economic consequences, resulting in the loss of valuable farm nutrients for land managers and increasing drinking water treatment costs and prices paid by consumers (Stuart and Lapworth, 2016). In the UK, groundwater is a particularly important resource, supplying 30% of the nation's drinking water (EA, 2018). However, the quality of groundwater aquifers has declined significantly as a consequence of increased nitrate leaching from agricultural land (Wang *et al.*, 2016). Aquifers now regularly exceed drinking water standards laid out in the EU Drinking Water Directive (Stuart and Lapworth, 2016).

Soil NC is recognised as playing a critical role in filtering nitrate and storing nitrogen, with soil condition and soil management determining losses to surface and groundwater resources (Beaudoin *et al.*, 2005; Harris *et al.*, 2006; Knudsen *et al.*, 2006). Soil structure, texture (particularly clay content), SOM, pH and cation exchange capacity (CEC) are all cited as important soil properties that influence pollutant retention (including nitrate) (Bone *et al.*, 2010; Makó *et al.*, 2017). Management relating to the timing, type and amount of nitrogen input is also critically important in determining nitrate loss. The function of nitrate leaching can be measured through methods such as lysimeters or porous pots, providing data on the quantity of nitrate lost from field sites and an indication of the scale of groundwater contamination under different land management. Nitrate leaching is used in this study to measure soil function and a suite of nutrient retention indicators are applied to determine the suitability of soil NC conditions to retain nitrate. This is combined with data from the water company on groundwater nitrate contamination to determine the condition of the groundwater NC stock.

Quantifying the costs of water treatment is a common method used in the valuation of the ecosystem disservices from agriculture (Pretty *et al.*, 2000; Keeler *et al.*, 2012; Graves *et al.*, 2015). Harris *et al.*, (2006), in their work on the economic valuation of soil functions suggest that cost-based approaches to valuing the “attenuating pollutants” function (which includes nitrate) can be used where changes in soil management have implications to the water sector. Management strategies to reduce nitrate leaching (below the drinking water limit of 50 mg NO_3 l^{-1}) offer savings for water supply companies, whilst improving the quality and provision of drinking water (Keeler *et al.*, 2012). A cost-based approach is applied in Chapter 7 to value the disservice of nitrate leaching and groundwater contamination.

3.4.4 Functional biodiversity; Pollinator stocks and pollinator services

Pollinator populations are important NC stocks (Hanley *et al.*, 2015) and pollinating insects (such as honeybees, bumblebees, solitary bees, wasps and hoverflies) deliver a critical ES in pollinating a number of commercial crops and wild plants (Klein *et al.*, 2007; Potts *et al.*, 2010; Ollerton, Winfree and Tarrant, 2011). Bumblebees are considered the most important wild pollinators in the UK in terms of pollination of wild plants and crops (Wood *et al.*, 2015a) and they are identified as a keystone species (Goulson *et al.*, 2011). They are used in this study as an indicator to assess pollinator stocks and to consider pollination and pollinator services.

Despite recognition of their importance, bumblebee populations are rarely measured, on account of complexity and resource demand (Wood, Holland and Goulson, 2015; Becher *et al.*, 2018). Whilst transect methods have been used to evaluate abundance differences between different field treatments, it is unclear whether these differences show real population change or whether results map the redistribution of bees within a landscape (Wood *et al.*, 2015). Bumblebee colony density provides a useful measure of bumblebee populations (Wood *et* *al.*, 2015) and recent advances in bee models (Becher *et al.*, 2014, 2016, 2018) present a resource-efficient means to quantify this under different land use scenarios. To understand how pollinator stocks might have changed under different land management scenarios, the recently published BEE-STEWARD software (Twiston-Davies, Becher and Osborne, 2021) was used.

The BEE-STEWARD software brings together Bumblebee-HAVE (Becher *et al.*, 2018) and BEESCOUT models (Becher *et al.*, 2016) in a user friendly interface and can be applied to estimate pollination services for insect pollinated crops (Becher *et al.*, 2018). The results provide information of bee visitation (or lack of) to crops. In principle, market valuation of pollination services can then be calculated by estimating the proportion of yield loss under different visitation scenarios (based on the Dependence Ratio – DR, of particular crops) (Hanley *et al.*, 2015). In this study, a number of problems were encountered with this approach specific to the case-study site. These are discussed further in Chapter 7.

3.5 Overview and justification of the NC, EF and ES measurements selected

Following the assessment of the measurements commonly used to assess the four selected pathways, a suite of metrics were selected (Table 3.1). Justification for most of these has already been provided above (Section 3.4) but further detail is given below on the importance of BD and n-potential (clay:SOC ratio) as a measurement of soil structure and decomposition rate as a proxy for nutrient and carbon cycling.

Table 3.1: A summary of the metrics used to measure NC condition, EF and ES in this study. It was not possible to measure pollination services in this study landscape – explanations for this are given below.

Metrics for NC	Metrics for EF	Metrics for ES
Soil: SOC, SOM, BioP, K, Mg, total N, total C, TN:TC, pH, texture, N-potential (ratio clay:SOC), SOM decomposition rate (TBI)	Carbon stocks Crop biomass production	Value of stored CO_2eq Crop producer surplus
Groundwater: Current state of nitrate contamination	Nitrate leaching	Cost savings in water treatment
Pollinator stocks: Bumblebee populations using BEE-STEWARD software	Not possible in this landscape	Not possible

Bulk density (BD): the mass of a unit volume of dry soil is an important indicator in the level of pore space (i.e. the space available for air and water) within a soil, providing information on the level of compaction (Cardoso *et al.*, 2013). This measure is frequently used in soil-based ES studies (Greiner *et al.*, 2017). Soil compaction has been associated with significant flooding and crop productivity issues (Graves *et al.*, 2015). BD is also important in the quantification of soil carbon stocks, important in understanding the climate regulation capacity of the soil.

N-potential: introduced by Merante *et al.,* (2017), is a measure of soil stability and the carbon storage potential of a soil. It is calculated as the ratio of clay (%) to SOC (%). Soil stability is closely linked to the content of SOC and fine soil particles (clay and silt), which become associated in the development of soil aggregates (Merante *et al.*, 2017). A high n-potential (>10) suggests a low SOC relative to clay content, suggesting the presence of non-complexed clays which are more easily dispersed in water and more vulnerable to soil degradation (e.g. compaction or erosion). These soils have the capacity to store more carbon. A low n-potential (<10) suggests a high SOC relative to clay content, showing that most of the clay is likely to be complexed with SOC which increases soil stability (Merante *et al.*, 2017). These soils have less capacity to store more carbon.

Other studies also use the ratio of clay to SOC as a metric to assess soil condition. For example Prout *et al.*, (2020) define bounds for the clay:SOC ratio with reference to soil degradation in the UK. They classify a ratio of less than 8 to refer to very good soil condition, 8 - 10 refer to good soil condition, 10 – 13 as moderate and greater than 13, as degraded soil condition. These classifications are used to interpret soil data in Chapter 4 but for consistency N-Potential is used to refer to the ratio of clay to SOC throughout the thesis.

Tea Bag Index (TBI): quantifies the decomposition of organic material by soil biota and provides an important insight into nutrient cycling and biological activity of the soil (Keuskamp *et al.*, 2013; Ghaley *et al.*, 2014a). The breakdown of organic material (e.g. plant litter or farm manures) is critical for the development of plants, ensuring that nutrients become available through decomposition and mineralisation (Ghaley *et al.*, 2014a). The processing of organic material and the role of soil biota is also integral to the carbon cycle influencing whether soil becomes a carbon sink or source (Paustian *et al.*, 2019a; Ray *et al.*, 2020). The TBI is described in full detail in Keuskamp *et al.*, (2013).

3.6 Introduction to methods

This section does not provide a comprehensive description of all methods used in this thesis but does cover those that are frequently used over two or more chapters and require more detailed explanation. The purpose of this section is to provide a reference point for the next four data chapters in order to reduce repetition of methods as much as possible. It covers the detailed methods on soil sample collection and analysis, organic matter decomposition rate, nitrate leaching, crop biomass quantification and crop costs of production.

3.6.1 Soil sample collection and analysis

Details below are relevant, to differing degrees, to chapters 4, 5 and 7.

All soil samples were collected using a metal corer (15cm deep x 4.8cm wide) following the Countryside Survey Soil Manual (Emmett *et al.*, 2008). Samples were placed in plastic bags, returned to the lab, weighed to establish wet weight and then dried at 50^{0}C until they reached a stable dry weight (measured to two decimal points). A stable weight was defined as a weight change of less than 1% from the previous measurement, after a minimum of 12 hours of drying.

Soils were ground using a pestle and mortar and passed through a 2mm sieve. Roots and stones >2mm were removed. Stones were weighed and the volume was determined through water displacement, as per UK Countryside Survey Soil Manual (2007) (Emmett *et al.*, 2008).

Soil BD was then calculated as:

$$BD = \frac{Dw - Sw}{Cv - Sv}$$

where *Dw* is the dry weight of the core (g), *Sw* is the stone weight (g), *Cv* is the volume of the core (271.43 cm^3) and *Sv* is the volume of stones (cm^3).

The dried sample was split and one part sent off for analysis (lab accreditation; BS EN ISO/IEC 17025.) of pH, SOM, bioavailable-P, Mg, K and particle size distribution (PSD). Soil pH was measured in a temperature controlled environment potentiometrically, by first creating a slurry of deionized water and soil (ratio of soil to water 1:2.5). Bioavailable-P was extracted from a sub-sample of soil using 0.5M sodium at pH 8.5 (Olsen-P extraction), shaken at 20^0C for 30 minutes. The concentration of bioavailable-P in the filtered extract is then determined by flow injection analysis/colorimetry (spectrophotometer at 880nm) using an ammonium molybdate reagent. Bio-available K and Mg were extracted from a sub-sample of soil using a 1M ammonium nitrate, shaken at 20^0C for 30 minutes. Samples were then filtered and the concentration of K and Mg in the extract determined using atomic absorption spectrometry. Instrumentation for available nutrients was calibrated using commercial phosphate, potassium and Mg standards traceable to the SI unit. SOM was determined by a sub-sample first being dried at 105^0C and then through loss on ignition (LOI) at 430^0C. The sub-sample was weighed before LOI to give an initial weight and then weighed again once the sample had cooled in a desiccator to calculate percentage loss. Loss was taken as SOM (%).

Using the retained sub-sample of dried soil, total C and total N analysis was conducted. The sample was initially ground to 500μm and homogenised. Roughly, 19mg was then weighed into a tin combustion capsule before being processed via an elemental analyser (Thermo Scientific FLASH 2000) (as in Glendell *et al.*, 2014)..

SOC was calculated as $SOC(\%) = SOM(\%) * \ 0.52$ where SOM is the percentage SOM (from LOI) and 0.52 represents the mass of carbon in SOM. The conversion figure of 0.52 was determined specifically for the soil samples in this study by calculating the mean ratio of total carbon to SOM (data in Appendix A.1). A standard ratio of 0.58 is often used to convert SOM from loss on ignition to SOC. However, it has been identified that this can significantly overestimate carbon stocks (Jensen *et al.,* 2018).

N-potential, was calculated as the ratio of clay (%) to SOC (%) (Merante *et al.*, 2017). Clay content was derived from particle size distribution analysis using a Laser Diffraction Particle Sizer to calculate sand (2.00 – 0.063mm), silt (0.063 – 0.002mm) and clay (<0.002mm) components.

Soil carbon stocks (t ha^{-1}) were calculated following Poeplau, Vos and Don (2017) as: $C\ stock\ = SOC * BD * d$

where SOC is soil organic carbon (%), BD is the BD of the soil (g cm^{-3}) (corrected for stone content) and *d* is the depth of the soil core (15cm).

3.6.2 Quantifying organic matter decomposition rate

Details below are relevant to chapters 5 and 7.

Organic matter decomposition was determined using the standardised and widely used Tea Bag Index (TBI) method (Keuskamp *et al.*, 2013). The TBI is a simplified litter bag test using commercially available Lipton Rooibos ™ and Green tea ™. The two different tea types (Figure 3.4: far right) have a different decomposability with green tea decomposing rapidly and rooibos tea, with more recalcitrant material, decomposing at a much slower rate. The use of two different types of material allows an assessment of a two-phase decomposition curve from a single incubation time frame (i.e. eliminating repeated retrieval and reweighing of samples). Using information on mass loss and the hydrolysable component of the tea, it is possible to determine a stabilisation factor (S) and decomposition rate/decay constant (k).

Eight pairs of uniquely labelled, pre-weighed tea bags (to the nearest mg) were buried at 8cm depth at each site following (Keuskamp *et al.*, 2013). The intention was to retrieve at least five pairs of tea bags from each site, with contingency for tea bag losses or damages on retrieval. Of the 54 field sites utilised in Chapter 5, 51 had at least five retrieved pairs, with three sites having four retrieved pairs. Tea bags were installed in May 2019 and were retrieved 52 days later. Whilst the Keuskamp *et al.,* (2013) method advises a 90 day incubation period, a shorter period was scheduled to fit in between maize drilling and whole crop harvesting across the different sites. Other agricultural experiments have also used reduced incubation periods. For example, Tóth, Hornung and Báldi (2018) use a one month incubation and Welsch *et al.*, (2019) a 30 day incubation period.

Upon retrieval, the tea bags were air dried, loose soil was brushed off and they were transferred to a pre-weighed paper case. Fine roots were removed with tweezers before samples were oven dried for 48 hours at 70°C. Once dry, samples were allowed to cool for 10 minutes before being re-weighed. The mass of the paper case (adjusted for moisture content) was subtracted from the total, to give the final tea weight.

TBI k was calculated for each site based on the five (at three sites only four) replicate tea bag decomposition results. TBI S calculations, however, were discounted on account of the shorted incubation time necessary for this study. Lower microbial activity and shorter incubation times make the interpretation of TBI S less reliable (Keuskamp *et al.*, 2013).

Figure 3.4: Field and lab work photos from left to right: Installing porous pots using a mechanised auger in October 2018, maize crop yield sampling in September 2019 and sorting contents of green and red tea ahead of drying and final weighing in August 2019.

3.6.3 Quantifying nitrate leaching

Details below are relevant to chapters 5 and 7.

Porous pots were selected as the best method to determine nitrate leaching through a drainage season and to identify when leaching occurred. Porous pots have a ceramic chamber for collecting the sample and a plastic chamber above. One tube enters the plastic chamber and the other enters the ceramic chamber below. The unit is sealed allowing the ceramic chamber to be held under vacuum, increasing the matric potential in the adjacent soil and drawing in soil pore water.

The sample (up to 25ml) can then be extracted by releasing the vacuum and drawing out the sample using a syringe. Porous pots or ceramic cups are widely used for measuring nitrate leaching from agricultural land (Lord and Sheperd, 1993; Goulding *et al*., 2000; Webb *et al*., 2001; Silgram and Chambers, 2002; Cooper *et al*., 2017) and have been applied to compare nitrate leaching from organic and conventional rotations (Benoit *et al.*, 2015).

Porous pots were installed in October 2018 at six field sites using a mechanised auger following ADAS (2005) (Figure 3.4: far left). Ten pots per field (2m spacing between each) were installed 90cm below the field surface. This depth is considered to be below most measured rooting depths of commercial crops. It is assumed that any nitrate in the soil water solution at this depth has effectively been lost to the crop and therefore will be leached from the field and eventually enter the groundwater aquifer.

Soil pore water collection and processing:

To estimate nitrate losses over the season, samples were collected from the porous pots every two weeks or wherever possible after 25mm of rainfall, during two seasons: November 2018 and April 2019; and October 2019 and March 2020. Porous pots were held under vacuum and a sample was extracted using a 60ml syringe in accordance with ADAS (2005). Sampling commenced after the soil moisture deficit (SMD) for the region rose above 25mm (ADAS, 2005). In 2019, sampling ceased after SMD fell below 25mm. In 2020, the last sampling date was dictated by the start of the national lockdown enforced due to covid-19. The lockdown did not negatively impact the quality of the nitrate leaching calculation.

Extracted samples were collected in sterile plastic tubes, kept cool and returned to the labs. Here they were refrigerated and analysed within 48 hours of collection. Samples were filtered and then NO_3-N was determined colourimetrically using a continuous flow auto-analyser 3 (Seal Analytical, Southampton, UK).

Data collected every two weeks on soil pore moisture nitrate concentrations was used in conjunction with daily drainage volumes from the ADAS IRRIGUIDE model (Baily and Spackman, 1996) to estimate the cumulative loss of nitrate from organic and conventional field sites over each drainage season.

Modelling drainage and nitrate leaching using ADAS IRRIGUIDE:

ADAS IRRIGUIDE (Baily and Spackman, 1996) is a field scale water balance model and was used[3] to estimate drainage from each field through the 2018-19 and 2019-20 seasons. The model requires soil type, crop details, crop canopy cover, rooting depth and daily agrometeorological data to estimate drainage volumes, runoff, evaporation and transpiration (ADAS, 2021). The model has been used to underpin key Defra projects in the UK (Lord *et al.*, 2007), in other academic studies (Webb, Harrison and Ellis, 2000; Anthony *et al.*, 2009) and is widely used by Wessex Water to predict nitrate leaching from agricultural fields.

Crop canopy cover was calculated during each sampling visit. 10 photos from ca.1m height were taken in each field, immediately adjacent to each porous pot site of the crop/field surface. The photos were then processed using the Canopeo phone application (Patrignani and Ochsner, 2015) to calculate mean field canopy cover.

Daily rainfall for the model was derived from the NIMROD system (Met Office, 2003) for each field site using rainfall radar data. NIMROD data are provided as gridded total rainfall with resolutions of 1km and 5min respectively. Cumulative daily rainfall (mm) was calculated for each site for 2018, 2019 and for the beginning of 2020. Data download and conversion was conducted using Python 3 following the method in (Puttock *et al.*, 2021)[4].

3.6.4 Quantifying crop biomass production

Details below are relevant to chapters 4 – 7.

All yield data were collected in 2019, immediately prior to harvest (or as close as possible). To assist in comparisons between sites, all cereal, grass and maize crops were treated as harvested forage crops. Biomass collection, therefore, aimed to collect only vegetation that would also be collected by the forage harvester. Vegetation was not cut to ground level but to the planned forage harvester cutter bar height, typically 8-10cm for grass and whole crop cereals and 20cm for maize.

---

[3] IRRIGUIDE modelling runs were kindly conducted by experienced staff at Wessex Water following permission being granted by ADAS, who administer the model.

[4] Data download and conversion was kindly conducted by Hugh Graham (University of Exeter).

The same sampling sites as used for soils were located in the field using a hand-held GPS (Nomad Trimble, Sunnyvale, CA, USA). Silage, hay, winter cereal and spring cereal were collected from each field site using a 0.5m x 0.5m quadrat. Vegetation was cut, bagged, a fresh weight taken in the field and then returned to the lab for drying. Samples were collected before each silage or hay cut for the whole 2019 season and the cumulative yield calculated.

The harvest of three fields (21 sample points) were missed on the Home Farm for their first silage cut due to miscommunication from the farmer. As this was an important omission, linear modelling was conducted to predict the likely volumes. The model used data from the second cut silage volumes to predict first cut silage volumes. The model was constructed using data on first and second cut silage volumes from eight fields where both sets of data had been successfully collected (33 points). The model identified a significant relationship between first and second cut silage (p = 0.001) and had an r^2 of 0.29. The model outputs can be seen in Appendix A.2.

As maize is a row crop, a different method of yield estimation was conducted, following Steinhilber, Shipley and Vvedenskaya (2016). The method aims to calculate yield across an area that represents 1/1000th of an acre based on the row spacing of the maize crop. Each site was navigated to using a hand-held GPS (Nomad Trimble, Sunnyvale, CA, USA) and a centre marker placed (Figure 3.4: centre). A distance, specified by the row crop spacing, was then measured in either direction from the centre marker along a row of maize. Markers were placed at each end. The total number of maize stems was recorded along the transect and every fourth stem was selected and cut. The stems were bagged, weighed in the field and an average stalk weight calculated. A single whole stem, considered representative of the site, was then bagged and returned to the lab to establish dry matter content. Yield (t ha^{-1}) was calculated as:

$$Yield = \frac{(As * Ns * Dm\%) * 1000a}{1000b} * 2.47$$

where *As* is the average stalk weight (kg), *Ns* is the number of stalks, *Dm* is the calculated stem dry matter, 1000*a* is to multiply up to an acre, 1000*b* is the number of kg in a tonne and 2.47 converts the figure from t per acre to t per ha.

All vegetation was dried in the lab at 60^0C and reweighed until a stable weight was achieved. A final dry weight was taken and a percentage DM content calculated. Site measurements were then scaled up to calculate the DM yield in tonnes per hectare (t ha^{-1}).

3.6.5 Quantifying crop costs of production

Details below are relevant to Chapter 4, 6 and 7. Calculating crop costs of production is an important stage in the calculation of producer surplus in chapters 6 and 7.

Crop management data were gathered for each study field from the participating farmer. Costs for each management activity were either provided by the farm or they were estimated based on local contractor costs or data in the Nix Farm Pocket Handbook 2019 edition (2018). Data on farm management costings was used to calculate what the costs of cultivations and drilling, crop management (applying slurry, fertilisers or sprays) and harvesting would be for each field. Variable costs were calculated using data from the farmers on fertiliser and spray products they used and the standard prices reported in Nix (2018) for the product (or for the most similar alternative). Costs for sprays and fertilisers were calculated on the active ingredients as opposed to the actual product used. Conventional seed costs were taken from Nix (2018) and organic seed costs from the ORC Handbook (ORC, 2017). All establishment costs for grass leys (seed costs, cultivation, drilling etc) were annualised over three years (the estimated length of the grass ley). Variable costs were combined with fixed costs to calculate the total crop cost of production (CoP, £ ha^{-1}) for each site.

Chapter 4: Utilising existing land management records to explore the drivers of soil natural capital condition and productive output under different soil management intensities

4.1 Introduction

There is growing interest in the application of the NC approach and NC accounting at the farm and estate management scales, incorporating these frameworks into performance monitoring and decision-making. There has been a flurry of recent articles (EFTEC, 2019; Beedell, 2021; CLA, 2021; Harris, 2021; Norton, 2021) within the land management sector regarding the potential for adopting these approaches – outlining its potential to build a more holistic understanding of the private and public benefits that arise from farm NC, support sustainable decision-making and enable access to future funding streams.

An important first stage in the NC approach and NC accounting is conducting an asset register. This involves identifying forms of NC at the farm or estate scale and determining its condition (quality and quantity) (EFTEC, 2019; Faccioli *et al.*, 2020). A number of forms of NC have been identified as important on farmland including surface and groundwater (Bergkamp and Cross, 2006; Khan and Din, 2015), hedgerows (Wolton, 2018), trees, woodland (Trenbirth and Dutton, 2020) and functional biodiversity (e.g. pollinators) (Hanley *et al.*, 2015) but that which has perhaps received most attention is soil (Robinson, Lebron and Vereecken, 2009; Dominati, Patterson and Mackay, 2010; Hewitt, Hedley and Rosser, 2010; Robinson *et al.*, 2012, 2013, 2017; Dominati *et al.*, 2014, 2016; Hewitt *et al.*, 2015). Soil is widely regarded by land managers as being a critical NC asset that 'underpins everything' (Prager and McKee, 2014), providing the medium for plant growth and playing a role in flood, climate and water quality regulation. Whilst the importance of soil health is no new concept to land managers, there is a growing interest in the holistic condition of soil, beyond just the monitoring of plant nutrients that sustain the basic needs of crops (Prager and McKee, 2014).

Despite recognition of the importance of soil NC, the few studies that have attempted to apply the NC approach at the farm or estate scale, have typically failed to measure soil NC condition (e.g. soil carbon stock or soil fertility) (EFTEC, 2018; Kieboom, Silcock and Russ, 2018; Silcock and Russ, 2018; Silcock *et al.*, 2018). Indeed, Silcock *et al.*, (2018) in their evaluation of NC assessments at Glenlivet Estate, Den Farm and Ruthven Farm identify that in future NC assessments it would be useful to consider measuring indicators of NC condition. They suggest SOC, pH, BD, earthworm counts and plant nutrients (Cu, Zu, N, P, K, Mg) could be useful metrics to track soil NC condition over time. The lack of measurements of soil NC condition in previous farm or estate scale NC studies is likely to be due to the fact that collecting these data at the field to farm scale can be costly, time consuming and requires interpretation (Prager and McKee, 2014). It is also unclear exactly which metrics should be used to categorise soil NC condition and despite the suggested suite from Silcock *et al.*, (2018), there is no established set of metrics for the assessment of soil NC condition. Whilst existing soil datasets do exist, as presented by the Natural Capital Committee (Natural Capital Committee, 2017) and in government guidance (Enabling the Natural Capital Approach, (Defra, 2020a)), these are at insufficient resolutions to be of particular use at local scales. Existing soil data, for example, has been shown to be poorly representative of a number of soil properties when compared to empirical data and has prevented the detection of differences in soil conditions between different land use and/or management (Glendell *et al.*, 2014). Currently, existing soil data sets do not align with the spatial resolution needed by land managers to evaluate field or farm conditions or to inform decisions (Prager and McKee, 2014). To support land management at the local scale, empirical soil data collection is required to establish a meaningful assessment of baseline soil NC condition.

Field collected soil data are important to enable monitoring over time, benchmarking against other farms, the identification of areas/practices that require improvement (Prager and McKee, 2014) and in some cases (i.e. soil carbon), support economic valuation of the services or disservices delivered (Bartkowski *et al.*, 2020). Taken at the estate scale, data can be valuable in monitoring maintenance of estate assets across tenant farms. Some studies have identified differences in soil degradation based on land tenure, with increasing risk of soil degradation under shorter term tenancies (Walmsley and Sklenička, 2017; Eder, Salhofer and Scheichel, 2021). In the future, such data (e.g. soil carbon stocks) may even prove to be important in accessing agri-environment or industry funding schemes for environmental improvement such as carbon sequestration (Paustian *et al.*, 2019b). Alongside other data on ES, soil NC condition can be used to consider the synergies and trade-offs that arise across the farm or estate as a consequence of management.

Given the cost and resource required to collect data to categorise soil NC condition, it is rational to maximise the information that can be derived from it. Indeed, interpretation and analysis of the data can provide some interesting insights into practices that have the capacity to enhance or degrade soil NC conditions. Whilst monitoring over time may offer a more robust solution to determining the trajectory of soil conditions under different management, this takes time, particularly when considering the slow response of some properties to land management change (Bünemann *et al.*, 2018). Information from farm records on the specifics of farm management including cultivation history, cropping, organic matter and fertiliser inputs, can shed light on those management practices that might be sustaining or degrading soil NC condition. This information can be used to ask a number of questions that are both of relevance at local and wider scales; supporting targeted future monitoring (e.g. identifying appropriate metrics or further detailed research), assisting in farm management and policy scale decision-making, whilst contributing to gaps in academic understanding. In this study, this exploration was conducted through the application of a multi-model inference approach (Burnham and Anderson, 2002), which has been widely applied in ecology, using abiotic and biotic predictors to understand drivers of ecosystem properties (Grueber *et al.*, 2011; Hu *et al.*, 2014; North *et al.*, 2015).

There are a number of important questions that are of interest at the farm (i.e the genuine management) scale and which have relevance across the soil NC literature. This study aims to first characterise soil NC condition at a field and farm scale before using data collected, alongside farm management records and crop biomass data, to address three research questions:

1. **How does farm intensity impact soil natural capital condition and productive output?**
2. **What are the likely drivers of differences in soil natural capital condition across the study farms?**

3. **Does degradation of soil natural capital impact on productive output?**

Findings will provide valuable evidence at farm and estate scales, to support future management decisions including farming strategy, changes in tenancy agreements and monitoring environmental performance of in-hand and tenanted farm holdings.

4.2 Methods

4.2.1 Study Site

The study was conducted on Clinton Devon Estate (CDE) in south east Devon (Figure 4.1). The estate is described in Chapter 3, Section 3.1.

4.2.2 Experimental design

4.2.2.1 Farm selection and ranking of agricultural intensity

Three of the 24 tenant farms (referred to as Farms 1, 2 and 4) and two in-house farms (referred to as Farms 3 and 5) (Figure 4.1) were selected on the estate, representing the typical agricultural practices of the area. The farms were selected based on the underlying soil association (Bromsgrove association (0541b), the existence of farm management records, a willingness to be a part of the study and differences in farm management. Farm 1 is conventionally managed and characterised as having a fairly rapid and highly structured two to three year rotation (maize into winter cereal into an overwintered fast grass and back into maize). Farm 2 is also conventionally managed and has a more varied rotation with a mix of grass and arable crops, with fields put to grass depending on the needs of the farm and the perceived field conditions. Farm 3 is organically managed with a structured six-year rotation with three years in arable and three years in grass and clover ley. Farm 4 (conventional) and 5 (organic) are both dairy farms practicing paddock grazing on rye-grass and white clover with year-round grazing.

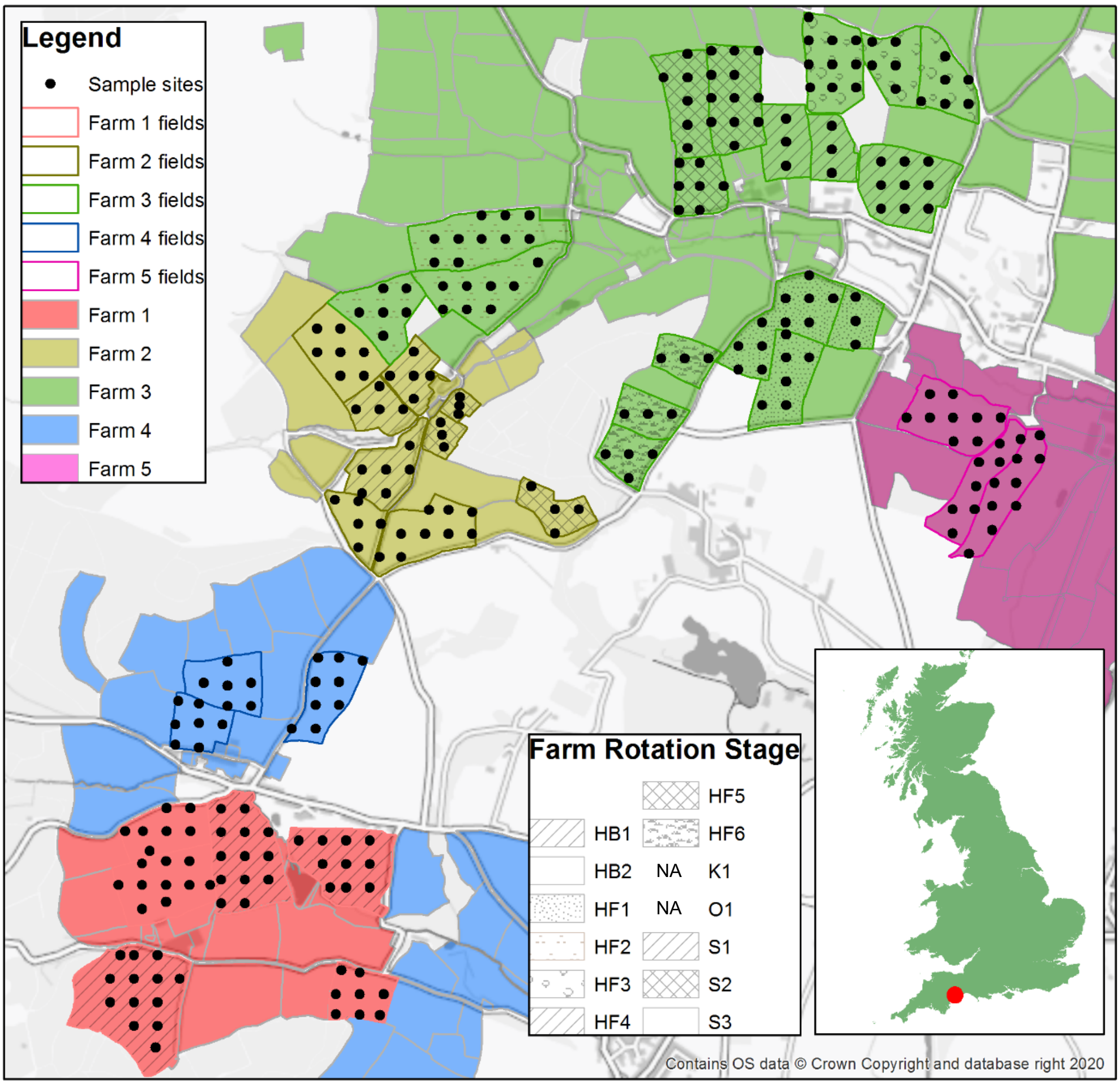


Figure 4.1: Map showing the study farms, field sites and sampling grid at its location in South West England. The five different study farms are highlighted in different colours, with the fields selected for sampling and sample points marked. Each stage of the rotation (specific to each farm) is demarcated by the key in the bottom right corner. Further explanation of each rotation stage can be found in Table 4.2

Farm management indicators were used to rank farm intensity in order to draw broad conclusions about the impact of intensification on soil NC condition. Indicators distinguishing different farm management practices were adapted from Büchi *et al.*, (2019). Büchi *et al.*, (2019) investigated a suite of agricultural indicators that could be used to “unveil the hidden side of cropping classification” (e.g. the similarities and differences between conventional, no-till and organic systems) and to draw conclusions about yield, environmental benefits and soil properties.

Other methods to develop a composite index of intensity have included additive methods, aggregating indicators after normalisation (Herzog *et al.*, 2006) or through use of multivariate analysis (e.g. PCA in Armengot *et al.,* (2011)). This study used a ranking approach as opposed to other methods for its simplicity and ease of comparing between the selected farms.

As recommended by Büchi *et al.*, (2019), a mixture of simple and complex indicators requiring differing levels of reference data or assumptions were applied. Table 4.1 shows farm management practices used to rank farming intensity from 1 (most intensive) to 5 (least intensive). Appendix B.1 provides further details on the assumptions made and reference data used.

Table 4.1: Showing the management variables used to rank farm intensity for each study farm including: Inputs (annual total N and P, inorganic N, stable organic matter (i.e. only the proportion of applied OM expected to be transformed into soil humus[5]) and sprays[6], cropping (mean number of crops grown (2013 – 2018), crop diversity (the number of different crops divided by total number of crops), mean number of years in grass (2013 – 2018)), tillage (mean number of primary tillage operations (2013 – 2018) and mean number of times mouldboard ploughed (2013 – 2018)), yield and the estimated crop cost of production (CoP). CoP is calculated including all costs before crop harvest/grazing. Ranking is scored from 1 (most intensive) to 5 (least intensive/most regenerative).

		Inputs (kg ha^{-1} unless specified)						**Cropping**			**Tillage**	**Yield**	**Cost of Production**		
Farm	**Type**	**Mean ann. tot N**	**Mean ann. inorg N**	**Mean ann. tot P**	**Mean ann. stable OM Input (IC)**	**Mean no. 2019 spray apps**	**Mean no. crops**	**Crop div.**	**Mean Yrs grass**	**Mean No. primary tillage ops**	**Mean times mb ploughed**	**Mean 2019 DM**	**Mean 2019 CoP**		
1	Arable	212.01	115.77	41.77	216.22	4.17	9.00	0.56	0.00	8.17	1.50	19.49	749.81		
2	Arable	144.41	141.27	33.43	592.80	1.33	7.00	0.59	1.11	6.44	5.11	15.29	490.17		
3	Arable	65.22	0.00	28.07	737.80	0.00	5.61	0.74	3.17	4.00	3.83	10.21	342.06		
4	Pasture	98.85	77.81	45.50	1999.33	0.00	1.33	1.00	5.67	1.67	1.33	8.87*	161.02		
5	Pasture	36.25	0.00	28.27	841.15	0.00	1.00	1.00	6.00	0.33	0.33	6.83*	64.77		
Ranking each farm based on the criteria:														**Rank Mean**	**Rank**
1		1	2	2	1	1	1	1	1	1	3	**1**	1	**1.33**	**1**
2		2	1	3	2	2	2	2	2	2	1	**2**	3	**1.92**	**2**
3		4	4.5	5	3	4	3	3	3	3	2	**3**	2	**3.38**	**3**
4		3	3	1	5	4	4	4.5	4	4	4	**4**	4	**3.71**	**4**
5		5	4.5	4	4	4	5	4.5	5	5	5	**5**	5	**4.67**	**5**
Rank Order		D	D	D	A	D	D	A	A	D	D	D	D		

*Here number of times paddock grazed in 2019 was used as a proxy for yield (otherwise for arable farm type yield is measured as mean t DM ha^{-1})
Rank Order: A = Ascending and D = Descending

[5] Isohumic coefficient = percentage of the organic waste or residue organic matter which is able to form humus in soil (Van-camp *et al.*, 2004). Further details and reference data used to calcuate are in Appendix B.2.
[6] Spray applications were the number of separate times that fungicide and/or herbicide were applied for the crop sampled in 2019.

4.2.2.2 Selecting metrics to assess soil natural capital condition

There is a wide literature on how best to assess the quality of agricultural soils, with many authors suggesting different soil quality indices (SQI) (Askari and Holden, 2015; Obade and Lal, 2016) or different indicators (Barrios, 2007; Brazier et al., 2011). Determining which indicators to use to assess soil condition is important in assessing the functionality of the soil, the ES services that flow from it and its NC value (Greiner *et al.*, 2017; Williams *et al.*, 2017). Despite this body of literature, there is still not an established consensus on how best to monitor soil condition in the UK (Humphries and Brazier, 2018). The UK Government have highlighted this in their recent 25 Year Environment Plan, by setting out to establish an agreed "soil health index" (HM Government, 2018). Greiner et al., (2017) in their review of the literature (n = 181), identify the most frequent soil properties used as indicators to assess soil function and infer ES delivery as: SOC, available water capacity, clay and silt contents (texture), soil type, depth and BD. At a national scale, the Countryside Survey for Great Britain conducted since 1978 as an 'audit' of the natural resources of the UK, monitors soil quality changes using BD, carbon, pH, total nitrogen (TN) and mineralisable-N, Olsen-P, metals and soil invertebrates (Emmett *et al.*, 2010).

The soil metrics used in this study were selected to, where possible, align with those used in the national monitoring scheme (Emmett *et al.*, 2010), while being relatively affordable and easily replicated by the estate. The metrics selected were used as indicators of soil NC condition when considering the suite of ES that soils can deliver. BD and n-potential (see paragraph below for definition) were selected to assess soil structural condition, linking to the soils capacity to regulate the water cycle and to support crop growth. SOM, pH, bioavailable-phosphorus (Olsen-P), Mg and K were selected as important in the provision of plant nutrients influencing crop growth. SOC, soil carbon stocks (t ha^{-1}) and n-potential (requiring textural analysis of the soil for sand, silt and clay content) were selected to assess soil carbon storage, linked to climate regulation. Further details on the justification of metrics and the links to soil-based ES can be found in Chapter 3, Sections 3.4 and 3.5.

Explained in Merante *et al.,* (2017), n-potential is the ratio of clay (%) to SOC (%). N-potential indicates the "potential" presence of non-complexed clay and as such, enables the assessment of soil stability (i.e. the soils vulnerability to soil erosion) and its capacity to store more carbon. A high n-potential (>10) suggests lower soil stability and greater capacity to store more carbon. A low n-potential (<10) suggests higher soil stability and lower capacity to store more carbon (Merante *et al.*, 2017). The relevance of clay:SOC ratio as a metric of soil condition is explained in more detail in Chapter 3, Section 3.5. Prout *et al.*, (2020) also define bounds for the clay:SOC ratio with reference to soil degradation in the UK: a ratio less than 8 refers to very good soil condition, 8 - 10 refer to good soil condition, 10 – 13 as moderate and greater than 13, as degraded soil condition. These boundaries are used in the interpretation of the n-potential data herein.

4.2.3 Field and sampling site selection

Study fields were selected from each of the five study farms across each stage of the rotation, within the constraints of ongoing management (details in Table 4.2). It was recognised that the stage of the rotation was likely to influence soil condition. All farm fields were characterised based on soil association, slope and aspect. All of which were considered to influence soil condition and crop growth. Grouping analysis using ArcGIS 10.5.1 (ESRI, Redlands, CA, USA) was then conducted to identify comparable fields across land-use intensities and rotational stage that, wherever possible, shared a similar slope and aspect and were on the same soil association. Three replicate fields per phase of the rotation for each farm were selected for soil sampling and further study (Table 4.2).

Table 4.2: Details of the number of fields sampled and the stages of rotation sampled at Clinton Devon Estate, with reference codes in brackets for each stage of rotation (relates to Figure 4.1). The typical crops grown and the farm system is also shown. Acronyms used: CON (conventional farming) and ORG (organic farming).

Farm	No. fields sampled	Stages of rotation soil sampled	Typical Crops	System
Farm 1 (cropped)	6	Rye (following maize) (HB1) Grass (into maize) (HB2)	Rye, maize, rye-grass	CON arable
Farm 2 (cropped)	9	Once grass (S1)* In grass (S2) No grass (S3)**	Rye-grass and clover, stubble turnips, winter wheat and barley, maize	CON arable
Farm 3 (cropped)	18	Year 1 arable (HF1) Year 2 arable(HF2) Year 3 arable (HF3) Year 1 grass (HF4) Year 2 grass (HF5) Year 3 grass (HF6)	Spring triticale and wheat, rye-grass and white clover, stubble turnips, cover crops	ORG arable
Farm 4 (grazing)	3	Pasture (K1)	Rye-grass and white clover	CON diary
Farm 5 (grazing)	3	Pasture (O1)	Organic rye-grass and white clover	ORG diary
Total:	39			

***Field has been in grass previously in the last six years**
**** Field has not been in grass in the last six years**

4.2.4 Soil sampling and analysis

Between November 2018 and January 2019, 268 soil samples were collected, each sample was collected following the completion of the majority of the cropping, final silage cut and main grazing season. A systematic sampling design with an 84m sampling grid (Figure 4.1) mapped in ArcGIS 10.5.1 (ESRI, Redlands, CA, USA) was selected following the findings of Peukert *et al.*, (2012). The study identified an 84m sampling density as adequate for capturing within field spatial variability of BD and associated nutrient stocks (carbon and nitrogen) in similar agricultural topsoil. Sampling sites were identified in the field using a hand-held GPS (Nomad Trimble, Sunnyvale, CA, USA) with an accuracy of 1 – 2m.

Soil samples were collected and analysed for BD, SOM, SOC, bioavailable-P, pH and soil texture (clay:silt:sand) and secondary indicators carbon stocks (t SOC ha^{-1}) and N-potential (Clay:SOC ratio) were then calculated. Detailed methods can be found in Chapter 3, Section 3.6.1.

4.2.5 Measuring productive output

4.2.5.1 Biomass yield

All biomass yield data were collected in 2019, immediately prior to harvest (or as close as possible). The same sampling sites (as for soils) were located in the field using a hand-held GPS (Nomad Trimble, Sunnyvale, CA, USA) and 0.5m x 0.5m cut plots taken for silage, hay and cereals. As maize is a row crop, a different method of yield estimation was conducted, following (Steinhilber, Shipley and Vvedenskaya, 2016). A fresh and dry weight was determined for all biomass samples before site measurements were scaled up to calculate the dry matter (DM) biomass yield in tonnes per hectare (t ha^{-1}).

Further detailed methods of the collection of biomass samples can be found in Chapter 3, Section 3.6.4.

4.2.5.2 Grazing frequency

The collection of biomass yield ahead of grazing was not feasible at grass paddock sites[7]. Instead, the number of times the field was grazed throughout 2019 was used to infer productivity. As the farms graze at similar covers (the amount of grass available) of between 2500 – 3000 kg/ha and leave similar residuals 1500 kg/ha, the frequency of grazing was considered a reasonable measure of performance. The same approach is used by one of the study farms to infer paddock performance. The simple assumption was made, as discussed with the farmer, that if the field was grazed more during the season, it was producing higher quantities of grass.

Unfortunately, the lack of biomass yield data at each sample point meant that pasture soil sample sites had to be omitted from analysis of the relationship between yield and soil condition.

4.2.6 Collecting farm management data

Field management data were collected from each farmer over the 6 years prior to soil sampling (2013 – 2018). Records relating to the establishment and harvest of the 2019 crop were also collected. Six years was chosen as it spans the full

[7] Paddocks are typically grazed between milking (for ca. 12 – 24 hours) and then rested for 20 – 30 days before the paddock is grazed again once the grass growth reaches the desired cover.

length of the longest farm rotation (i.e. the organic rotation for Farm 3). It was also possible to collect fairly reliable management data from the farmers for this period. Where specific records were not available, for example, regarding organic inputs or exact cultivations, assumptions had to be made based on knowledge of the cropping at the time and the current protocol for that crop. These assumptions were first discussed and agreed with the farmer before being finalised in the data set. The management records collected are shown in Appendix B.1 and follow suggestions from Büchi *et al.*, (2019), regarding management indicators that are likely to influence yield and soil condition.

Farm management data for the establishment of the 2019 crop was used to estimate a crop Cost of Production (CoP, £ ha^{-1}) before harvest. This helped rank farm intensity (as a composite measure of mechanical and chemical inputs pre-harvest) and to see if management effort had a more significant impact on soil NC condition than soil properties. Costs for each management activity were either provided by the farm or they were estimated based on local contractor costs or data in the Nix Farm Pocket Handbook (2018). All fertilizer and spray costs were estimated using Nix (2018). Further details can be found in Chapter 3, Section 3.6.5.

4.2.7 Character maps for soil condition and productive output

Soil and yield maps were created to visualise data across fields, field rotations and farm sites using interpolation following the Inverse Distance Weighting (IDW) method. IDW predicts values for unmeasured locations using the measured values surrounding that location. Measured values that are closer to the predicted value have greater influence on the predicted value than those measured values further away. All spatial analysis was undertaken in ArcGIS 10.5.1 (ESRI, Redlands, CA, USA).

4.2.8 Statistical analysis and model terms

4.2.8.1 The impact of farm intensity on soil properties and productive output

To compare the impact of farm intensity on soil NC conditions (BD, SOC, bioavailable-P and n-potential) and biomass yield, the data were first assessed for normality. Linear mixed-effects models with random terms field and farm rotation stage, were used to determine if intensity had a significant effect on BD and SOC. Generalized linear mixed effect models (family = Gamma) with random terms field and farm rotation stage, were used for the analysis of Olsen-P and n-

Potential data. Mixed model analysis was conducted using the lme4 package (Bates *et al.,* 2015) in R (R Core Team, 2020). *Post-hoc* analysis for pairwise comparisons between each intensity rank were conducted using the emmeans package (Lenth, 2020). Significance was tested at $p<0.05$. All analyses were conducted in R (R Core Team, 2020).

4.2.7.2 Impact of different agricultural management practices on soil properties

Data exploration was conducted using a multi-model inference approach (Burnham and Anderson, 2002) applying mixed effects models. Mixed effects models were considered necessary to account for variance caused by field and within farm rotation stage (referred herein as field group). Specific management terms relating to soil condition outcomes were defined as predictor variables. Field and field group were used as random effects. Farm system (organic or conventional) was always included as a fixed effect to account for systematic differences between systems. All possible combinations of fixed effects were ranked using their Akaike Information Criterion (AIC) score (Akaike, 1973). AIC score is a commonly used information criteria (IC) in comparing mixed models and selecting the 'best' or most parsimonious model. IC are based on the likelihood of the data giving a fitted model whilst prioritising model simplicity (Nakagawa and Schielzeth, 2013). Model averaging was applied across all models to determine effect size of each management variable and whether it had a positive or negative impact on soil condition. Analyses were conducted using the lme4 package (Bates *et al.,* 2015) to compute linear mixed effects models and the glmulti package (Calcagno, 2020) for automated model selection, model ranking and averaging of importance terms. Summaries of the best model (lowest AIC score) were conducted using the jtools package (Long, 2020), which provides details of the pseudo R^2 (using the Nakagawa and Schielzeth (2013) method). The Pseudo-R^2 method allows the extension of the useful summary statistic, explaining variance in linear and generalized linear models (R^2), to linear and generalized linear mixed effects models (Nakagawa and Schielzeth, 2013). N-potential and Olsen-P data were log transformed to improve the normality of the data ahead of analysis.

4.2.7.2.1 Selecting management terms

The management variables selected as predictors for each of the soil NC metrics are shown in Table 4.3.

Multi-collinearity was checked prior to selection of terms for use in multi-model inference. A global linear mixed effects model was run using all selected management variables and a multicollinearity test conducted. Where significant issues of multicollinearity arose (VIF > 10), the management variables considered least likely to explain the soil condition variable were removed. *A-priori* justification of management variables and issues with and solutions necessary to address multicollinearity are explained further in Appendix B.3. Both *a-priori* justification of predictor variables and assessing multicollinearity have been identified as important phases in conducting multi-model inference analysis (Grueber *et al.*, 2011).

Table 4.3: Management variables selected as predictors of each soil property in the multi-model inference process. The code for the term is included in brackets after the description.

Management variables used in multi-model inference:			
Bulk density	**Carbon stocks**	**N-potential**	**Bioavailable-P**
Time since tillage (Years_since_tillage)	Time since tillage (Years_since_tillage)	Time since tillage (Years_since_tillage)	Mean annual inorganic P inputs (Mean_ann_inorgP)
Estimated stable organic carbon inputs (IC_input_tha)	Estimated stable organic carbon inputs (IC_input_tha)	Estimated stable organic carbon inputs (IC_input_tha)	Mean annual organic P inputs (Mean_ann_orgP)
Number of field traffic passes in the year prior to sampling (Est_passes_2018)	Crop diversity from 2013 - 2018 (Num_diff_crops)	Crop diversity from 2013 - 2018 (Num_diff_crops)	Number of primary field cultivations (2013-2018) (Num_tillage)
Number of primary field cultivations (2013-2018) (Num_tillage)	Number of primary field cultivations (2013-2018) (Num_tillage)	Number of primary field cultivations (2013-2018) (Num_tillage)	Mean annual organic matter input (from FYM, slurry and est. from grazing animals) (OM_tha)
Number of times field mouldboard ploughed (2013 – 2018) (Num_mb_plough)	Number of cover crops (2013 – 2018) (Num_yrs_CC)	Number of cover crops (2013 – 2018) (Num_yrs_CC)	Soil pH (to account for management adjustments to soil pH – liming etc) (pH_mgl)
Crop type in field during sampling (Crop_type_sampled)	Number of years the field has been grazed (2013 – 2018) (Num_grazing_yrs)	Number of years the field has been grazed (2013 – 2018) (Num_grazing_yrs)	Number of years the field has been grazed (2013 – 2018) (Num_grazing_yrs)
System	System	System	System

4.2.7.3 Impact of soil natural condition on productive output

Linear mixed-effects models were used to determine whether soil NC condition had a significant impact on biomass yield. Two models were run: the first with only soil properties as explanatory variables (BD, SOC, pH, P, K and n-potential) and the second, which included the same soil properties along with a management term (crop cost of production). Crop cost of production was used as a measure of the relative mechanical and chemical inputs applied to the growing process. It was included to determine the importance of management inputs relative to soil NC condition. Field, field group and crop type were used as random effects in both models. For presentation purposes the same multi-model inference approach was conducted as for the drivers of soil condition using lme4 package (Bates *et al.,* 2015) to compute linear mixed effects models and the glmulti package (Calcagno, 2020) for automated model selection, model ranking and averaging of importance terms. All analyses were conducted in R (R Core Team, 2020).

4.3 Results

4.3.1 Characterising soil properties and productive output across the five study farms

The spatial interpolation of soil NC properties is shown in Figure 4.2. The maps identify the spatial variability of soil properties and biomass yield. A summary of the data is presented in Table 4.4 for each study farm. Data are compared alongside existing soil data including the National Soils Resources Institute (NRSI, 2005) data for the same Bromsgrove soil association, other national soils datasets for similar land-use (Emmett *et al.*, 2010) and other agricultural studies on similar soil types (Table 4.5).

It is evident that soil properties are not spatially uniform within fields, within rotation stage or within farms. Increasing variability is seen in biomass yield, BD, SOC, n-potential and Olsen-P data when assessing relative standard deviation (RSD) from the field to field group to farm scale. The mean RSD across all soil properties and biomass yield was 17.74% at the field scale, 22.08% at the field group scale and 25.02% at the farm scale.

Comparing data from this study with NSRI (2005) data shows that SOC (%), estimated carbon stocks (t ha^{-1}) and BD (g cm^{-3}) are all consistently different between the two datasets (Table 4.5). Data collected in this study do not reflect the mean conditions for the Bromsgrove soil association, as collected as part of the National Soils Inventory (NSRI, 2005), with SOC and carbon stocks consistently lower and BD consistently higher across the land-use types. SOC data from this study is also consistently lower than the national mean collected through the Countryside Survey England (Emmett *et al.*, 2010) and from most other studies on sandy loam soils. Data from Loveland and Webb (2003) do show a similar pattern in mean SOC (%) compared to the data collected at the case-study sites across the three land-uses, with lowest SOC under arable cropping (SOC 1.75%), medium SOC under ley (2.29%) and highest SOC under pasture (2.59%). The use of existing data (for similar soil types) does facilitate benchmarking of soil NC conditions against other farms/national averages. However, these comparisons highlight that the use of existing soil datasets do not adequately reflect ground conditions or account for the spatial variability seen in Figure 4.2.

The obvious patterns that emerge from the data are highlighted below. Mean values are shown with ± standard deviation.

The highest consistent SOC levels (2.26% ± 0.52%), lowest n-potential (7.43 clay:SOC ±2.62), lowest BD (1.18 g cm^{-3}±0.12) and lowest bioavailable P (25.58 mg l^{-1}±9.61) are seen across the pasture fields sampled at Farm 5.

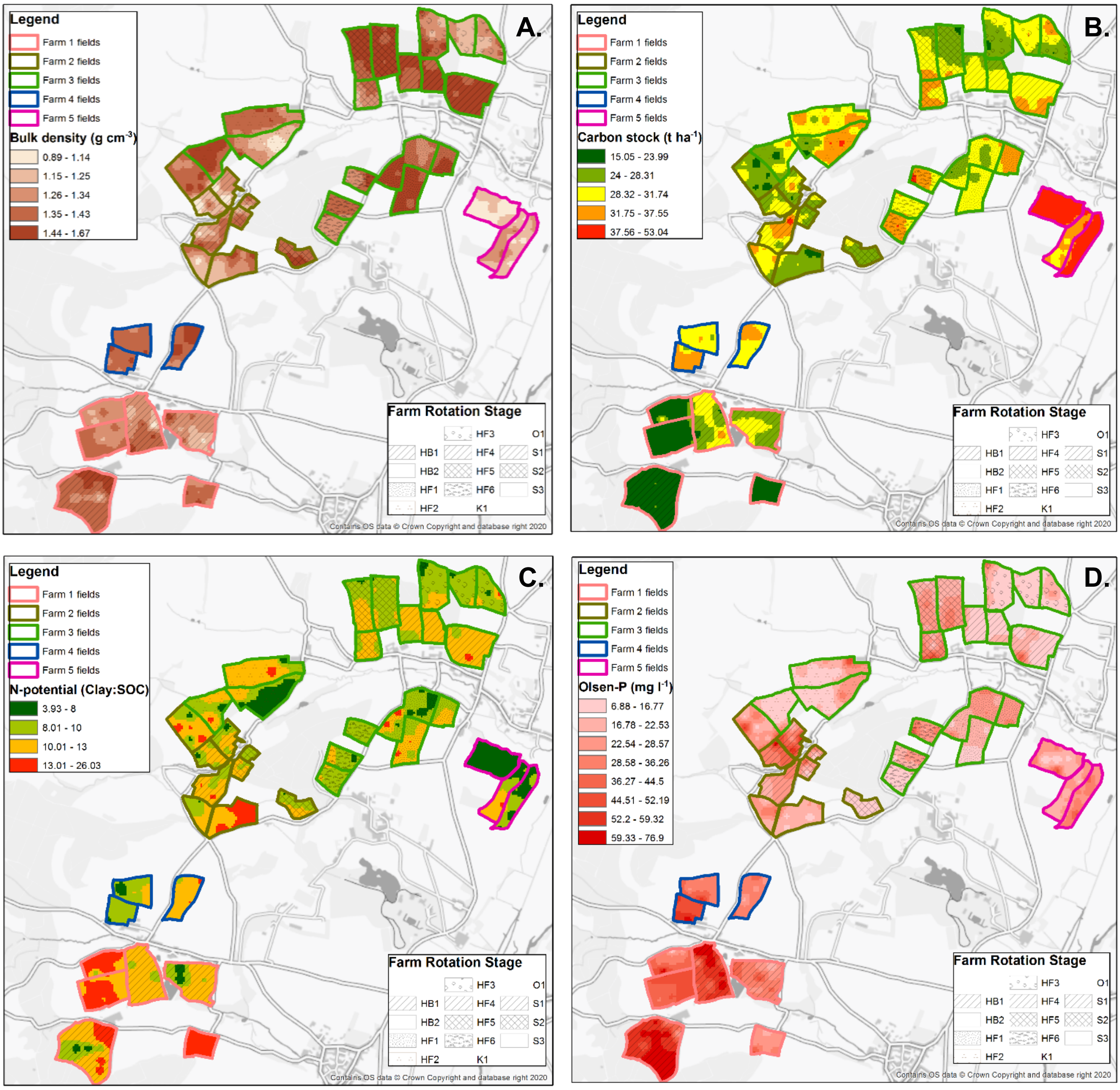
Farm 1 exhibits uniformly higher bioavailable P (43.65 mg l^{-1}±16.04), particularly in three fields that are part of the HB1 phase of the rotation (fast grass after cereal) (48.58 mg l^{-1}). Lower SOC (1.19% ± 0.27) and higher n-potential (12.78 clay:SOC ± 4.04) are seen across the farm but are lowest in the HB2 phase (winter cereal after maize) of the rotation, 1.03% SOC and 15.33 n-Potential. BD is fairly moderate (1.34 g cm^{-3} ± 0.11) with few compacted areas.

Across the Farm 3 rotation there appears to be moderate spatial variability within most stages of the rotation, with the exception for the HF4 stage (1st year grass). The HF4 stage is fairly homogenous (lower SD) for SOC (1.41% ± 0.08) and BD (1.45 g cm^{-3} ± 0.06) compared to all other field groups. The highest mean SOC is seen in the HF6 (3rd year grass) of the rotation (1.66%). SOC hotspots are also seen across HF2 (2nd year arable - max 2.9%), HF3 (3rd year arable - max 1.83%) and HF5 (2nd year grass - max 1.96%) stages. BD appears to be uniformly higher across fields in the early grass stages of the rotation, HF4 (1st year grass - 1.45 g cm^{-3}, max 1.54 g cm^{-3}) and HF5 (2nd year grass - 1.44 g cm^{-3}, max 1.59 g cm^{-3}). High BD is also seen in the HF1 (1st year arable - 1.43 g cm^{-3}, max 1.61 g cm^{-3}) stage. N-potential and Olsen-P are both relatively low across the whole of Farm 3: 9.76 clay:SOC and 19.76 mg l^{-1} respectively. The lowest consistent Olsen-P values across sites are seen in the HF4 stage (1st year grass) of the rotation, 15.77 mg l^{-1}.

Farm 2 BD data appear to be uniformly moderate (1.31 g cm^{-1} ± 0.16) with increases seen in parts of the grass phase of the rotation (1.37 g cm^{-3}). N-potential and SOC seems to be variable between fields (1.21% - 1.69%, 9.19 – 13.58 clay:SOC) and within rotation stage. Although SOC appears to be consistently higher in all fields in the S1 (once in grass) stage of the rotation

(1.63%±0.24). Olsen-P is variable but is relatively low across most fields (25.89 mg l^{-1} ± 13.56).

The pasture at Farm 4 exhibits low variability in BD (1.4 g cm^{-3} ± 0.09), SOC (1.52% ± 0.16) and n-Potential (9.95±1.73). Olsen-P is higher across all sites apart from Farm 1 and more variable, 39.48 mg l^{-1} ± 13.09.



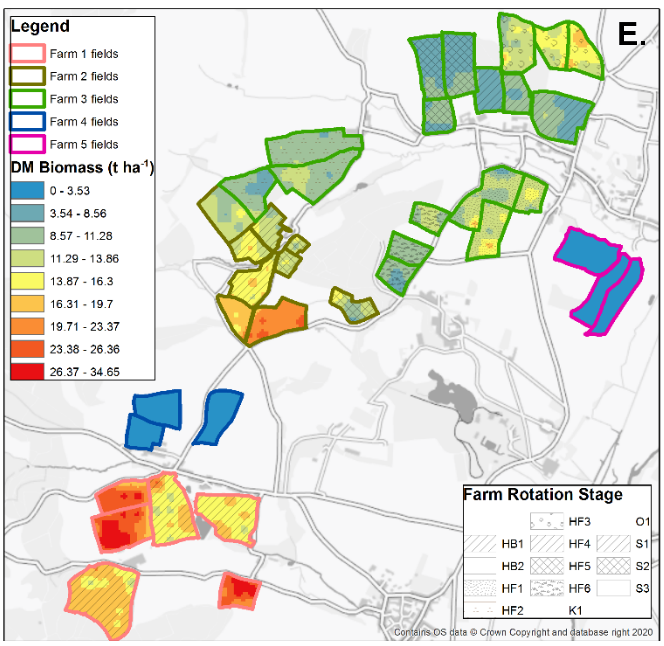


Figure 4.2: Showing spatially interpolated soil NC properties by Inverse Distance Weighting method across the five study farms. A. Shows bulk density (g cm^{-3}), B. Soil organic carbon (%), C. n-potential ratio (clay:SOC), D. Bioavailable P (Olsen-P mg l^{-1}) and E. DM biomass yield (t ha^{-1}).

Table 4.4: Summary statistics (mean and standard deviation) for productive output (biomass yield or frequency of grazing in 2019), BD, SOC, carbon stocks, n-potential, Olsen-P and soil texture (Sa = sand, Si = silt and Cl = clay) for each of the study farms. Yield types are distinguished as DM tonnes ha^{-1} shown as Y and number of times field grazed as g.

Farm	**Yield 2019 (all crops)**		**Bulk Density**		**SOC**		**Carbon stock**		**nPotential**		**Olsen-P**		**pH**		**Soil Texture**	**n**
	Mean	SD	Mean (g cm^{-3})	SD	Mean (%)	SD	Mean (t ha^{-1})	SD	Mean (Clay:SOC)	SD	Mean (mg l^{-1})	SD	Mean	SD	Sa:Si:Cl	
Farm 1	19.49 ^{y}	5.50	1.34	0.11	1.19	0.27	23.68	4.57	12.78	4.04	43.65	16.04	6.43	0.34	65:21:14	59
Farm 2	15.29 ^{y}	4.71	1.31	0.16	1.44	0.27	27.85	3.94	10.88	2.43	25.89	13.56	6.17	0.42	68:18:14	46
Farm 3	10.19 ^{y}	3.25	1.39	0.12	1.45	0.28	29.77	3.84	9.76	2.16	19.76	9.00	6.62	0.34	67:18:15	115
Farm 4	8.87 ^{g}	1.15	1.40	0.09	1.52	0.16	31.68	2.70	9.95	1.73	39.48	13.09	6.67	0.35	66:18:16	22
Farm 5	6.83 ^{g}	1.44	1.18	0.12	2.26	0.52	39.43	6.40	7.43	2.62	25.58	9.61	6.59	0.45	64:21:15	26

Table 4.5: Summary data on soil properties for National Soils Research Institute (2005). Records for the Bromsgrove soil association are compared with summary data collected in this study. Additional sources are also presented from the Countryside Survey 2007 (CEH, 2007) and other agricultural studies on sandy loam soils.

Data source	Soil texture	Core depth	n	BD (g cm^{-3})		SOC (%)		Carbon Density (t ha^{-1})		N-potential	
				Mean	Range	Mean	Range	Mean	Range	Mean	Range
NSRI soil series data: Bromsgrove soil series											
NSRI (2005): Bromsgrove soil series, Arable	SL	0 - 25	*NS*	1.34	NS	2	NS	40.2 (est.)		7	NS
NSRI (2005):Bromsgrove soil series, Ley grass	SL	0 - 25	*NS*	1.21	NS	2	NS	36.3 (est.)		7	NS
NSRI (2005): Bromsgrove soil series, Pasture	SL	0 - 25	*NS*	1.19	NS	2.20	NS	39.27 (est.)		6.36	NS
Study data:											
Clinton Devon Estate: Arable and horticultural	SL	0 - 15	*140*	1.35	0.88-1.68	1.34	0.67-2.90	26.73	14.91-40.00	11.49	4.83-26.39
Clinton Devon Estate: Ley grass	SL	0 - 15	*80*	1.37	0.93-1.59	1.44	0.94-2.09	29.88	20.41- 39.16	9.74	6.9-14.39
Clinton Devon Estate: Pasture	SL	0 - 15	*48*	1.28	0.9-1.67	1.92	1.24-3.26	35.88	25.89-53.38	8.59	3.88-14.39
Other data sources for sandy loam:											
Loveland and Webb (2003)*: Arable	S	0 - 25	75			1.75	0.4 - 9.1				
Loveland and Webb (2003)*: Ley grass	S	0 - 25	7			2.29	0.7 - 4.5				
Loveland and Webb (2003)*: Permanent grass	S	0 - 25	28			2.59	0.6 - 5.4				
Johnston *et al.*, (2017): All treatments 2000 - 2009 (ley - arable)	SL	0 - 25	NS			1.25		43.95	NS		
Countryside Survey England mean data:											
Emmett *et al.*, (2010): Arable and Horticultural land	All	0 - 15	*NS*	1.23	NS	3		46.9	NS		
Emmett *et al.*, (2010): Improved grassland	All	0 - 15	*NS*	0.97	NS	5.31		64.6	NS		

* Data derived from National Soils Inventory: England and Wales - Est.: Estimated data based on assuming a 15cm soil depth for comparison purposes. NS – Not supplied, where only mean data was presented. Soil textures: SL = Sandy loam, S = Sandy

4.3.2 The impact of farm intensity on soil natural capital condition and productive output

Farming intensity had a significant impact on soil NC condition and biomass yield. Figure 4.3 shows soil NC properties and crop biomass yield (only applicable for Farm 1 – 3) for each of the farming intensities. Clear patterns are visible for carbon stocks (Figure 4.3B), n-potential (Figure 4.3C) and biomass yield (Figure 4.3E). Mixed effects models show that increasing intensity has a significant negative effect on soil carbon stocks (p <0.001), a significant positive effect on n-potential (p = 0.007) and a significant positive effect on biomass yield (p = 0.003). The data suggest that increases in productive output are associated with a trade-off in soil carbon and soil stability (n-potential).

Pairwise comparison (further details in Appendix B.4) shows significant differences in carbon stocks between farms, with Farm 5 (Intensity 5) storing significantly more carbon than Farm 1 (p = 0.001), Farm 2 (p = 0.008) and Farm 3 (p = 0.018) farms. N-potential has a clear declining trend with decreasing intensity in Figure 4.3C but the only close-to-significant difference between farms is for the highest (Intensity 1) and lowest (Intensity 5) values (p = 0.050). The only pairwise significant differences in biomass yield are between the Farm 3 and those from Farm 1 (p = 0.01). Statistical analysis was not conducted on the small number of data observations for number of grazing occurrences but data in Table 4.4 shows that fields at Farm 5 (Intensity 5) are typically grazed less frequently (6.83 times in 2019) than those at Farm 4 (Intensity 4) (8.87 times in 2019).

Figure 4.3A shows that BD is fairly uniform for farm intensities 1 – 4 but is considerably lower on the pasture at Farm 5 (model estimate = - 0.15 g cm^{-3}). However, whilst this produced an upward trend, this was not recognised as significant (p = 0.07). Olsen-P is evidently variable between farms but the only significant difference is between the farms with the highest and lowest Olsen-P values, Farm 1 and Farm 3, respectively (p = 0.04).

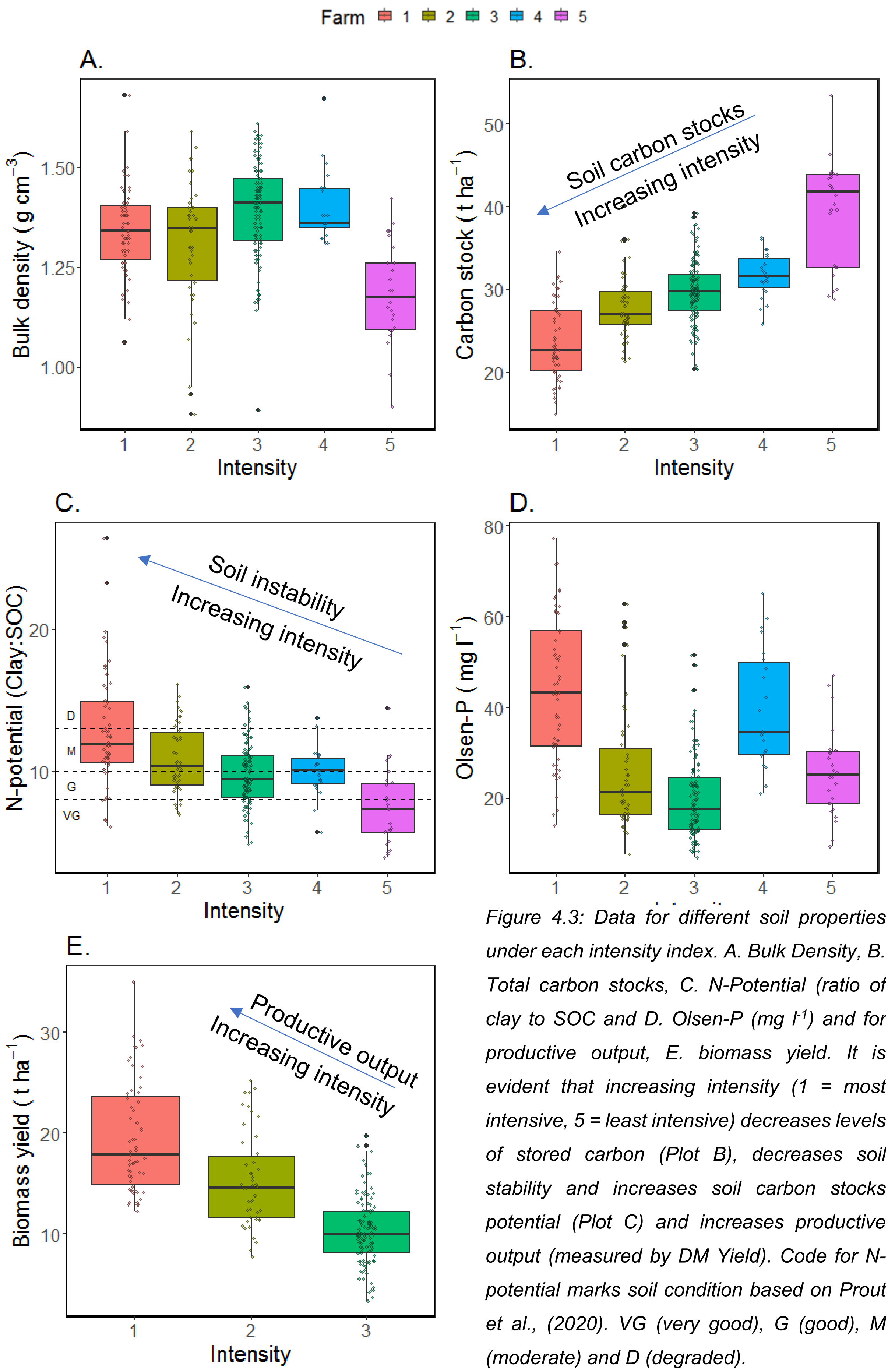


Figure 4.3: Data for different soil properties under each intensity index. A. Bulk Density, B. Total carbon stocks, C. N-Potential (ratio of clay to SOC and D. Olsen-P (mg l^{-1}) and for productive output, E. biomass yield. It is evident that increasing intensity (1 = most intensive, 5 = least intensive) decreases levels of stored carbon (Plot B), decreases soil stability and increases soil carbon stocks potential (Plot C) and increases productive output (measured by DM Yield). Code for N-potential marks soil condition based on Prout et al., (2020). VG (very good), G (good), M (moderate) and D (degraded).

4.3.3 Drivers of soil natural capital condition

The scale and significance of field management practices on soil structure (BD), soil carbon stocks, soil stability (n-potential) and bioavailable-P are presented in Figure 4.4. All terms used in the model selection process are listed in order of importance (model averaged order of importance, using AIC method). Effects to the right of zero are considered to have a positive effect and effects to the left a negative effect on soil properties. Only model terms with confidence intervals (CI: 0.025 – 0.975) that do not span zero are considered to have a significant effect.

Figure 4.4A shows that no field management variables were identified as being significant predictors of BD, though time since tillage is identified as the highest model averaged importance term. Whilst not significant the model output suggests that increasing time since a field was last subject to primary cultivations reduces soil BD. Interestingly when investigated further and plotting time since tillage against BD, it is evident that it is not a linear relationship but there is a distinct bell-shaped pattern in the data. Reduced BD are seen shortly after tillage, rise to a maximum at around 40 – 60 weeks since tillage and then fall again to their lowest after a long period of no tillage. See plot in Appendix B.5 for a visual demonstration of this.

Figure 4.4B shows that the time since the field was last tilled and the number of primary tillage operations in the last 6 years are the best predictors of carbon stocks from the input management terms. Increasing time since the field was last tilled increased carbon stocks (CI: 0.17, 1.46), whereas increased tillage frequency decreased carbon stocks (CI: -2.31, -0.04). None of the remaining terms were significant (confidence intervals spanned zero). It is important to note that multicollinearity issues were identified in preliminary data assessment between: the number of tillage operations, the number of crops, the number of years the field was in grass and the mean annual total input of N (see Appendix B.3 for more details). The relationships between carbon stocks and each term are shown in Appendix B.6.

Figure 4.4C shows that the time since the field was last tilled had a significant negative effect on n-potential (CI: -0.07, -0.003). The number of different crops and the number of times the field was tilled in the last 6 years were also identified as important model terms but were not significant in the model selection process. The data suggests that fields that have not been tilled for longer periods (in this case, usually fields in grass) have a more stable soil structure (higher proportion of clay complexed with carbon). In contrast, fields that have experienced recent tillage exhibit lower soil stability (less clay complexed with carbon) and higher potential to store more carbon.

Organic management, soil pH and mean annual inorganic P inputs were considered the top three importance terms across all models in explaining bioavailable P concentrations (Figure 4.4D). Only farm system was identified as significant (CI: -1.10, -0.42), with organic fields having significantly lower bioavailable P than conventional fields. Ph (CI: -0.01, 0.34) and mean annual inorganic P inputs (CI: -0.05, 0.01) are close to having a significant effect and both are in the highest ranked model.

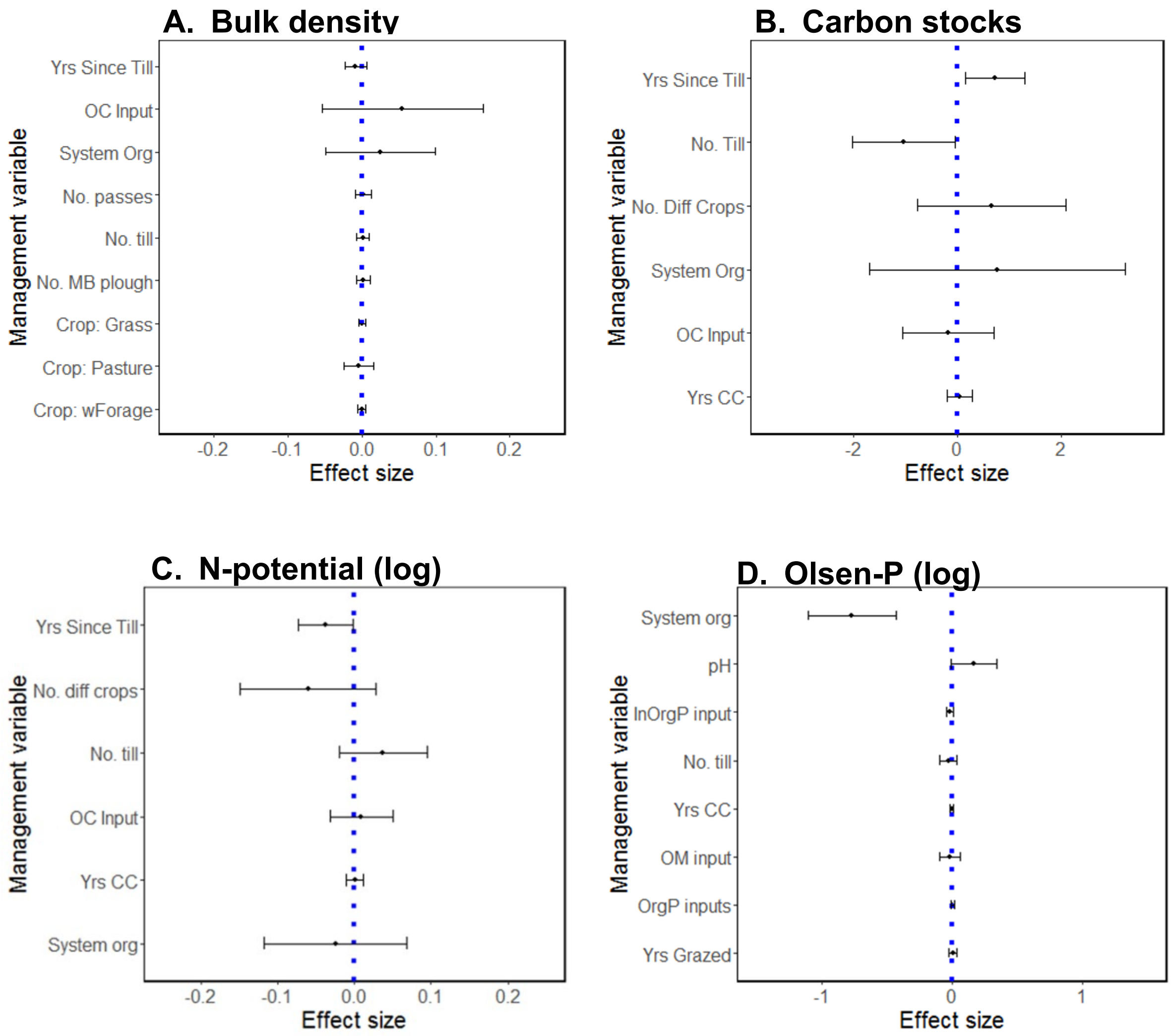


Figure 4.4: Effect size of different management practices/records on A.) Bulk density, B.) Carbon stocks (t ha-1), C.) N-potential and D.) Olsen-P. Confidence intervals (0.0250 – 0.975) are shown. Terms that do not span zero are taken as having a significant effect. No management terms have an effect on BD. Years since tillage and number of times field was tilled over rotation have a significant effect on carbon stocks. Years since tillage has a significant effect on N-potential and system having a significant effect on olsen-P (although pH is also close to significant). Key to terms: Yrs since tillage (time since tillage), OC input (stable organic matter inputs), system (con taken as intercept), No. passes (number of passes from machinery in 2018), No. tillage (number of times primary tilled from 2013 – 2018), No. MB plough (number of times mouldboard ploughed from 2013 – 2018), InOrgP input (mean annual inorganic inputs of P), Yrs CC (number of years cover cropped 2013 – 2018), OM input (estimated stable organic carbon inputs as organic matter), OrgP (organic P inputs as FYM) and Yrs Grazed (number of years field grazed from 2013 – 2018), No. Diff Crops (number of different crop types 2013 – 2018) and System (Con used as intercept).

4.3.4 The impact of soil natural capital condition on productive output

Mixed linear-effects model results using all measurements of soil NC condition showed no significant predictor terms when controlling for field group and crop type. The model showed a pseudo-R^2 of 0.02 for fixed effects and a pseudo-R^2 of 0.86 including random effects, suggesting that the random terms explain more of the variance than soil conditions. The data suggests that changes in soil conditions do not have a systematic impact on crop yield across the study sites.

The inclusion of a management term (crop 'Cost of Production' (CoP)) significantly improved the model, improving the pseudo-R^2 for fixed effects to 0.50, with the pseudo-R^2 of random effects remaining the same. All model terms were used in the model selection process to identify the relative importance of each predictor variable in explaining variance in crop yield. The results are presented in Figure 4.5. Increasing SOC (%) showed a positive but insignificant effect on biomass production with increasing pH and BD showing a negative but also insignificant effect on biomass production. The only significant predictor term was cost of production (CI: 1.22, 2.86). The data suggests that management effort is more likely to drive crop yields than soil condition at the case-study sites. Further detailed analysis, split into field groups receiving the same management, is presented in Appendix B.7.

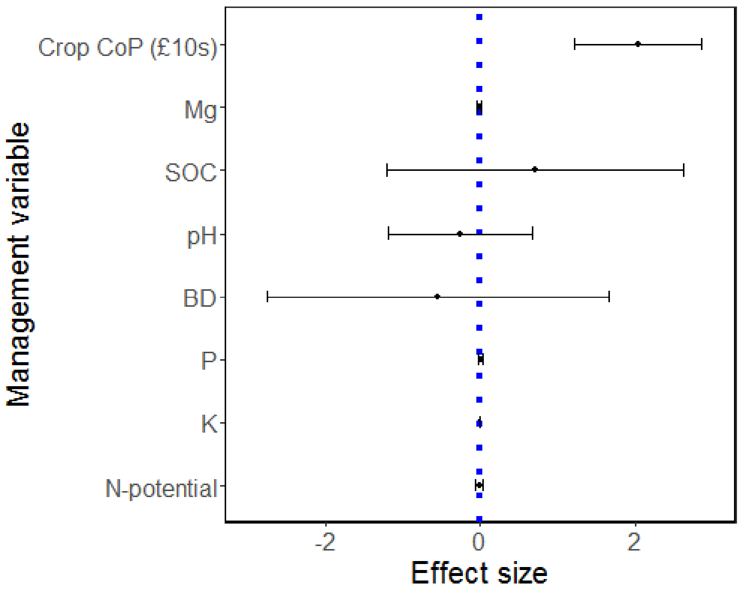


Figure 4.5: Effect size of soil properties and cost of production (CoP) on biomass yield. Confidence intervals (0.0250 – 0.975) are shown. Terms that do not span zero are taken as having a significant effect on biomass yield, which in this case only applied to the crop CoP.

4.4 Discussion

The results from this study show that following the categorisation of baseline soil NC conditions it is possible to apply farm management data to better understand drivers of soil condition. The data suggests that intensification across the study farms can have a significant impact on soil NC condition, driving decreases in soil carbon stocks and soil stability (n-potential). Based on management data, it was possible to discern that the frequency of tillage and the time since the field was last tilled were potential drivers of carbon losses and reduced soil stability. However, despite degraded soil carbon and soil stability, there was no evidence to suggest that this systematically reduced biomass yields across the study sites. The discussion around these key findings is expanded on below after first highlighting the importance of categorising soil NC condition at the farm scale.

4.4.1 The importance of categorising soil NC at the farm scale

The results from this study show that existing national datasets (NSRI, 2005) do not provide an adequate resolution for evaluating changes in soil NC condition at the field or farm scale. Use of the national dataset would have led to an overestimation of soil carbon stocks (t ha^{-1}) across four of the study farms (intensities 1 – 4) and the underestimation of soil carbon stocks at the lowest intensity farm (under long-term pasture). Glendell *et al.,* (2014) also found that the national dataset (NSRI, 2005) underrepresented the variability of BD and SOC in clay and loam soils. They advocate the need for higher spatial resolution mapping for practical land and water management purposes (Glendell *et al.*, 2014) .

As shown in this study whilst the comparison with other data can facilitate benchmarking of soil NC condition with other agricultural sites it does not provide the detail required in making or evaluating local land management decisions. It highlights the need for the collection of primary soil condition data in attempts to undertake the initial tier of the NC approach (an audit of NC quantity and quality). The collection of this detailed data supports the assessment of what might be driving soil condition and can inform future decisions at the farm scale.

4.4.2 Addressing research question 1: How does farm intensity impact soil natural capital and productive output?

By applying high-resolution soil NC data alongside a classification of soil management intensity it was possible to detect significant differences in soil condition between different farms. Notably increasing farm intensity resulted in degraded soil carbon and decreased soil stability (n-potential). Intensity was classified on the level of inputs, frequency of cultivations, cultivation type, crop diversity and length of grass ley included in the rotation. The impact of agricultural intensification of soils is fairly well documented and the finding here supports others who have found that intensive arable cropping depletes soil carbon (Squire *et al.*, 2015; Berdeni *et al.*, 2021). Given the role that soil carbon plays in the delivery of multiple other soil functions, there is a high likelihood that this could affect other ES (e.g. water storage and drought and flood resilience) (Grand-Clement *et al.*, 2013; Lal, 2015; Lal, 2016)

Using the data presented here it is possible to consider the scale of soil degradation across each level of intensity. This can provide valuable evidence to the estate in understanding the performance of in-hand and tenanted farms, whilst evaluating where management changes need to be made. A recent study on soil degradation in the UK has classified soil conditions based on the ratio of clay and SOC (shown here as n-potential, i.e. clay:SOC), with each tier defined as very good <8, good 8 – 10, moderate 10 – 13 and degraded >13 (Prout *et al.*, 2020). By these standards, the mean soil conditions for the highest intensity farm (n-potential 12.78 ± 4.04) were moderate but towards degraded in some fields. For farm intensities 2 – 4 conditions would be classified on average as good with some evidence of moderate condition. Only under intensity 5 (long-term organic pasture) were soil conditions considered to be very good (n-potential 7.43 ± 2.62). The data suggest that practices conducted under the most intensively managed farm (Farm 1) will degrade estate soils whereas well-protected lower intensity paddock grazed pasture (Farm 5) is likely to lead to good soil NC condition.

In contrast to the patterns for soil carbon stocks and n-potential, biomass yield significantly increased with increasing farm intensity. The data shows a clear trade-off, therefore, between productive output and soil carbon stocks and soil stability (n-potential). Whilst it might be expected that certain soil properties would be degraded under agricultural intensification, this is a concerning finding, particularly under the changing state of agri-environment policy due to Brexit. It has been noted that there is an increasing risk of intensification, with farms pursuing higher yields to offset losses in CAP Pillar One payments (Arnott *et al.*, 2021). The implications of the trade-off between soil carbon storage and biomass yield is discussed further in Section 4.4.4.

4.4.3 Addressing research question 2: What are the likely drivers of differences in soil natural capital condition across the study farms?

Using baseline soil NC condition data alongside field management data, it was possible to identify potential drivers of soil NC condition.

The time since the field was last tilled and the frequency of primary tillage were identified as the best predictors of soil carbon stocks (t ha^{-1}). However, it is important to discuss the high multicollinearity issues identified between the number of tillage operations, the number of crops, the number of years the field was in grass and the mean annual total input of N. In the model selection process the number of primary tillage operations was selected as it had a lower VIF than the other terms in the alternate model. The strong relationship between these management variables makes it hard to identify an exact driver(s) of soil carbon depletion. However, all of these variables relate to the intensity of the rotation and the turnover of crops that involved primary tillage cultivations during the previous 6 years. Therefore, it seems reasonable to conclude that it is likely that short term rotations and frequent changes in cropping (necessitating primary tillage) reduces soil carbon stocks. This finding also supports literature identifying the benefits of reduced tillage (Busari *et al.*, 2015; Büchi *et al.*, 2017; Haddaway *et al.*, 2017) and paddock grazed pasture (Whitehead, 2020) on building/maintaining soil carbon stocks.

The best predictor of soil stability (n-potential) was also linked to tillage, identified as the time since the field was last tilled. As discussed above increasing the length of time without tillage - where the fields were under grass ley or pasture - increases SOC and improves soil stability (i.e. reduces risk of soil erosion). This suggests that a higher proportion of clay is complexed with SOC under reduced tillage scenarios, particularly under longer-term grassland. The finding supports others noting that reduced soil disturbance, particularly in sandy-loam soils, improves carbon storage and stability (Haddaway *et al.*, 2017; Merante *et al.*, 2017). It suggests that increasing soil disturbance will reduce soil stability and increase the vulnerability of the soil to erosion (Merante *et al.*, 2017) potentially increasing the risk of soil run-off and pollution of surface waters.

Interestingly, the input of organic matter did not seem to have a significant effect on soil carbon stocks or n-potential. However, this could be explained by soil texture. Merante *et al.,* (2017) note that on sandy-loamy soils, one of the most critical practices to improve carbon storage and to reduce n-potential is to protect the soil through reduced tillage, suggesting that subsequent practices to increase SOC (e.g. FYM additions) must be carried out under no-till systems.

Reporting on the drivers of soil NC change based on the multi-model selection approach does have uncertainties, not least due to the multi-collinearity issues outlined above. It is recognised that monitoring change under experimental conditions would offer a more robust means of understanding the drivers behind soil conditions. However, what is shown here is that when taking baseline soil NC condition measurements and making some *a-priori* judgements it is possible to select appropriate predictor terms that can provide valuable insight into the potential drivers of change. Such an approach is valuable when applied at the farm scale, providing a relatively rapid understanding of what practices could impact soil NC condition and warrant further investigation. For example, the findings suggest that no or minimum tillage and the incorporation of longer-term leys in the rotation require further investigation and could offer a means to address soil carbon degradation issues at the most intensively managed farms on CDE.

4.4.4 Addressing research question 3: Does degradation of soil natural capital impact on productive output?

Data show that the soil NC conditions measured here had no significant impact on productive output. The plots in Figure 4.3 alone provide compelling evidence to suggest that degradation of soil carbon stocks and soil stability under increasing farm intensity does not significantly impact productivity, which increased with increasing intensity. Model results presented in Figure 4.5 further validate this, showing that only the cost of crop production had a significant effect on biomass yield.

The finding that cost of production had a significant impact on crop yield suggests that higher management and fertiliser/herbicide inputs ensure that having some poor soil conditions (e.g. depleted soil carbon and lower soil stability) does not appear to limit production. The data shows that with increasing intensity it is possible to deliver higher yields (although this is associated with higher costs), despite some soil conditions being degraded. This could suggest that farmers do ‘feel’ (to some extent) the damage from poorer soil condition (i.e. higher costs). However, on balance, the increase in yield outweighs the implications of poor soil condition. Further research would be useful to understand where there might be diminishing returns on production costs under certain soil conditions and where balance points are achieved between soil condition, CoP and productive output. The finding highlights the complexity of disentangling natural drivers (e.g. soil condition) from management drivers (e.g. crop pest and disease management). In the case of crop production it is clearly not only soil NC condition that impacts biomass yield but also inputs of manufactured capital.

What is clear from the data is that some soil properties can be degraded without obviously impacting crop yields. This suggests that there are clear externalities in the production process: most notably, carbon depletion is a negative externality that arises from increasing productive output. This finding aligns with those of Graves *et al.*, (2015) who report that many of the impacts of soil degradation in England and Wales occur “off-site” with the costs borne by third parties without compensation. The data reinforces the need for mechanisms that incentivise farmers to conserve soil carbon in order to ensure the delivery of soil-based ES such as climate regulation and flood alleviation.

It is perhaps unsurprising that no clear links were found between soil properties and biomass yield. Despite an understanding that properties such as SOC and SOM play a critical role in addressing yield limiting factors like the provision of nutrients and water retention - few studies have directly linked individual or composite soil condition metrics to crop yields (Miner *et al.*, 2020). In contrast to this study, Oldfield, Bradford and Wood (2019) in their global meta-analysis, did link higher maize and wheat yields to increasing SOC (up to a threshold of 2%, which would cover most of the soils in this study). They recognise, however, that work is still required at local scales to disentangle the causative effects of SOC on yield (Oldfield, Bradford and Wood, 2019).

There is a chance that monitoring yield over multiple years would change the relationship between soil NC condition and biomass yield. The yield data presented here only covered 2019[8] and it is recognised that sampling over multiple years would offer the advantage of identifying the impact weather has on yield. 2019 was a good year for growing with higher than mean yields for many crops (DEFRA and National Statistics, 2021). However, the sandy-loam soils local to the area are vulnerable in drought years and it might be that sites with higher soil carbon would sustain higher yields in these drought conditions, as reported in other studies (Iizumi and Wagai, 2019; Kane *et al.*, 2021). Further work at the study site would be required, including the monitoring of soil conditions and crop biomass production to investigate whether such impacts are likely to occur.

4.5 Conclusion

This study shows that there is a clear need for primary data on soil NC condition at the farm scale when applying the NC approach to land management decisions and evaluation. Existing national soil data are not adequate to detect differences in soil NC at management appropriate scales. Indeed, application of national soil data would have over-estimated soil carbon stocks at four of the study farms and underestimated them at one of the study farms.

This data can be resource intensive to collect but, when examined alongside farm management records, can detect the significant impact of farm intensity on soil carbon and soil stability and be beneficial in understanding what might be driving changes in soil NC condition. Notably, the data suggests that the intensity of the rotation with more frequent tillage and reduced use of long-term grass leys was likely to degrade soil carbon stocks and decrease soil stability. This information is valuable for future estate decision-making, such as investigating reduced tillage and ley management and promoting it to tenants, monitoring tenancies and determining future tenancy agreements.

The data also helped highlight a clear trade-off in soil carbon stocks and crop production, two indicators closely linked to understanding climate regulation and provisioning ES. Quantifying these trade-offs is an important reason for undertaking the NC approach. However, whilst findings offer some useful information about soil condition and potential ES, they do not provide information about how other soil functions might be impacted by soil management.

[8] Covid-19 prevented further planned field work in 2020

Furthermore, it does not show whether higher soil carbon is “better” or “worse” than lower crop yields. Further, more resource intensive work is required to conduct these analyses and this is explored in chapters 5 and 7.

Chapter 5: Does conversion to organic farming improve soil natural capital condition and soil function?

5.1 Introduction

5.1.1 Soil natural capital and ecosystem services

Soil is one of our most important forms of NC and contributes to the delivery of multiple soil-based ES (Robinson, Lebron and Vereecken, 2009; Dominati, Patterson and Mackay, 2010; Baveye, Baveye and Gowdy, 2016). Soil NC can be considered the stock from which ecosystem goods and services flow (Dominati *et al.*, 2014), with soil functions playing a critical role in the transformation of static soil properties into services or disservices delivered to society. Healthy soil is important for the provision of food and fibre (Kopittke *et al.*, 2019) and the UK National Ecosystem Assessment (Smith *et al.*, 2011) identify that "Soil quality is linked to almost all other regulating services (e.g. nutrient cycling, biomass production, water quality, climate regulation, pollination, etc.) through the soil's capacity to buffer, filter and transform". The total value of these ES globally has been estimated at US$11.4 trillion (McBratney, Morgan and Jarrett, 2017).

Important soil-based ES and the connections with soil function and soil NC are shown in Figure 5.1 with examples detailed below. Soil is the most important medium for plant growth and with global population set to exceed 9 billion by 2050 (Muller *et al.*, 2017), it is important that healthy soils are sustained on which to grow food. The provision of clean drinking water is also strongly influenced by soil health with good soil structure, adequate SOM, good biological functioning and cation exchange capacity (CEC) important in attenuating pollutants in soil (Beaudoin *et al.*, 2005; Harris *et al.*, 2006; Knudsen *et al.*, 2006). Many of these soil attributes also play an important role in flood management (Ellis, Anderson and Brazier, 2021) and resilience to drought (Lal, 2016). In addition, recently there has been particular interest in the capacity for soils to sequester carbon and mitigate climate change (Minasny *et al.*, 2017). The '4 per mille Soils for Food Security and Climate' initiative launched by the COP21, has the goal of increasing SOM stocks by 0.4% per year. The supporters of the initiative suggest that meeting these targets could lead to the sequestration of between 2 – 3 Gt C year^{-1}, effectively offsetting 20 – 35% of global anthropogenic greenhouse gas emissions (Minasny *et al.*, 2017). Whilst there is disagreement over whether the '4 per mille' target is feasible (Van Groenigen *et al.*, 2017; Poulton *et al.*, 2018), it is widely accepted that maintaining soil carbon storage and increasing sequestration will help regulate the climate (Paustian *et al.*, 2019a) whilst contributing to the delivery of other soil-based ES.

Despite recognition of its importance, soil is being seriously degraded across the globe. Associated issues include erosion by wind and water, losses in soil carbon and soil biodiversity, contamination, salinization, acidification and reduced fertility (Lal, 2015; Smith *et al.*, 2016; Kopittke *et al.*, 2019). Globally, 52% of agricultural land is thought to be moderately or severely affected by soil degradation (ELD Initiative, 2015) and in the UK, 38.2% of arable land has been identified as having degraded soil (Prout *et al.*, 2020). The intensification of agriculture and the inappropriate management of soils is considered to be fundamentally responsible for soil degradation (Brazier *et al.,* 2011; Graves *et al.*, 2015) driving soil erosion, compaction and the mineralisation of carbon (Bilotta, Brazier and Haygarth, 2007; Bilotta *et al.*, 2008; Batey, 2009; Peukert *et al.*, 2014; Graves *et al.*, 2015). In the UK, damage to our soils is already having significant economic consequences with the annual costs of soil degradation in England and Wales estimated at £1.2 billion per year (Graves *et al.*, 2015). Soil degradation and agricultural management is also driving declines in water quality, with the quality of groundwater aquifers in the UK frequently exceeding drinking water standards due to nitrate leaching from agricultural soils (Stuart and Lapworth, 2016; Wang *et al.*, 2016). There is a critical need for agricultural solutions that reverse the degradation of soil whilst supporting multiple other ES, including food production.

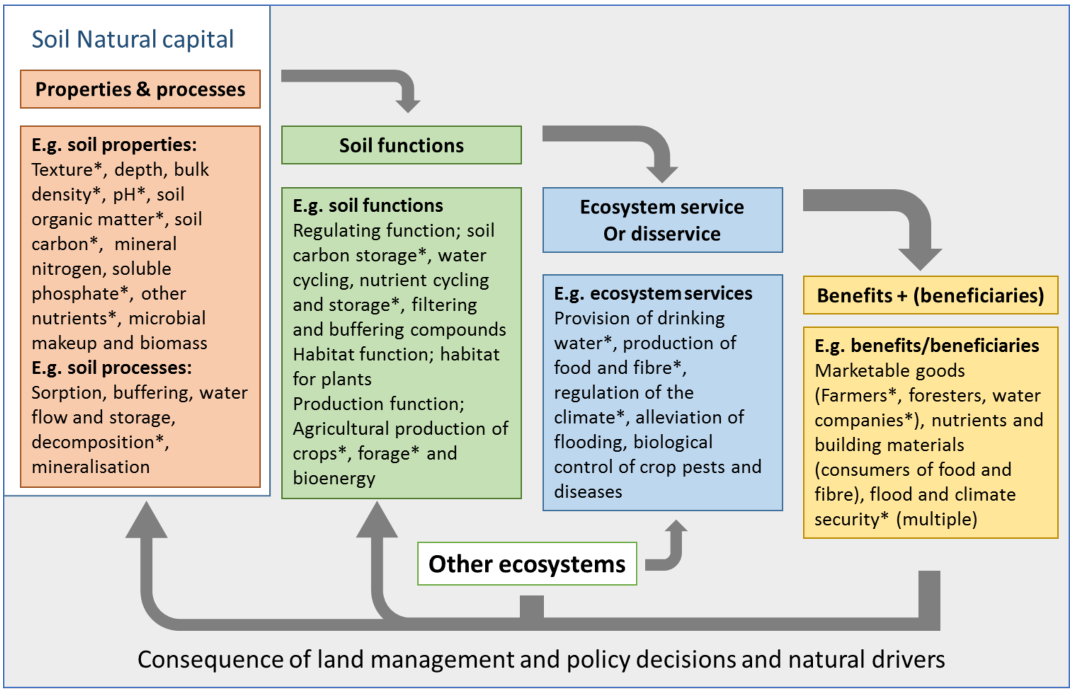


Figure 5.1: Flow pathway linking soil NC (its component properties), soil functions, ES and beneficiaries. Metrics at each stage are included (shown with asterisks). Framework developed by Haines-Young and Potschin (2008), as shown in Greiner et al., (2017).

5.1.2 Organic agriculture - a solution to restore soil natural capital?

Advocates of organic agriculture promote it as a potential solution which addresses some of the negative impacts of intensive agriculture and offers a balance between the provision of food and the delivery of environmental goods and services (Muller *et al.*, 2017; Wilbois and Schmidt, 2019). Tully and Mcaskill (2020) highlight that the concept of supporting a living soil is not a new one to organic farmers and central to the organic standards of the largest UK organic accreditation organisation (The Soil Association) is the preservation of soil health. The Soil Association (2021) specify that “organic matter, fertility and biological activity” should be maintained and increased through varied crop rotations, legumes, green manure crops and the application of livestock manures or other organic composts. These practices are commonly adopted by organic farmers, who cannot use artificial fertilisers or pest control, which leads to longer crop rotations, more frequent temporary fodder, higher crop diversity, the use of nitrogen fixing crops and the inclusion of more mixed farming practices (i.e. livestock and livestock manures) (Barbieri, Pellerin and Nesme, 2017). A number of these practices have been shown to enhance soil condition and function (Ball *et al.*, 2005; Powlson *et al.*, 2012; Döring *et al.*, 2013; Barbieri, Pellerin and Nesme, 2017; Sharma *et al.*, 2017).

Despite these promising features, there is widespread debate over the efficacy of organic farming in improving soil health and delivering soil-based ES, with scepticism over its role as a mainstream agricultural system (Connor and Mínguez, 2012). A number of field studies and meta-analyses have shown that organic farming, perhaps not surprisingly, delivers lower crop yields (De Ponti, Rijk and Van Ittersum, 2012) but can enhance water retention and soil structure (Lotter, 2003; Gomiero, 2013; Williams *et al.*, 2017), increase soil carbon stocks (Mondelaers, Aertsens and Huylenbroeck, 2009; Gomiero, Pimentel and Paoletti, 2011; Gattinger *et al.*, 2012; Tuomisto *et al.*, 2012), reduce nutrient losses (nitrate, nitrous oxide and ammonia) (Snapp, Gentry and Harwood, 2010; Tuomisto *et al.*, 2012; Benoit *et al.*, 2014; Biernat *et al.*, 2020) and improve soil biology and biological processes (e.g. the decomposition of organic matter) (Domínguez *et al.*, 2014; Lori *et al.*, 2017; Martinez-Garcia *et al.*, 2021). Other studies, however, have expressed doubts regarding some of these claims (Leifeld *et al.*, 2013; Kirchmann *et al.*, 2016) and some have found no significant differences in soil carbon concentrations (Gosling and Shepherd, 2005; Williams and Hedlund, 2013) or nutrient losses (Williams and Hedlund, 2013). It is likely that the conflicting results arise due to the broad differences in management that can occur both within organic and conventional systems (Williams and Hedlund, 2013). With growing interest in the expansion of organic farming - such as the European Union target to have 25% of the utilisable agricultural area under organic management by 2030 (Comissão Europeia, 2020) - there is a need to understand the consistency of organic farming in delivering soil-based ES better. The development of more context specific case-studies is therefore important in furthering our understanding of the implications of conversion to organic agriculture and the impact of the specific management practices adopted (e.g. cropping rotations). Indeed, Barbieri, Pellerin and Nesme (2017) highlight that there are few real farm system comparisons between organic and conventional farms (most have been on constrained experimental farms) and we have seen very few comparative studies on soil condition and/or soil function in the UK (although see Stopes *et al.*, 2002 and Gosling and Shepherd, 2005 for UK on-farm studies comparing soil condition and nitrate leaching, respectively).

5.1.3 Measuring soil natural capital and soil function to assess organic conversion

While the condition of soil NC underpins the functionality of the soil and the delivery of soil-based ES, how best to measure that condition remains a moot question (Greiner *et al.*, 2017; Williams *et al.*, 2017). Indeed, in the UK, there is still not an established consensus on how best to monitor soil condition (Humphries and Brazier, 2018): a fact acknowledged by the UK Government in their recent 25 Year Environment Plan where they set out plans to establish an agreed “soil health index” (HM Government, 2018). Therefore, despite the growing interest in using the NC approach to assess the sustainability of different land management systems, metrics to quantify soil NC condition and soil function to understand the delivery of soil-based ES are not well established. There is, of course, an extensive literature proposing methods that might best assess the quality of agricultural soils, with many authors suggesting different soil quality indexes (SQI) (Askari *et al.*, 2015; Obade and Lal 2016) or different indicators (Barrios, 2007; Brazier et al., 2011). However, the relationship between many of these indexes or indicators and the delivery of soil-based ES is still not well understood. Indeed, linking soil conditions with soil function and soil-based ES is complex, with the output of services or disservices strongly dependent on the interaction between soil conditions, processes and land management (Bartkowski *et al.*, 2020). In their review of soil ES and NC literature, Baveye, Baveye and Gowdy (2016) identify that most studies have failed to measure soil function directly. Measuring soil functions is particularly important as those functions represent the link between soil NC and the soil-based ES that it supports.

In lieu of an established consensus on measuring soil NC condition and soil function linked to soil-based ES, other studies have taken different approaches when inferring land management impact on soil-based ES delivery. Some have investigated multiple ES through the use of soil indicators (Williams and Hedlund, 2013; Calzolari *et al.*, 2016) or have modelled soil functions based on static soil properties (Dominati *et al.*, 2014). Others have focused on measuring specific individual soil functions such as nitrate leaching (Stopes *et al.*, 2002), organic matter decomposition (Lori *et al.*, 2017; Martinez-Garcia *et al.*, 2021), carbon storage (Machmuller *et al.*, 2015) or soil nutrients (Gosling and Shepherd, 2005).

In this study, it is proposed that to understand relationships between soil NC condition, soil functioning and soil-based ES, a combined approach is needed. Accordingly, this study examines a whole suite of indicators of soil NC condition as well as collecting measurements of detailed soil functions and examining how those might relate to multiple soil-based ES. Such an approach allows for a detailed assessment of the trade-offs that arise in the delivery of different potential soil-based ES as a result of pursuing organic as compared to conventional management.

The first phase of this study involved selecting appropriate properties to measure to assess NC condition or, where more appropriate, soil functions which could be linked to soil-based ES, attempting to trace these through the framework presented in Figure 5.1. Five broad categories were selected which link to one or more soil-based ES. These measurements of soil NC and function include: 1) measurements of soil structure and stability (BD and N-Potential (clay:SOC ratio)) as important indicators of water cycle regulation and the provision of food; 2) measurements of soil fertility (pH, SOM, bioavailable P, K and Mg) and the function of biomass production as indicators of food provision; 3) the measure of soil carbon storage as an indicator of climate regulation; 4) the measure of organic matter decomposition as an indicator of nutrient and carbon cycling, considering the links to crop production and climate regulation; and 5) measurements of nutrient storage and filtration (focusing on nitrate) to consider the implications for provisioning of clean drinking water. These selected metrics were used to compare between conventional and organic field sites to contribute to understanding on whether conversion to organic farming has the capacity to alter soil NC and soil function in ways that could enhance the delivery of soil-based ES.

5.1.4 Objectives of this study

The research focuses on a landscape in South West England that is taken as being broadly representative of the agricultural production practiced in lowland parts of the UK. A large part of the estate was converted to organic agriculture in 2007 and this study compares those organic field sites with those that remained under conventional management, making the assumption that prior to organic conversion in 2007 the organic fields would have had the same soil conditions as the neighbouring conventional fields. The aim was to establish whether conversion to organic agriculture has the capacity to enhance soil NC condition and soil function. Applying the suite of soil NC and soil function metrics introduced above, the research addressed the following two questions:

1. Do organic field sites have better NC condition than conventional sites and what practices might explain any differences?
2. Do organic field sites have enhanced soil function compared to conventional sites and what practices might explain any differences?

In addition, the study explored which metrics, based on the suite of soil properties and functions selected here, are likely to help inform the quantification of final soil-based ES.

5.2 Methods

5.2.1 Study site

The study was conducted at Clinton Devon Estate (CDE) in South West England (further details in Chapter 3, Section 3.1). A large part of the estate (ca. 900ha) was converted to organic agriculture in 2007, primarily for dairy and arable production. The agriculture before conversion was typical of neighbouring conventional farms, with winter cereals, maize, rye-grass/clover silage leys and improved pasture. Figure 5.2 shows the location of the site, study fields and soil sampling locations.

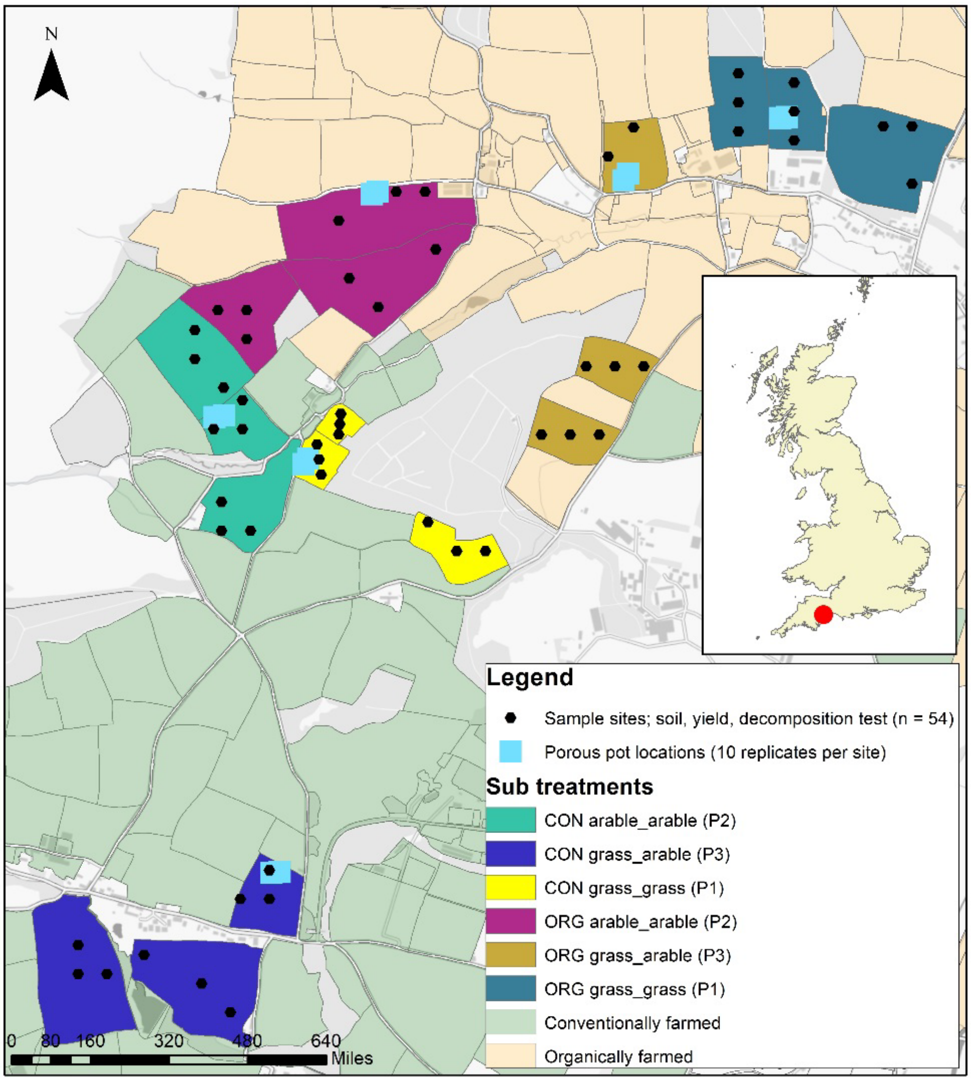


Figure 5.2: Map of the study site in SW England, including organic (ORG) and conventional (CON) cropped areas and the different stages of the rotation. Sample sites where soil and yield samples were collected and decomposition sites are marked as black dots. Porous pot sites are marked in light blue.

5.2.2 Site selection and experimental design

Nine conventional (CON) and nine organic (ORG) fields were selected, reflecting typical rotational land uses at CDE (Table 5.1). Three replicate fields for each system were selected that were within the grass phase of the rotation (Pair 1), the arable phase of the rotation (Pair 2) and transitioning out of the grass phase and into arable (Pair 3) (map of sites in Figure 5.2). Organic fields were selected from a single farm, with conventional fields selected from three neighbouring farms, to pair fields based on rotation stage. This was conducted to control for the effect of rotational stage or crop selection. It was not always possible to match field sites exactly[9] as the two systems vary in their rotations and cropping selection (Barbieri, Pellerin and Nesme, 2017). However, it was considered most important to select study fields representative of management of each system under the same soil and climatic conditions.

The performance of the farms managing both the organic and conventional study fields were similar (average/slightly above average) within their respective group (i.e. compared to other organic and conventional farms in UK). For example, in 2019, mean organic spring wheat yields were 4.13 t ha^{-1}, whereas the national average for organic spring wheat that year was 4 t ha^{-1} (Scott, 2020). Mean conventional winter wheat yields were 9.69 t ha^{-1}, also above the national average of 9 t ha^{-1} (DEFRA and National Statistics, 2021). Organic fields were managed on a five-to-six year rotation, with three years under grass-clover ley (cut for silage) before spring cultivation and two to three years sown as spring whole-crop cereals (primarily triticale and wheat). The conventional fields were typical of management on the organic farm before 2007 (dominated by maize and winter cereals) and had a shorter cropping rotation, typically a reduced length of time under grass leys, a higher turnover of crops and higher inputs of nutrients (as shown in Table 5.1). All the fields selected (ORG and CON) were planted with crops typically used as feed for dairy animals.

Table 5.1 provides a summary of the cropping, cultivations, frequency of cropping, use of legumes and the organic and inorganic inputs for each study field. Based on farm records and discussions with each farmer, estimates were made of annual inputs in the six years prior to the study, of organic and inorganic N, P and organic matter (OM). Stable organic matter inputs were calculated as the proportion of applied OM expected to be transformed into soil humus and are based on isohumic coefficients for slurry and FYM used in Büchi *et al.*, (2019). All fields were on the same soil association (Bromsgrove 0541b), in an effort to minimise variability caused by variation in soil parameters, and were selected to have a similar aspect and slope (using Grouping analysis in ArcGIS 10.5.1 (ESRI, Redlands, CA, USA)).

[9] Some rotation stages are different between the organic and conventional fields selected. This relates to differences in previous management or the crops selected. Most notably the grass leys (in the grass – arable phase) were typically longer on the organic fields compared to on the conventional fields.

Three replicate sites were selected within each field, informed by previous sampling results (sub-set of sites from Chapter 4) for the measurements of soil NC condition and soil function. Nutrient filtering and retention (linked to the provision of clean drinking water) was determined on a smaller sub-set of six fields (three organic and three conventional) (sites shown in Figure 5.2).

Table 5.1: Overview of the management of the study fields, including crop details; current crops, previous crop, primary cultivation method, number of all crops and number of years with legume crop (including clover in leys) included in the six years prior. The organic and inorganic fertiliser inputs for 2019 are shown along with the estimated average annual inputs leading up to sampling (2013 – 2018); for organic and inorganic fertilisers, dry organic matter and likely stable OM that will be retained in the soil. The table reports higher frequency of cropping of the conventional system, the reduced reliance on legume crops, higher annual inorganic nitrogen inputs and lower organic matter applications. P1 – relates to the grass-grass phase, P2 – the arable-arable phase and P3 – the grass-arable phase of the rotation.

				2013 - 2018 cropping details			**2019 crop inputs (kg ha^{-1})**				**Pre-sampling estimated average annual inputs (kg ha^{-1})**			
Field code	**P**	**2019 Crop**	**Previous crop**	**Primary culti-vation**	**No. cro-ps**	**No. legu-mes**	**Inorg N input**	**Inorg P input**	**Org N input**	**Org P input**	**Inorg& org N input**	**Inorg & org P input**	**Organic DM input**	**Stable OM inputs**
Organic fields:														
O1	P3	Spring wheat	Grass ley (clover)	MB Plough	5	3	0	0	9	29	56	26	2360	657
O2	P3	Spring wheat	Grass ley (clover)	MB Plough	4	4	0	0	18	57	47	30	2270	635
O3	P3	Spring wheat	Grass ley (clover)	MB Plough	4	4	0	0	18	57	47	30	2270	635
O4	P2	Spring triticale	Stubble turnips	MB Plough	7	5	0	0	51	47	78	32	2303	626
O5	P2	Spring triticale	Stubble turnips	MB Plough	7	5	0	0	51	47	78	32	2303	626
O6	P2	Spring triticale	Stubble turnips	MB Plough	7	5	0	0	51	47	72	31	2192	598
O7	P1	Grass ley (clover)	Grass ley (clover)	MB Plough	6	3	0	0	111	26	67	24	2250	630
O8	P1	Grass ley (clover)	Grass ley (clover)	MB Plough	6	3	0	0	111	26	67	24	2250	630
O9	P1	Grass ley (clover)	Grass ley (clover)	MB Plough	6	3	0	0	111	26	67	24	2250	630
Organic summary:					**5.78**	**3.89**	**0**	**0**	**59**	**40**	**64**	**28**	**2272**	**629**

Conventional fields:														
C1	P1	Grass ley (clover)	Grass ley (clover)	MB Plough	6	1	205	0	0	0	139	35	1544	667
C2	P1	Grass ley (clover)	Grass ley (clover)	MB Plough	5	2	125	0	0	0	141	34	1081	667
C3	P1	Grass ley (clover)	Grass ley (clover)	MB Plough	5	2	128	0	0	0	167	35	1081	667
C4	P3	Fodder beet	Grass ley (clover)	MB Plough	?	3	104	0	23	38	102	16	560	140
C5	P3	Maize	ley (winter only)	TD & MB	9	0	97	0	95	33	190	34	1247	206
C6	P3	Maize	ley (winter only)	TD & MB	9	0	97	0	95	33	188	34	1247	206
C7	P2	Maize	Stubble turnips	MB Plough	9	1	62	31	6	37	142	36	1544	667
C8	P2	Maize	Stubble turnips	MB Plough	8	1	148	0	6	37	155	30	1029	445
C9	P2	Maize	Stubble turnips	MB Plough	7	2	62	31	6	37	127	33	1029	445
Conventional summary:					**7.25**	**1.33**	**114**	**7**	**26**	**24**	**150**	**32**	**1151**	**456**

Acronyms: MB = mould board plough, TD = top down cultivator, Org = organic (from FYM or slurry), Inorg = inorganic (from synthetic or mineral fertiliser), DM = dry matter and OM = organic matter

139

5.2.3 Indicators and functions selected to measure soil-based ES

As highlighted in the introduction, there are difficulties associated with linking measurements of soil condition and soil functions to soil-based ES. In this study, a suite of ecosystem goods and services were investigated. The selection of metrics used as indicators of these ES are shown in Table 5.2. The objective was to select soil properties or, where more appropriate, soil functions which could be linked to soil-based ES, attempting to trace these through the framework presented in Figure 5.1. Further justification for each of the study indicators is provided in Chapter 3, Section 3.5. Additional justification of why each of these indicators/measurements is important can be found in Appendix C.1.

Table 5.2: Soil properties and soil functions measured in this study and the associated link to soil NC and soil-based ES. Acronyms include: SOM (soil organic matter), SOC (soil organic carbon), P (phosphorus), K (potassium), Mg (magnesium), TC (total carbon), TN (total nitrogen) and con (conventional fields).

NC component	**Soil function**	**Soil-based ES**	**Study NC indicators**	**Study Soil functions**
Soil structure and stability	Filter and store water Support plant growth	Flood and drought alleviation Food provision	Bulk density (BD) N-potential (Clay:SOC ratio)*	NA
Soil fertility and medium for plant growth	Plant growth	Production of market good/Food	Soil components important for crop growth; soluble P, K, Mg, SOM, pH, BD, TC, TN, TC:TN ratio	Crop biomass yield
Carbon stock	Carbon sequestration	Climate regulation	Carbon stocks; SOC and BD Carbon storage potential: N-potential*	Carbon stocks above con baseline considered as sequestered carbon
Soil biological activity	Decomposition of organic matter Cycling of nutrients and carbon	Production of market good/Food Climate regulation	NA	Tea Bag Index Method (litter decomposition)**
Nutrient storage	Filtering and storage of nutrients	Supporting provision of clean drinking water	NA	Nitrate leaching

*N-potential indicator presented by Merante *et al.,* (2017)
** Tea Bag Index introduced by Keuskamp *et al.,* (2013)
NA – Not measured in this study

5.2.4 Soil sampling

54 soil samples were collected between November - December 2018, following the completion of the majority of the cropping, final silage cut and main grazing season across the sites. Sites were located in the field using a hand-held GPS (Nomad Trimble, Sunnyvale, CA, USA) (1 – 2m accuracy).

Soil samples were collected and processed following methods outlined in Chapter 3, Section 3.6.1.

5.2.5 Measuring soil-based ecosystem service indicators

5.2.5.1 Measuring soil structure and stability (linking to water cycle regulation and food provision)

BD, the mass of a unit volume of dry soil, is an important indicator in the level of pore space (i.e. the space available for air and water) within a soil, providing information on the level of compaction (Cardoso *et al.*, 2013) and is frequently used in soil-based ES studies (Greiner *et al.*, 2017). Soil compaction has been associated with significant flooding and crop productivity issues (Graves *et al.*, 2015). Soil BD was calculated as shown in Chapter 3, Section 3.6.1.

N-potential, is a measure of soil stability and the carbon storage potential of a soil and is calculated as the ratio of clay (%) to SOC (%). Soil stability is closely linked to the content of SOC and fine soil particles (clay and silt), which become associated in the development of soil aggregates (Merante *et al.*, 2017). A high n-potential (>10) suggests lower soil stability and greater capacity to store more carbon. A low n-potential (<10) suggests higher soil stability and lower capacity to store more carbon (Merante *et al.*, 2017). N-potential was calculated as shown in Chapter 3, Section 3.6.1.

5.2.5.2 Measuring soil fertility and crop yield (linking to production of market good and food provision)

Crop growth and yield potential is strongly influenced by the macro and micro-nutrients including N, P, K and Mg available to the plant and the pH of soil (which limits the uptake of nutrients) (Brady and Weil, 2008; Dungait *et al.*, 2012). SOM is critical in the release and storage of these nutrients (Brady and Weil, 2008): such as carbon (C) and nitrogen (N).

5.2.5.2.1 Soil fertility

Soil pH, bioavailable-P, K and Mg, total C and total N were analysed following methods in Chapter 3, Section 3.6.1.

5.2.5.2.2 Crop biomass yield

Crop biomass data were collected as a measure of crop yield in order to compare the productive output from organic and conventional field sites. All crops were grown as forage crops for dairy feed and the simplified assumption was made that higher forage biomass production (in dry matter (DM) tonnes per hectare) would equate to higher milk provision. The assumption does not account for quality differences in the forage produced. It is also important to note here that whilst biomass yields for different organic and conventional crop types are compared, they are all important feed crops and whole-crop cereals in organic dairy systems are a typical starch rich replacement to the maize used in conventional dairy systems.

Crop biomass yield was determined following methods in Chapter 3, Section 3.6.4.

5.2.5.3 Measuring carbon stocks (linking to climate regulation)

Carbon stocks are a frequently used indicator in soil-based ES studies (Greiner *et al.*, 2017) and provide information on the current carbon and CO_2 equivalents stored in soil. The assumption is made here that organic field conditions were very similar to neighbouring conventional fields prior to conversion in 2007, allowing an assessment of whether conversion to organic agriculture has increased carbon storage over time.

Carbon storage (t ha^{-1}) was calculated following Poeplau, Vos and Don (2017) as

$$C\ stock\ = SOC * BD * d$$

where SOC is SOC (%), BD is the BD of the soil (g cm^{-3}) (corrected for stone content) and *d* is the depth of the soil core (15cm).

The capacity for the soil to store carbon (N-potential) was calculated following Merante *et al.,* (2017).

5.2.5.4 Measuring organic matter decomposition (linking to nutrient and carbon cycling and contributing to climate regulation and food production)

The decomposition of organic material (e.g. plant litter or farm manures) by soil biota is an important soil process, ensuring the bioavailability of nutrients and determining whether soils become a carbon sink or source (Keuskamp *et al.*, 2013; Ghaley *et al.*, 2014a; Paustian *et al.*, 2019a; Ray *et al.*, 2020).

Organic matter decomposition rate was determined using the standardised and globally applied Tea Bag Index (TBI) method (Keuskamp *et al.*, 2013). Detailed explanation of the method can be found in Chapter 3, Section 3.6.2.

5.2.5.5 Measuring nitrate leaching (linking to the provision of clean drinking water)

The current loss of nitrate from agricultural soils is an ecosystem disservice, with management strategies to reduce nitrate leaching (below the drinking water limit of 50 mg NO_3 l^{-1}) offering the potential to improve the quality and provision of drinking water.

Three organic and three conventional study fields were selected to compare nitrate leaching and assess the capacity of each system to enhance the delivery of clean drinking water. The three fields that were selected on the organic land were chosen to represent the full range of the 6 year organic rotation (albeit over the two seasons). Three conventional fields were selected to match, as closely as possible, the cropping transitions of the organic rotation. A summary of these is provided in Table 5.3 (for further comparative details for each field pair see Appendix C.2). All sites were on the same soil association (Bromsgrove 0541b) and an initial scoping field exercise was conducted to confirm that the topsoil texture was the same at all sites (i.e. a sandy loam).

Table 5.3: Field pairs selected for porous pot sites and the crop transitions over the sample seasons: 2018 – 2019 and 2019 – 2020. The table also shows the rotation stage (either grass-grass, grass – arable, arable – arable or arable – grass) that the sampling was conducted at over the two seasons. Acronyms: Org = organic and Con = conventional.

Field pairs	**Crop transition for 2018 - 2019**		**Crop transition for 2019 – 2020**	
	Org field	**Con field**	**Org field**	**Con field**
Field Pair 1 (FP1)	Rye-grass clover mix staying in rye-grass clover (grass – grass)	Rye-grass clover mix staying in rye-grass clover (grass – grass)	Rye-grass clover mix staying in rye-grass clover[10] (grass – grass)	Rye-grass clover mix staying in rye-grass clover (grass – grass)
Field Pair 2 (FP2)	Grazed cover crop following spring cereal, into second spring cereal (arable – arable)	Grazed stubble turnips flowing winter cereal, into maize (arable – arable)	Rye-grass clover ley following spring cereal, staying in rye-grass clover ley (arable – grass)	Winter wheat following maize, staying in winter wheat[11] (arable – arable)
Field Pair 3 (FP3)	Rye-grass clover ley, into spring wheat (grass – arable)	Rye-grass clover ley, into fodder beet (grass – arable)	Grazed cover crop following spring cereal, into second spring cereal (arable – arable)	Grazed fodder beet following fodder beet, into maize[12] (arable – arable)

Porous pots were used to quantify nitrate leaching, with soil pore water collected on a fortnightly basis over two winter drainage seasons (2018 – 2019 and 2019 – 2020) (as in Lord and Sheperd, 1993; Stopes *et al.*, 2002). The cumulative loss of nitrate (kg N ha^{-1}) over the drainage season was calculated using a field scale water balance model ADAS IRRIGUIDE (Baily and Spackman, 1996), underpinned by field collected data on nitrate concentration from porous pot samples and additional information on soil type, crop details, ground cover, rooting depth and daily agrometeorological data.

Detailed methods on soil pore water sample collection and analysis and on the field scale water balance model are described in Chapter 3, Section 3.6.3. The

[10] The management of this field was actually scheduled to change. With a spring cereal crop introduced prematurely into the organic rotation due to poor grass performance. All porous pot sampling took place before these cultivations took place however and therefore the site is still considered representative of the grass –grass phase of the rotation.
[11] This was the closest comparison between the organic and conventional rotation at the time of site selection. Whilst there was some talk of the conventional field at the time going into grass it was instead put to winter wheat.
[12] The plans for this conventional field were altered due to a change in farm management but remained fairly comparable.

final output calculated was nitrogen (as nitrate, NO_3 - N) leached in kg N ha^{-1} across each of the six study fields (three organic, three conventional) for two drainage seasons.

5.2.6 Statistical analysis

Field means of soil properties, carbon storage, nitrate leaching, crop production and for organic and conventional fields were analysed for significant differences using Wilcoxon-rank/Mann Whitney-U tests in R (R Core Team, 2020). The impact of system (organic or conventional) on each soil function was determined using all data (n = 54). Linear or generalized linear mixed effects models were selected depending on the data distribution. System (organic and conventional) and rotation stage (grass-arable, grass-grass, arable-arable) were included as fixed effects and study field as a random effect. Rotation stage was included as a fixed effect to control for potential variance attributed to the phase of the rotation. It was not included as a random effect as it is not considered prudent to include random effects that have less than five levels (Harrison, 2015). Analyses were conducted using the lme4 package (Bates *et al.,* 2015). Post-hoc analysis for pairwise comparisons for each rotation stage were conducted in the emmeans package (Lenth, 2020). Significance was tested at $p<0.05$. All analyses were conducted in R (R Core Team, 2020).

5.3 Results

Figure 5.3 displays data on the key measurements of soil condition and function between the two systems and across the rotation stages. Table 5.4 provides a summary of all soil data collected from the different rotational stages sampled across organic and conventional field sites and includes a pairwise comparison of the means between organic and conventional across all field sites. Mixed effect model outputs can be seen in Appendix C.3.

5.3.1 Soil structure and stability

Organic field sites on average showed slightly more compaction than conventional sites, though there was no significant difference observed between the two systems (p = 0.4). The lowest BD values were found under the conventional arable-arable phase of the rotation (1.29 g cm^{-3}), with the highest (i.e. most compacted soils) found under the grass-grass phase of the organic rotation (P1) (1.44 g cm^{-3}) (Figure 5.3A).

The difference in N-potential between the two systems was not significant (p = >0.9). N-potential was marginally lower across the grass-arable and arable-arable phases of the organic rotation (Figure 5.3B), suggesting higher soil stability with more of the available clay in the soils bound to SOC. However, the grass-grass phase of the organic rotation had a higher n-potential, which coincides with the highest compaction (BD) and suggests impaired soil physical structure. The conventional arable-arable phase of the rotation had the highest overall n-potential, suggesting lower soil stability despite reduced levels of compaction.

5.3.2 Soil fertility and crop production

5.3.2.1 Soil nutrients and pH

Addressing crop nutrients, conventional sites had higher bioavailable-P (Olsen-P, 32 mg l^{-1}) and K (124 mg l^{-1}) when compared to organic fields sites, 20 mg l^{-1} and 99 mg l^{-1} respectively (Table 5.4). Olsen-P differences were close to being significant (p = 0.077) and the high levels may reflect the ready application of inorganic phosphate fertilisers by conventional growers. Organic fields, however, had significantly higher levels of Mg (77 mg l^{-1}, p = 0.003) than conventional fields (53 mg l^{-1}). pH was significantly higher across organic fields (6.61, p = 0.008), with conventional field sites having a fairly low pH (6.07). Particularly low pH was observed under conventional arable-arable and grass-grass fields, 5.91 and 5.83 respectively. Under such low pH these fields could show signs of reduced nutrient availability and as a consequence, diminished yields.

Whilst mean SOM (%), TC (%) and TN (%) were higher across organic fields there was no significant difference between the two systems. However, when calculating TN (kg N ha^{-1}) and carbon stocks (kg SOC ha^{-1}) (incorporating soil bulk densities), organic field sites had significantly higher TN and carbon storage (p = 0.01). This suggests greater quantities of TN and organic matter within the topsoil (0 – 15cm) across the organic field sites.

Under organic management, bio-available P (mg l^{-1}), SOM (%), TC (%), TN (%) and Mg (mg l^{-1}) were highest in the grass-arable stage of the rotation (P3, Table 5.4). Samples were taken from fields before cultivation and prior to this, they had been under grass clover ley management for three years. Whilst the same response was not seen for K (mg l^{-1}) this could highlight the important role that the grass phase of the rotation plays in building soil fertility.

5.3.2.2 Biomass yields

Conventional biomass yields (13.93 t DM ha^{-1}), as expected, were significantly higher than organic yields (8.93 t DM ha^{-1}, *p* = <0.001) (Figure 5.3C). This shows that mean organic dry matter biomass yield (across grass silage and wholecrop cereals) was around 36% lower than the conventional dry matter biomass yield (across grass silage, maize and fodder beet yields). The same pattern is seen across all stages of the rotation. The highest biomass yields (16.4 t DM ha^{-}1) were observed in conventional forage crops (maize and fodder beet) following the grass phase of the rotation and the lowest from organic grass ley silage cuts (7.2 t DM ha^{-1}). Comparing yields from the conventional grass clover leys (mean 12.4 t DM ha^{-1}) with the organic clover leys (mean 7.2 t DM ha^{-1}), we see that conventional biomass is 42% higher.

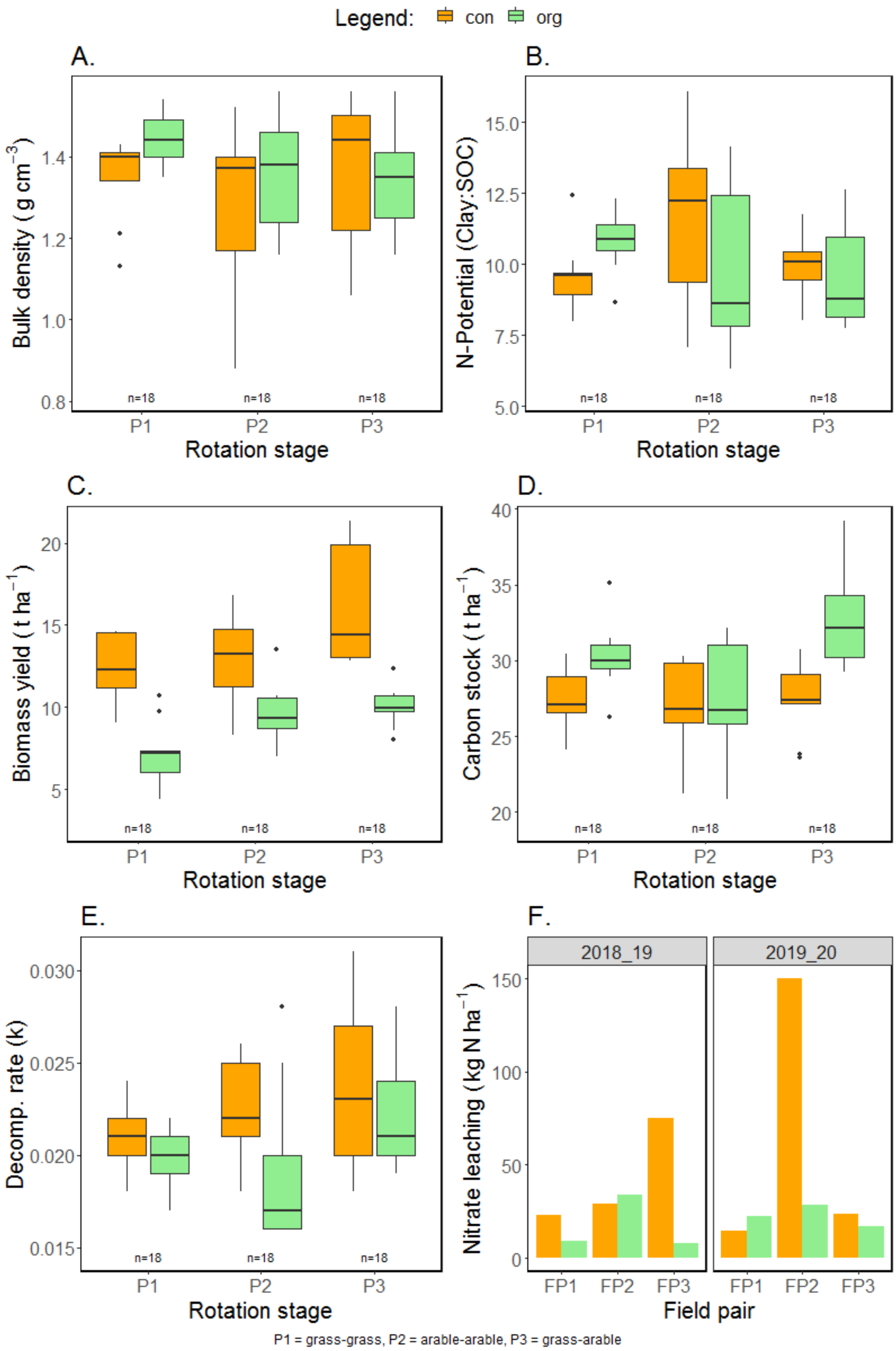


Figure 5.3: Soil properties and soil functions across the different stages of the rotation for conventional (orange) and organic (green) field sites. The charts are as follow: A) BD (g cm^{-3}); B) N-potential (ratio of clay to SOC); C) Biomass yield (t ha^{-1}); D) Carbon stocks (t ha^{-1}); E) decomposition rate (TBI k); and F) Nitrate leaching (kg N ha^{-1}). Nitrate leaching shows the comparisons across the three field pairs (FP1 – FP3) and the rotation stages these relate to can be found in Table 5.3.

5.3.3 Soil carbon storage

Carbon stocks were significantly higher under organic fields (30.28 t ha^{-1}) than under conventional fields (27.15 t ha^{-1}) when controlling for differences between rotation stages and within fields (p = 0.01). Similar to the results for SOM and TC, Figure 5.3 and Table 5.4 show that the highest mean levels of carbon stocks are under the organic grass-arable phase of the rotation (32.8 t ha^{-1}), sampled after the fields had been under grass-clover ley management for three years. Again, this highlights the potential role that the grass ley rotation phase of the rotation is having in increasing carbon storage. The lowest levels of carbon stocks were observed in the arable-arable phase of the conventional rotation (26.8 t ha^{-1}), after a period where soils for most fields had been under continuous arable cropping for some time (four years).

There were no significant differences in the n-potential between conventional and organic field sites. However, the higher n-potential values (>10) observed in the conventional arable-arable and the organic grass-grass phases of the rotation show that these field sites, in particular, have the potential to store more carbon (I.e. a higher abundance of clays not bound to soil carbon).

5.3.4 Organic matter decomposition rate

Conventional field sites show signs of having a higher decomposition rate (0.022) compared with the organic sites (0.020) and there was a greater mass loss of organic material from both red and green teabags. However, there was considerable variability between sites and decomposition rate constant (k) was not significantly different between each system (p = 0.14). The pattern of higher decomposition rates is seen under all rotation stages but the difference is most distinct between treatments in the arable-arable phase of the rotation (Figure 5.3E).

5.3.5 Nitrogen storage and nitrate leaching

The mean loss of nitrogen (as nitrate) was much higher from conventional sites 52.64 kg N ha^{-1}, compared to organic field sites 19.85 kg N ha^{-1} (over the 2018-19 and 2019-20 drainage seasons). Wilcoxon Rank Sum test results show that the difference between systems, however, is not significant (p = 0.18). Figure 5.3F shows the pairwise comparison of nitrogen lost from the fields for the 2018-19 and 2019-20 seasons. There is considerable variability between different sites, with high losses under the conventional grass-arable stage of the rotation (FP3) in 2018-19 (where heavy winter grazing was observed) and the arable-arable phase (FP2) of the rotation in 2019-20 (a winter wheat crop following maize). It is likely that the low sample size and the significant variability in N leaching data between sites explains the insignificance of the statistical test. As previously discussed, TN stocks were significantly higher ($p = 0.01$) across organic field sites (3.08 t ha^{-1}) compared with conventional field sites (2.57 t ha^{1}). The pattern matches that of carbon stocks and it is likely that much of the stored nitrogen is in organic matter form as opposed to bioavailable and water-soluble forms (which are more easily lost due to leaching or runoff).

Table 5.4: Summary statistics (mean and standard deviations) for each sampled soil property and soil function across the different paired stages of the conventional and organic rotations. Means for each system and pairwise statistical analysis results (using Wilcoxon Rank Sum test) are also shown.

	Pairwise comparison between rotation stages						**System comparison (field means)**		
	Conventional			**Organic**					
Variable	**Arable-Arable (P2) (c), N = 9[1]**	**Grass-Arable (P3) (c), N = 9[1]**	**Grass-Grass (P1) ((c), N = 9[1]**	**Arable-Arable (P2) (o), N = 9[1]**	**Grass-Arable (P3) (o), N = 9[1]**	**Grass-Grass (P1) (o), N = 9[1]**	**Organic , N = 9[1]**	**Conventional, N = 9[1]**	**p-value[2]**
SOM (%)	2.75 (0.68)	2.64 (0.51)	2.60 (0.23)	2.67 (0.55)	3.20 (0.52)	2.69 (0.16)	2.85 (0.39)	2.66 (0.40)	0.5
SOC (%)	1.43 (0.35)	1.37 (0.27)	1.35 (0.12)	1.39 (0.29)	1.66 (0.27)	1.40 (0.08)	1.48 (0.20)	1.38 (0.21)	0.5
TC (%)	1.45 (0.34)	1.36 (0.23)	1.30 (0.16)	1.42 (0.36)	1.63 (0.30)	1.40 (0.18)	1.48 (0.22)	1.37 (0.20)	0.2
TN (%)	0.14 (0.04)	0.13 (0.02)	0.13 (0.03)	0.14 (0.03)	0.17 (0.06)	0.14 (0.02)	0.150 (0.035)	0.131 (0.021)	0.2
C:N Ratio	10.88 (0.85)	10.23 (0.54)	10.55 (1.51)	10.53 (1.36)	9.79 (1.88)	10.05 (1.28)	10.12 (0.97)	10.55 (0.61)	0.5
BD (g cm^{-3})	1.29 (0.21)	1.36 (0.19)	1.35 (0.11)	1.36 (0.14)	1.33 (0.13)	1.44 (0.06)	1.38 (0.10)	1.33 (0.11)	0.4
pH	5.91 (0.16)	6.48 (0.25)	5.83 (0.38)	6.78 (0.50)	6.40 (0.37)	6.64 (0.18)	6.61 (0.30)	6.07 (0.39)	0.008**
Olsen-P (mg l^{-1})	28 (15)	50 (12)	20 (9)	19 (13)	27 (8)	14 (5)	20 (8)	32 (16)	0.077
K (mg l^{-1})	137 (65)	129 (91)	106 (25)	143 (154)	102 (34)	54 (13)	99 (66)	124 (57)	0.2
Mg (mg l^{-1})	52 (16)	59 (16)	47 (4)	74 (20)	74 (27)	82 (13)	77 (17)	53 (12)	0.003**
Clay (%)	16 (4)	14 (3)	13 (2)	13 (4)	16 (4)	15 (1)	14.70 (1.87)	14.15 (2.45)	0.6
N-potential	11.43 (3.05)	9.88 (1.20)	9.65 (1.21)	9.62 (2.83)	9.67 (1.91)	10.84 (1.08)	10.04 (1.21)	10.32 (1.42)	>0.9
DM biomass yield (t ha^{-1})	12.9 (2.8)	16.4 (3.6)	12.4 (2.0)	9.5 (1.9)	10.0 (1.3)	7.2 (2.0)	8.93 (1.79)	13.93 (2.99)	<0.001***
Carbon stocks (t ha^{-1})	26.8 (3.5)	27.4 (2.4)	27.2 (2.0)	27.8 (3.8)	32.8 (3.1)	30.2 (2.4)	30.28 (2.67)	27.15 (1.95)	0.011*
Nitrogen stocks (t ha^{-1})	2.52 (0.39)	2.66 (0.28)	2.52 (0.49)	2.73 (0.59)	3.46 (1.23)	3.05 (0.57)	3.08 (0.60)	2.57 (0.23)	0.019**
TBI_k	0.023 (0.003)	0.023 (0.005)	0.021 (0.002)	0.019 (0.004)	0.022 (0.003)	0.020 (0.002)	0.020 (0.0028)	0.022 (0.0021)	0.2
N leaching (kg N ha yr^{-1})							19.85 (8.67)[3]	52.64 (30.67)[3]	0.18

[1]Mean (SD)
[2]Wilcoxon rank sum exact test; Wilcoxon rank sum test
[3] n for nitrate leaching means is 6 per system (3 fields * 2 years)
(c) denotes conventional rotational stage and (o) denotes organic rotational stage

5.4 Discussion

The results from this study show that conversion to organic agriculture has the capacity to significantly increase soil carbon stocks. However, this comes with a clear trade-off in crop biomass yield. Applying a suite of different indicators of soil-based ES, findings showed that whilst conversion to organic agriculture can deliver some benefits, it did not systematically improve soil NC condition or soil function. The discussion over the impacts of organic conversion first on NC condition and second on soil function is expanded below, considering what land management practices might explain these differences and the potential trajectory of soil conditions into the future. To finish, the use of different metrics as indicators of soil-based ES is discussed identifying those that were more or less useful in evaluating the impacts of land management change.

5.4.1. Addressing Research question 1: Do organic field sites have better NC conditions than conventional sites and what practices might explain these differences?

5.4.1.1 Soil structure and stability (Indicators of water cycle regulation and good crop growing conditions)

Despite literature suggesting organic farming (Gomiero, Pimentel and Paoletti, 2011; Williams *et al.*, 2017) and the practices commonly incorporated within organic management (e.g. grass-clover leys) (Berdeni *et al.*, 2021) can enhance soil structure, there was no significant improvements in BD or n-potential in this case-study. Williams *et al.,* (2017) also found no significant differences in bulk densities between conventional and organic treatments during a 40-year study and it seems reasonable to suggest that appropriate soil management, particularly appropriately timed management (e.g. timing of cultivations, harvest, applications) is arguably more important than agricultural system.

This study observed poor soil structure and soil stability across both conventional and organic field sites at different stages of the rotation. Poor soil structure here is judged against the National Soils Research Inventory (NSRI, 2005)[13] for the same soil series (Bromsgrove). NSRI (2005) shows a mean BD of 1.34 g cm^{-3}

[13] Note even NSRI (2005) soil conditions are likely to show a shifting baseline, representing soil conditions typical across agricultural landscapes that have been intensively managed for at least a couple of decades.

and calculated n-potential[14] of 7 (clay:SOC) for arable land and 1.21 g cm^{-3} and 7 (SOC:clay) for ley grassland.

Results show that during the arable phases of both organic (1.36 g cm^{-3}) and conventional (1.29 g cm^{-3}) rotations, soil BD data were in-line with the NSRI (2005) soil database. Conversely, in the grass ley phase of the rotation, BD data were on average higher (Conventional, 1.35 g cm^{-3}; Organic, 1.38 g cm^{-3}). The grass-grass phase of the organic rotation had a notably high mean BD (1.44 g cm^{-3}). These fields had been under grass for only one year with one unobserved, but potentially important, factor being whether the grass ley was established or had been tracked in wet field conditions, where the latter could have contributed to this compaction. Furthermore, it would be interesting to observe whether BD declines (i.e. there is a recovery in soil structure) over time under ley management, in fitting with the grass-arable phase of the rotation (where organic field sites had been in grass for three years). Under current conditions, the grass-grass field sites show impaired soil structure, likely to reduce infiltration and exacerbate run-off. N-potential (Clay:SOC) was typically higher across both organic (10.04) and conventional (10:32) field sites than the NSRI (2005) database, highlighting the typically low levels of SOC reported in this study. N-potential was notably high across conventional arable sites which, whilst having the lowest BD, had an n-potential of 11.43. This suggests a high proportion of non-complexed clay, increasing vulnerability of the soil to erosion (Merante *et al.*, 2017), which could exacerbate the loss of soils and the pollution of surface waters. The arable-arable soils were subject to the most frequent disturbance (regular cultivations), compared to other phases of the rotation and it is likely that this is driving the loss of SOC and contributing to reduced soil stability (Powlson *et al.*, 2012; Squire *et al.*, 2015; Büchi *et al.*, 2017; Jarvis and Woolford, 2017). This is covered in more detail in Chapter 4.

The study provides some positive evidence of organic management improving soil structure with relatively low mean BD (1.33 g cm^{-3}) and low n-potential (9.67) during the organic grass-arable phase of the rotation (following three years under grass-clover ley). This finding supports evidence that incorporating longer-term grass-clover leys in a rotation can support improvements in soil structure and

[14] N-potential figures are not presented in the NSRI (2005) data but can be calculated using figures on soil organic carbon (%) and clay (%).

could contribute to improving the delivery of water storage and flood attenuation ES (Berdeni *et al.*, 2021).

5.4.1.2 Soil nutrients (Indicators of supporting crop growth and provisioning services)

The results showed that organic conversion did not systematically increase all soil nutrients that play an important role in influencing crop production. Whilst some important nutrients were significantly higher at organic field sites (e.g. bioavailable Mg), other properties (e.g. bioavailable P and K) were not significantly different between the two systems.

Findings did report significantly higher nitrogen stocks under organic management (in agreement with Birkhofer *et al.*, (2008)). However, given the similar C:N ratios of the two systems, this is likely to be down to the higher SOM content. Despite higher presence of nitrogen, mineralisation is required in order for it to become available to the plant, which typically does not coincide with when crops require it most (Berry *et al.*, 2002; Wilbois and Schmidt, 2019). Wilbois and Schmidt (2019) report that natural sources of nitrogen therefore play a lesser role than mineral nitrogen fertilisers in determining crop yields and organic yields are often fundamentally limited by the availability of mineral N (Berry *et al.*, 2002; Seufert, Ramankutty and Foley, 2012; Bilsborrow *et al.*, 2013). Interestingly, the highest nitrogen stocks (3.46 t ha^{-1}) and lowest C:N ratio (9.79) were observed under the organic grass-arable phase of the rotation (where the fields had been in rye-grass and white clover ley for three years). This supports the principles of including fertility building leys in the rotation, which are likely to provide further bio-available N following cultivation (Mäder *et al.*, 2007). The same pattern was also observed under the conventional grass-arable phase of the rotation.

Despite lower TN at the time of sampling[15] , the conventional fields generally had higher mean bioavailable P and K. This is perhaps unsurprising given the lower inputs across organic compared to conventional field sites shown in Table 5.1. This trend is in keeping with Gosling and Shepherd (2005) who found significantly lower P and K on organic fields that had been under organic management for 15 or more years. They argue that long term organic management runs the risk of

[15] It would be interesting to observe if sampling immediately after spring applications of fertilisers would alter results, showing higher N at conventional field sites. Note all sampling in this study was conducted in autumn.

mining existing P and K resources from previous conventional management (Gosling and Shepherd, 2005) implying that without appropriate management these nutrients will become increasingly scarce in the organic system (Cooper *et al.*, 2018). Interestingly, however, across the organic fields, the highest bioavailable P concentrations, along with higher SOM, TN and TC, were observed following the 3-year organic grass clover ley phase of the rotation. The finding demonstrates the capacity for extended grass and legume leys to build fertility and suggests that when incorporated into organic rotations, they can go some way to maintaining bioavailable P. This outcome is contingent on the fields being managed as a closed loop with the crop P removed (following silage harvest) being replaced via farmyard manures.

5.4.2. Addressing Research question 2: Do organic field sites have enhanced soil function when compared to conventional sites and what practices might explain these differences?

5.4.2.1 Crop biomass production (Indicator of the market good production and provision of food)

As would be expected, crop biomass production was significantly lower from organic sites. This is in agreement with the majority of the literature comparing the outputs from organic and conventional farming (De Ponti, Rijk and Van Ittersum, 2012; Seufert, Ramankutty and Foley, 2012; Ponisio *et al.*, 2014; Wilbois and Schmidt, 2019). Organic DM biomass yields in this study were 68% of those achieved on the conventional field sites. Although it is difficult to make comparisons of yield quantity between some of the different crops in this study (such as maize and cereals), when comparing like-for-like conventional and organic grass-clover leys relative organic yield were 58% of conventional yields (although this was skewed by one particularly low yielding organic site). The relative yield findings in this case-study are lower than those reported by De Ponti, Rijk and Van Ittersum (2012) in their meta-analysis of the difference in yields between the two systems (79%), though they highlight that there is a considerable difference between crops across different regions.

The lower biomass yields are likely to be explained by one or a combination of fertility management, weed competition and pests and disease (Mäder *et al.*, 2007; Seufert, Ramankutty and Foley, 2012; Bilsborrow *et al.*, 2013; Wilbois and Schmidt, 2019). Mäder *et al.*, (2007) in a 21-year comparative study of wheat yields in central Europe attributed lower organic yields to the organic system receiving a 71% lower input of soluble nitrogen inputs. A similar pattern was observed in this study, with nitrogen inputs being 57% lower on organic compared to conventional field sites (Table 5.1). Interestingly, the highest DM biomass yields were observed in the grass-arable phase of the rotation (i.e. following cultivation out of a grass ley and into a cereal/maize crop) across both the organic (10 t DM ha^{-1}) and conventional (16.4 t DM ha^{-1}) field sites. As discussed previously, these field sites also had the highest nitrogen stocks (t N ha^{-1}) and lowest C:N ratios within both systems. The data supports other studies where higher yields followed cultivation out of grass/clover leys and further highlights the importance of the grass ley phase of a rotation in building soil fertility (Mäder *et al.*, 2007; Bilsborrow *et al.*, 2013).

5.4.2.2 Carbon storage (Climate regulation)

Organic field sites showed significantly higher soil carbon stocks than conventional field sites, which supports other literature showing that organic systems typically store more soil carbon (Mondelaers, Aertsens and Huylenbroeck, 2009; Snapp, Gentry and Harwood, 2010; Gattinger *et al.*, 2012; Tuomisto *et al.*, 2012).

If the assumption is made that conventional and organic soil sites were the same at CDE immediately ahead of conversion to organic agriculture in 2007, it is possible to consider the potential sequestration rate across organic field sites during that time. This is a reasonable assumption given the sites were on the same soil association, sharing the same climate and previously being subjected to similar farm management practices. Taking the difference between carbon stocks between conventional and organic field sites (mean increase of 3.13 t ha^{-1}) and the time since conversion (11 years), it can be estimated that organic conversion has sequestered around 285 kg C ha^{-1} yr^{-1}. This is lower than the mean carbon sequestration rate used in Harris *et al.*, (2006) (479 kg^{-1} C ha^{-1}) to estimate the ES benefits of organic conversion in the UK but it is well within the range of the findings from the Gattinger *et al.,* (2012) global meta-analyses (mean = 450 ± 1005 kg C ha^{-1} yr^{-1}).

A meta-analysis by Tuomisto *et al.*, (2012), show that higher carbon sequestration in organically managed soils is mainly explained by higher inputs of organic matter (which can be seen for the study fields, Table 5.1). It would also appear that the incorporation of longer grass clover leys into the organic rotation is contributing to higher carbon storage (alongside other nutrients) and the highest carbon stocks and lowest n-potential (<10) were observed at sites that had been under ley management for the previous three years (the grass-arable phase of the rotation). Long-term studies incorporating grass leys into conventional arable rotations in the UK (on similar sandy soils), have also found that the intervention significantly improves carbon stocks (Poulton *et al.*, 2018). This finding supports the principles that reduced soil disturbance, particularly in sandy-loam soils improves carbon storage and stability (Haddaway *et al.*, 2017; Merante *et al.*, 2017).

Interestingly, under the organic arable-arable phase of the rotation reduced carbon stocks and higher n-potential (in-line with conventional arable-arable sites) were observed. This could suggest that following soil disturbance (ploughing out the grass clover ley) much of the stored carbon becomes mineralised, potentially caused by the destruction of soil aggregates and the exposure of the previously "protected" carbon stored within aggregates (Paustian *et al.*, 2019b). However, due to the relatively shallow soil sampling depth (15cm) used in this study it could be that the soil carbon accumulated during the grass ley phase has actually been 'buried' following ploughing. As both organic and conventional fields were primarily cultivated using traditional mouldboard ploughing with a plough depth of ca.20cm, it could be that the inverted soil carbon was below the sampled depth. This is a limitation of the 15cm sample depth used in this study, which, whilst selected to match the Countryside Survey (Emmett *et al.,* 2010) and be resource efficient, fails to detect SOC changes at depth. Optimising sampling depth in soil carbon sequestration studies is an ongoing debate (Baker et al., 2007; Olson and Al-Kaisi, 2015; Vandenbygaart *et al.,* 2011) and should be considered in future studies. A deeper sampling core, for example to 30cm (Vandenbygaart *et al.,* 2011), would be needed in the future to validate whether the carbon accumulated during the organic grass phase of the rotation is being mineralised or is actually being stored at depth at the study sites.

Whilst higher carbon stocks were observed at organic field sites, levels across both systems were relatively low and in all but the organic grass-phase of the rotation, n-potential was greater than 10 suggesting that there is some capacity for greater carbon storage. It is reasonable to expect that the sandy-loamy soils, typical of the study area, would have fairly low carbon storage potential (Merante *et al.*, 2017) but the carbon stock levels fall within the central to lower end of the range reported for the sandy soils (13% clay) at the long-term Woburn Trials, 22.7 t ha^{-1} – 40 t ha^{-1} (0 – 23cm) (Poulton *et al.*, 2018). They are also considerably lower than mean stocks, reported in topsoil (0 – 15cm) across the UK, as part of the Countryside Survey 2007 for arable (46.9 t SOC ha^{1}) and improved grassland (64.6 t SOC ha^{-1}) (Emmett *et al.,* 2010). Given the changes made under organic management (e.g. longer rotation, the incorporation of grass clover leys and higher farmyard manures (FYM) inputs), it is surprising that the differences in carbon stocks between the two systems are not more noticeable. There is, however, a chance that the impact of management changes are still coming into effect. Poulton *et al.,* (2018) report consistent carbon sequestration under the addition of FYM for 40 – 60 years at Broadbalk and Hoosfield experimental sites, gradually slowing after this period as a SOC stock equilibrium was reached. This suggests that over decadal time scales, continued organic management, incorporating FYM and extended grass clover leys could deliver enhanced benefits in the form of carbon storage. Merante *et al.,* (2017) are explicit, however, that on sandy-loamy soils, one of the most critical practices to improve carbon storage and to reduce n-potential is to protect the soil through reduced tillage, suggesting that subsequent practices to increase SOC (e.g. FYM additions) must be carried out under no-till or minimum tillage systems. Reduced and, particularly, no-tillage systems present a significant challenge for organic farmers, who cannot use herbicides and rely on cultivations for weed control (Tully and Mcaskill, 2020). Continued tillage under organic systems, to reduce weed burdens and maintain crop yields could therefore limit the capacity to store more carbon in light sandy and sandy-loamy soils.

Considering the efficacy of organic management on different soil textures to deliver soil-based ES is important and the findings highlight the need for the wider expansion of comparative farm studies on other soil types in the UK. This case-study has also demonstrated the importance of long-term monitoring, to identify whether the practices adopted under organic management will continue to increase carbon storage over time. Further research would also be beneficial to understand carbon fluxes (perhaps using recent advances in low-cost eddy covariance (see Ramshorst *et al.*, 2020)) through organic rotations to determine whether, despite losses between grass leys, the rotation has a net benefit on soil carbon stocks.

5.4.2.3 Organic matter decomposition (supporting nutrient cycling and carbon sequestration)

Decomposition rates were not significantly different between organic and conventional field sites. Whilst it had been predicted that decomposition rates might be higher on organic soils - in-line with meta-analyses showing organic farming enhanced soil microbial abundance and activity (Domínguez *et al.*, 2014; Lori *et al.*, 2017) - this was not the case. DiekOtter *et al.*, (2010) also found no significant differences in litter-bag decomposition between conventional and organic sites in arable fields in Germany, in contrast to other studies which have found higher decomposition rates under organic agricultural management (Domínguez *et al.*, 2014; Martinez-Garcia *et al.*, 2021). Whilst not significantly different, conventional sites actually had higher mean decomposition rates across all phases of the rotation and there was a notable contrast in rates during the organic arable-arable and conventional arable-arable phase of the rotation.

Significantly higher application of nitrogen fertiliser (shown in Table 5.1) could be contributing to higher decomposition rates at conventional sites. Research has suggested that, whilst the scale and processes are still not fully understood, nitrogen fertilisation can alter SOM decomposition and carbon storage (Man *et al.*, 2021). Birkhofer *et al.*, (2008) suggest that SOC is more accessible to microorganisms in conventional systems, with excess nutrients in the soil shifting the structure of the microbial community towards early successional species with higher turnover rates, at the expense of competitive species more efficient in the use of nutrient and carbon resources. Their findings support early studies that show applying mineral nitrogen fertilisers increased the decomposition rate of organic matter residues by satisfying the N requirements of microorganisms (Jenkinson, Fox and Rayner, 1985; Birkhofer *et al.*, 2008). Furthermore, Poulton *et al.,* (2018) reported that the exclusive use of mineral fertilisers on long-term arable trials resulted in degraded carbon stocks. The higher decomposition rates observed under conventional management could, along with lower organic matter inputs and differences in ley management, help explain the significantly lower carbon stocks observed across conventional field sites.

It is important to note here that some studies investigating the differences between organic and conventional management have only reported higher decomposition of organic material after extended study durations. Martinez-Garcia *et al.*, (2021) only reported higher decomposition (higher C loss) in organic sites after two months and they suggest that this could explain why shorter studies (e.g. DiekOtter *et al.*, (2010)) found no significant differences. Domínguez *et al.*, (2014) kept their litter bags in for considerably longer in their field-based experiments (up to 12 months), in which they identified increased decomposition rates under organic agriculture. It is therefore possible that a longer incubation time, more than the 52 days used in this study, could have detected changes over time and this should be considered when applying the Tea Bag Index method (Keuskamp *et al.*, 2013) in other agricultural experiments.

5.4.2.4 Nutrient storage and filtration (the provision of drinking water)

The results show that organic field sites retained more nitrogen and leached, on average, 62% less nitrogen (19.85 kg N ha^{-1} ± 8.67) than conventional fields sites (52.64 t N ha^{-1} ± 30.67). Whilst the difference between leaching from conventional and organic fields was not significant, the findings support other literature on the reduced losses of nitrogen per unit area from organic land (Snapp, Gentry and Harwood, 2010; Tuomisto *et al.*, 2012; Benoit *et al.*, 2014; Biernat *et al.*, 2020). It is likely that the high variability in the data (Org sites, 8 – 34 kg N ha^{-1}, con sites 15 – 150 kg N ha^{-1}) and the limited replication (sites and years) reduced statistical power of the analysis. However, the leaching calculations are within the range of other studies. Benoit et al., (2014) found organic leaching rates ranged from 14 – 50 kg NO_3-N ha^{-1} , compared to 32 – 77 kg NO_3-N ha^{-1} on coventional farms across 37 fields in the Seine Basin. Rakotovololona *et al.*, (2019) report a mean nitrate leaching of 15 kg N ha^{-1} (3 – 46 kg N ha^{-1}) across 35 organic fields in Northern France and Zhou and Butterbach-bahl (2014) meta-analyses show nitrate losses ranging from 0.3 – 325 kg N ha^{-1} across conventional maize and wheat fields.

The main explanation for reduced nitrogen losses under organic agriculture is lower nitrogen fertiliser applications (both as organic and inorganic nitrogen), which is also evident at CDE (see Table 5.1) (Knudsen *et al.*, 2006; Tuomisto *et al.*, 2012). Harris *et al.*, (2006), however, also highlight the importance of good soil structure, adequate SOM, good biological functioning and cation exchange capacity (CEC) in attenuating pollution export from soil, including nitrates. Additionally, the timing of nitrogen fertiliser application is important. When taking into account total N inputs (including inorganic and organic inputs) for each field monitored for nitrate leaching, organic fields lost on average 33% and conventional fields sites 38.6% of the N inputs applied. Given that the percentage losses are not equitable between organic and conventional fields sites (i.e. a higher percentage of nitrogen applied on conventional sites), it could be that at organic field sites the higher organic matter and the application of nitrogen in combination with organic material (i.e. as FYM) are contributing to reduced nitrate leaching. It is recognised that significant losses can occur in organic systems following the cultivation out of the grass ley phase (Stopes *et al.*, 2002) but this was not observed in this study.

Whilst nitrate leaching was not significantly different between the two systems, the finding of lower nitrate leaching under organic agriculture is important as it contributes to growing evidence that the system can lower nitrate losses per unit area. Some critics raise concerns over drawing comparisons in nitrate leaching losses based on per unit area measurements (Kirchmann *et al.*, 2016). Arguing that when using a measure of N leached per unit crop produced, lower yielding organic systems typically perform less favourably than conventional systems (i.e. higher nitrate leaching per unit of crop produced) (Kirchmann *et al.*, 2016). However, based on the potential positive implications for drinking water quality and a globally sustainable rate of organic farming conversion, there is significant scope for strategic, spatially targetted implementation of organic agriculure to improve drinking water supply, with limited impact on agricultural production.

5.4.3 Indicators that inform the quantification of soil-based ecosystem services

The use of indicators of soil-based ES and soil functions is common in the assessment of the goods, services and disservices that could be derived from soil NC (see review from Greiner *et al.*, (2017)). Findings highlight that some of these indicators or measurements of soil functions can provide more information on final soil-based ES than others and it is important to consider this in future assessments of soil quality. Three key measurements were identified as being useful in providing underpinning evidence to quantify soil-based ES; carbon stocks, nitrate leaching and biomass production. All were used as a measure of soil-function in this study, highlighting the importance of functional measurement of soil condition rather than static properties that cannot directly inform an understanding of soil-based ES delivery (Baveye, Baveye and Gowdy, 2016). Whilst measurements of soil condition (such as indicators relating to fertility and decomposition rates), might be useful in providing other information such as informing fertiliser management plans, they are unlikely to advance the quantification of soil-based ES.

Measuring soil carbon is common in agricultural studies (Greiner *et al.*, 2017) and this kind of data is increasingly being collected by farmers during soil-testing. However, it is important to be explicit here about what is beneficial in understanding the soil-based ES of climate regulation. Carbon stocks (t ha^{-1}), rather than SOC (%), are required to understand how much carbon is stored in topsoil and this requires both data on percentage carbon content in the soil (SOC %) and BD (g soil cm^{-3}). Collecting BD data is more time consuming and therefore less frequently measured. However, without a measure of BD, it is impossible to determine the stock of carbon within the topsoil and the subsequent soil-based ES of climate regulation. Interestingly, the results from this study show that under organic management, whilst SOC (%) levels were higher, they were not significantly different to conventional fields, whereas carbon stocks (t ha^{-1}) were. SOC (%) alone did not provide information on the differences in carbon storage and limits the capacity to make conclusions about the climate mitigation potential of different agricultural practices. Furthermore, as discussed previously, monitoring of carbon stocks (t ha^{-1}) is ultimately needed to quantify carbon sequestration over time. Other studies have acknowledged that given a good understanding of carbon stocks it is relatively straightforward to then understand the value of the soil-based ES of climate regulation (Harris *et al.*, 2006; Keeler *et al.*, 2012; Duncan, Thompson and Pettorelli, 2015).

Similar to measurements of carbon storage, an understanding of crop biomass production can inform crop yields and from this, it is relatively straightforward to understand the value of the soil-based ES of food provision. Measuring crop yields is not always conducted in agricultural studies comparing different land management solutions, but it is critical in understanding the trade-offs or win-wins between environmental and agricultural performance.

In contrast to carbon stocks and measurements of crop yield, nitrate leaching is a valuable measure to assess environmental performance. However, it is important to note that it is not straightforward to then understand how it will subsequently impact on the provision of final ES (such as the provision of clean drinking water). Whilst higher nitrate leaching was observed under conventional management in this study (as in others: Snapp, Gentry and Harwood, 2010; Tuomisto *et al.*, 2012; Benoit *et al.*, 2014; Biernat *et al.*, 2020), this data alone is insufficient to understand whether there is a significant impact on surface or groundwater drinking water quality, something which is spatially specific (Wang *et al.*, 2011, 2016). Whilst this measure is relevant at the case-study site presented, where there is a significant drinking water aquifer with a nitrate contamination issue, it might not be as important elsewhere. This highlights the need for spatially specific targeting in the selection of indicators of soil-based ES when assessing the sustainability of different land management practices.

Through applying a suite of indicators, carbon stocks (t ha^{-1}), biomass production (t ha^{-1}) and nitrate leaching were recognised as providing important information to understand the output of soil-based ES. This highlights the importance of functional assessment of soil condition rather than static properties that cannot directly inform an understanding of soil-based ES delivery. Even with robust indicators, it is difficult to establish whether higher carbon storage but lower yield is "better" or "worse", than having lower carbon storage but higher yields. When considering trade-offs between soil-based ES it is useful to have a common measure, with advocates supporting the economic valuation of these goods and services. Valuation of soil-based ES was beyond the scope of this chapter but is explored further in Chapter 7.

5.5 Conclusion

This study provides a robust UK case-study comparing a suite of indicators of soil NC condition, soil functioning and soil-based ES under two management systems. The study found that despite significantly higher carbon stocks (t ha^{-1}), organic field sites did not have systematically improved NC condition compared to conventional field sites. There was no significant difference in soil structure and stability and despite higher mean SOM, Mg and total N, organic field sites had lower mean bioavailable-P and K. Significant differences were observed in soil function between organic and conventional field sites, with organic field sites having significantly higher carbon storage (t ha^{-1}) but significantly lower biomass production (t ha^{-1}) than neighbouring conventional field sites. No significant differences were observed in organic matter decomposition or nitrate leaching, although mean nitrate leaching at conventional field sites was notably higher than at organic field sites.

The differences in soil NC condition and function can probably be explained by differences in fertiliser regime (much higher N inputs at conventional field sites), organic matter inputs (much higher at organic field sites) and the management of the rotation. It was particularly interesting to observe that under the organic rotation and to some extent the conventional rotation, some measurements of soil fertility (bioavailable-P, total N and C:N ratio), soil carbon stocks and biomass production were highest in the grass-arable phase of the rotation[16]. Under organic management, this involved three years under a grass-clover ley and it provides compelling support for other studies that have also recognised the benefits of this management practice in improving soil conditions in both organic and conventional systems (Conant, Paustian and Elliott, 2001; Johnston *et al.*, 2017; Bliss, 2018; Berdeni *et al.*, 2021).

It evident from the research presented here that organic agriculture can enhance some aspects of soil function (e.g. reduced nitrate leaching and increased carbon storage) but systematic improvements were not observed in other aspects of soil NC condition (e.g. soil structure and stability). Given the interest in the expansion of organic agriculture, with calls for the UK Government to match the European Union's Farm to Fork strategy target of 25% of agricultural land to be under organic management by 2030 (ORC, 2021), further work is required to better understand how organic agriculture impacts the flows of soil-based ES. Specifically, this work should be applied to a variety of management situations, under different soil conditions (e.g. across different soil texture classes) and over

[16] Here soil samples were taken in autumn before the grass ley was cultivated out in spring and put to maize in conventional field sites and to wholecrop spring wheat in organic field sites

longer time frames. This research should additionally focus on harder to measure soil-based ES such as the impacts to flood regulation and water storage, due to changes in soil structure and soil erosion.

Chapter 6: Organic conversion and long-term pollinator stocks: A landscape-scale analysis using the BEE-STEWARD software

6.1 Introduction

The annual value of pollination services to UK agriculture have been estimated at £430 - £603 million (Smith *et al.*, 2011; Hanley, Ellis and Breeze, 2013) and globally, from US$195 to US$387 billion (Porto *et al.*, 2020). Pollinator populations are important NC stocks (Hanley *et al.*, 2015) and pollinating insects such as honeybees, bumblebees, solitary bees, wasps and hoverflies deliver a critical ecosystem service (ES) pollinating both commercial crops and wild plants (Klein *et al.*, 2007; Potts *et al.*, 2010; Ollerton, Winfree and Tarrant, 2011). In addition, healthy pollinator populations have been shown to boost crop yields (Garratt *et al.*, 2014; Pywell *et al.*, 2015; Knapp *et al.*, 2019). Despite recognition of their importance, pollinators, their habitats and the floral resources that support them, have experienced significant declines in the last 100 years (Carvell *et al.*, 2006; Potts *et al.*, 2010; Baude *et al.*, 2016). One of the main drivers of pollination declines has been agricultural intensification through the 20th century (Ollerton *et al.*, 2014), resulting in the simplification of landscapes, the loss of semi-natural habitats and the increased use of agri-chemicals (Potts *et al.*, 2010). Whilst declines in pollinators are thought to have slowed in recent years (Carvalheiro *et al.*, 2013), there is still urgent need for agricultural solutions that restore pollinator stocks whilst (ideally) balancing the delivery of other ES.

A number of different practices can potentially improve pollinator abundance and diversity on farmland. These include targeted, within-field habitat interventions (Pywell *et al.*, 2006; Carvell *et al.*, 2007; Pywell *et al.*, 2015; Dicks *et al.*, 2015; Wood *et al.*, 2015) and systematic changes: such as shifting to organic agriculture (Kennedy *et al.*, 2013; Hardman *et al.*, 2016a; Geppert *et al.*, 2020). The capacity for these practices to restore pollinator stocks whilst balancing the delivery of other ES is, however, not fully understood. Contributing to this understanding is the focus for this study, which aims to investigate the capacity for these practices to enhance bumblebee populations on a lowland estate in South West England using the recently published BEE-STEWARD software (Twiston-Davies, Becher and Osborne, 2021). The study considers the consequences of organic conversion and within-field habitat interventions on provisioning ES, quantifying the impact on productive output and producer surplus (a measure of farm profitability). The study finishes by investigating the cost-effectiveness of the different practices in making bumblebee population improvements. An introduction to the themes of the study is provided below before outlining the detailed objectives.

6.1.1 Organic agriculture: a solution to address pollinator declines and balance productive output?

Organic agriculture is the most widely acknowledged alternative to conventional farming (Ponisio *et al.*, 2014). Whilst the distinctions can be broad, the key difference is that organic agriculture does not use synthetic fertilisers, herbicides or pesticides (Ponisio *et al.*, 2014). A number of meta-analyses have identified the environmental and ecological benefits of organic agriculture (Tuomisto *et al.*, 2012; Tuck *et al.*, 2014; Stein-Bachinger *et al.*, 2021) and recent studies have found that it has the capacity to enhance resources for pollinators (Hardman *et al.*, 2016a), increase pollinator abundance and diversity (Holzschuh, Steffan-dewenter and Tscharntke, 2008; Kennedy *et al.*, 2013), improve bumblebee colony development (Geppert *et al.*, 2020) and increase pollination services (Hardman *et al.*, 2016a). The beneficial effect of organic management on pollinators has been found to vary depending on landscape structure and heterogeneity (Holzschuh *et al.*, 2007; Rundlöf, Nilsson and Smith, 2008) and despite growing evidence, the impacts on biodiversity are still subject to significant debate (Tuck *et al.*, 2014). Tuck *et al.,* (2014) argue that there is still a need to understand the precise benefits delivered by organic farming. There is also continuing debate over the role organic agriculture might play in the transition to a more sustainable economic system particularly in light of the widely accepted belief that organic agriculture has lower yields (De Ponti, Rijk and Van Ittersum, 2012; Ponisio *et al.*, 2014; Röös *et al.*, 2018; Smith *et al.*, 2019). Lower yields translate into more land needed to produce food, raising concerns that organic farming may precipitate higher environmental impacts per unit produced (Connor and Mínguez, 2012). On the other hand, others advocate its expansion, highlighting the positive delivery of multiple environmental benefits and higher profitability for producers (Crowder and Reganold, 2015; Wilbois and Schmidt, 2019). There is a clear need to better understand the scales of potential biodiversity benefits, trade-offs in yield and profitability for producers.

6.1.2 Habitat interventions on farmland: a targeted method to enhance pollinator populations?

Pollinator-friendly interventions, including those that create nesting areas (tussocky grass margins) and foraging resources (flower rich margins and plots), are considered a reliable way to improve habitats on farmland (Dicks *et al.*, 2015). Forage resource availability has been closely aligned with the fate of pollinators (Baude *et al.*, 2016), with increasing resources offering the potential to enhance a wide range of species (Sutter *et al.*, 2017) and improve pollination services (Hardman *et al.*, 2016a). Studies have found that sowing flower-rich habitats (particularly those targeted at providing pollen and nectar) can increase bumblebee abundance and diversity (Pywell *et al.*, 2005; Carvell *et al.*, 2007; Holland *et al.*, 2015). More recently, studies have identified improved bumblebee colony development next to sown flower rich margins (Geppert *et al.*, 2020) and increased bumblebee population sizes on farms with targeted habitat interventions (Wood *et al.*, 2015). As with organic agriculture, pollinator response to habitat interventions has been shown to be strongly influenced by landscape heterogeneity (Carvell *et al.*, 2011; Geppert *et al.*, 2020). Wood et al., (2015) provide one of the few examples that have considered population-level change as a response to habitat interventions. They highlight that a problem with earlier studies has been that it is unclear whether increased bumblebee abundance in and around sown flower habitats is simply due to attracting workers to that location or corresponds with a genuine population increase. Further work is required to understand the impact of interventions on population-level change and advances in bee models (Becher *et al.*, 2014, 2016, 2018) present a resource-efficient means to do this.

6.1.3 The cost-effectiveness of agri-environment interventions

Both organic farming and habitat interventions are supported through government payments in the UK and European Union through agri-environment schemes (AES). Introduced in Europe in the late 1980s, AES has the aim of tackling the environmental impacts of industrial agricultural practice (Batáry *et al.*, 2015). Conversion to organic farming and annual support payments are distributed in the UK and across Europe (Latacz-Lohmann and Renwick, 2002) and the European Union have recently set a target to have 25% of the utilisable agricultural area under organic management by 2030 (European Commission, 2021). Recent developments of the UK AES, the Countryside Stewardship Scheme (CSS), has resulted in the creation of the Farm Wildlife and Pollinator Package (FWPP), specifically targeted at creating a minimum baseline habitat to support pollinators on farmland (Dicks *et al.*, 2015). Despite studies comparing the efficacy of the two different solutions in delivering benefits for pollinators (e.g. Hardman *et al.*, 2016; Geppert *et al.*, 2020) and some hinting at the possible trade-offs in yield (Hardman *et al.*, 2016a) few have quantified yield trade-offs or synergistic benefits for other ES. Furthermore, studies typically do not include an assessment of the cost-effectiveness of the different AES strategies, i.e. what scale of pollinator response is seen for the investment. This could be relevant to the costs or benefits to the farmer (if undertaking practices voluntarily) or to society (if payment is made through AES streams). Ansell *et al.*, (2016), in their review of the literature on cost-effectiveness of AES, report that very few studies have considered economic as well as ecological data in the evaluation of different strategies. They call for the inclusion of cost-effectiveness assessments in future evaluation studies to aid conservation strategy.

6.1.4 Study objectives

There is a clear need to develop understanding on the role of organic agriculture and habitat interventions in sustaining healthy pollinator populations in different landscapes. Given the scarcity of conservation funding, it is important to understand the cost-effectiveness of these AES approaches and the financial implications for land managers. Such information will be particularly important as the government exits the European Union and develops its new Environmental Land Management Scheme. The following work contributes to these knowledge gaps by building understanding of forage resource availability, bumblebee population response and the trade-offs and benefits to farmers under different realistic landscape scenarios on a SW England farming estate. A large part of the study area was converted to organic agriculture in 2007 and this study considers the response to this change, collecting ecological and economic data from neighbouring conventional fields as a baseline (before 2007) and organic fields that represent current conditions. Pollen and nectar rich plots and margins, tussocky grass margins and un-cut clover strips were added into the landscape between 2018 – 2021 and now cover an area in line with the FWPP target (>1ha pollen and nectar habitat per 100ha mixed or arable farm land). The study incorporates these changes into a second scenario, considering the role that habitat interventions play in generating changes in a baseline conventional landscape or in the new majority organic landscape. The cost-effectiveness of each approach in delivering benefits to bumblebee populations is considered.

The study focuses on bumblebee populations due to their importance as a keystone species (Goulson *et al.*, 2011) and the recent publication of the user-friendly BEE-STEWARD software (Twiston-Davies, Becher and Osborne, 2021) which enables the modelling of bumblebee population dynamics over time. The BEE-STEWARD software brings together the bumblebee model Bumble-BEEHAVE (Becher et al., 2018) and the bee foraging model BEESCOUT (Becher et al., 2016). It is not the only method to model bumblebee populations (see Becher *et al.*, (2018) for a review of existing models) but has the advantage of taking into account multiple stressors within realistic landscapes. The software has been used in other studies considering bumblebee population response to different agricultural practices (Knapp *et al.*, 2019) and two earlier models focusing on honeybees, the BEEHAVE and BEE-SCOUT models (Becher *et al.*, 2014, 2016) have been used to investigate honeybee response to farming changes (Horn *et al.*, 2016, 2021).

The study had three key objectives:

1. To apply the BEE-STEWARD model to answer the following questions:

    A. Does a landscape-scale shift from conventional to organic agriculture enhance floral resources available to insect pollinators?

    B. Does the addition of pollen and nectar habitat interventions enhance floral resources within conventional and organic dominated landscapes?

    C. Does a landscape-scale shift from conventional to organic agriculture enhance long-term bumblebee populations?

    D. Do pollen and nectar habitat interventions enhance long-term bumblebee populations at a landscape scale?

2. To quantify the trade-offs in yield and the returns to farming associated with:

    A. A landscape scale shift from conventional to organic agriculture

B. The addition of pollen and nectar habitat interventions in the farmed landscape

3. To estimate the cost-effectiveness of different land management changes in enhancing pollinator stocks, tackling the question: What is the most cost-effective strategy to increase bumblebee populations?

6.2 Methods

6.2.1 Study site

The study was conducted on Clinton Devon Estate (CDE) in South West England (Figure 6.1) (a full description can be found in Chapter 3, Section 3.1). A large holding on the estate, the Home Farm (ca. 900ha), is farmed in-hand and was converted from conventional arable and dairy to organic arable and dairy in 2007. The agriculture before conversion was typical of neighbouring conventional farms, with winter cereals, maize, rye grass/clover silage leys and improved pasture.

The modelled landscape (3km x 3km model grid; Figure 6.1) centres around the Home Farm. Figure 6.1 shows the land use in 2019, habitat interventions on the farm and locations of survey transects. The 3km x 3km black square shows the extent of the model landscape used in floral resource, economic trade-off calculations and bumblebee modelling. The current organic extent is marked in hatch and whilst it covers most of the cropped area, some conventional fields remain in the study landscape.

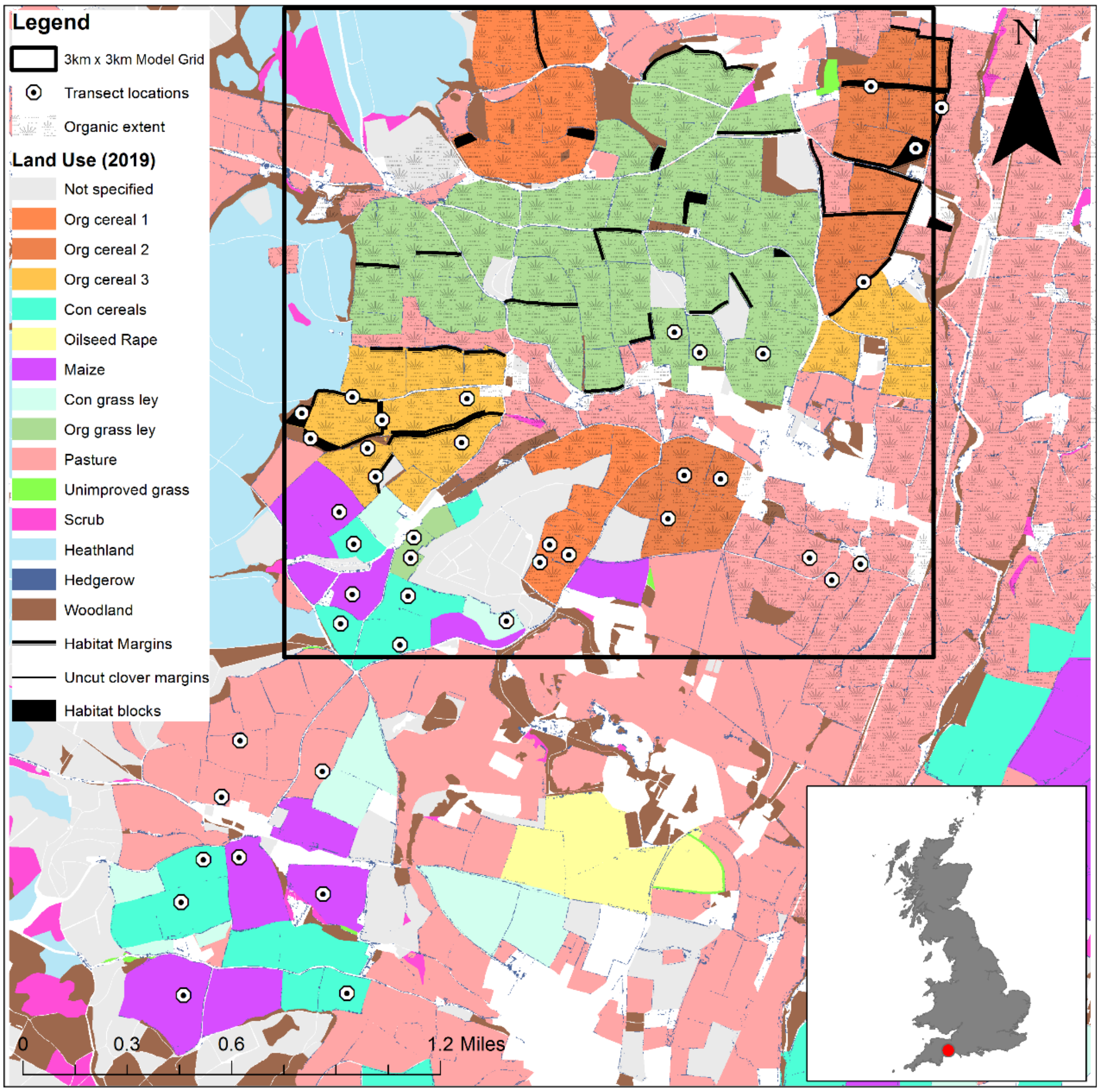


Figure 6.1: Map of the study area showing the land use in 2019, the location of habitat interventions on the Home Farm (e.g. habitat blocks and margins), flower count transect sites and the area of the 3km x 3km landscape used in floral resource and economic trade-off calculations and in bumblebee modelling.

6.2.2 Data inputs for the BEE-STEWARD software

Landscape-scale floral resources (pollen and nectar availability) and bumblebee colony development were modelled using the BEE-STEWARD software (Twiston-Davies, Becher and Osborne, 2021). BEE-STEWARD has a user-friendly interface and brings together the bumblebee model Bumble-BEEHAVE (Becher *et al.*, 2018) and the bee foraging model BEESCOUT (Becher *et al.*, 2016). Bumble-BEEHAVE is an agent-based model that simulates colony growth and survival in landscapes where pollen and nectar resources can be calculated from habitat maps. The model accounts for multiple stressors and competition that take place in realistic landscapes, modelling nest establishment and colony development, brood needs, bee foraging and reproduction (Becher *et al.*, 2018). For the purposes of this study, it provides valuable information on the number of different individual bumblebees and bumblebee colonies (population stocks) that a landscape can support over time. A full description of the model can be found in Supplementary Material S03 (“ODD protocol”) of (Becher *et al.*, 2018).

The BEE-STEWARD software has three main inputs that can be tailored to a study landscape:

1. a landscape map(s) to classify the crop and non-crop habitats used in the model (these can be for a single year or include a suite of maps for each year of a farm rotation);
2. a habitat input file which defines the density of each flowering species in each crop or non-crop habitat; and
3. a flower species input file which defines the daily pollen and nectar resources available for each flowering species along with details of flower phenology and corolla depth[17]

Generating the inputs for BEE-STEWARD therefore required the following data:

- **Spatial data**: information on the spatial location and extent of crop and non-crop habitat for each year of the rotation;
- **Flower density data**: number of flowering units for each insect-rewarding flowering species per m^2 for each land use type mapped in the study landscape; and
- **Flowering plant data**: daily quantity of nectar (ml) and pollen (g), quality of nectar sugar (mol/l) and pollen (percentage protein), corolla length and period of flowering (start and stop dates) for each flowering species included in the model

The methods used to collect the required data are outlined below.

6.2.2.1 Spatial data – creating habitat maps

All mapping was conducted in ArcGIS 10.5.1 (ESRI, Redlands, CA, USA). Polygon data from the OS Vector Local dataset was used to classify non-crop habitats: woodland, scrub, heathland and unimproved grassland (Ordnance Survey, 2020). An automated LiDAR filtering protocol was used to extract woody

[17] Corolla depth plays an important role in determining whether nectar resources can be accessed by bumblebees of different tongue lengths (Rollin *et al.*, 2016).

landscape feature geometries (i.e. hedgerows, trees and woodland) from basic gridded LiDAR data products following the method in Luscombe *et al.,* (2022)[18]. Non-crop habitats remained unchanged for all model scenarios and for each year of the rotation.

Cropping and grazing land were mapped using farm management records provided by the Home Farm and neighbouring tenant farmers for 2016 – 2021. Data from these 6 years was used to create six crop maps, one for each year, which were used in the model to represent the years of the landscape rotation. Where cropping data was not available from farm records, polygon data from the CEH Land Cover plus Crops map dataset (CEH, 2020) was used to define field crops.

Recent habitat interventions voluntarily installed on the Home Farm were digitised from a paper map provided by the farm manager. All un-defined habitats (e.g. urban areas and pine woodland) were considered devoid of resources. Heathland habitat was omitted from the habitat maps as local data was unavailable for flowering densities and no existing data for this habitat type are present in the BEE-STEWARD software.

Using the actual farm management data and non-crop habitats, four landscape scenarios were mapped. Each of which had a set of six maps, one for each year of the rotation. The four cropping scenarios were:

- Con_Base: representing a 'business as usual' scenario, which would have persisted had the estate not converted the Home Farm to organic agriculture in 2007. The pasture area and grass ley areas remain the same as for the actual organic maps but the cropping of the Home Farm area includes more maize (as forage for the dairy herd) and less cereal production. Cropping areas were generated following a discussion with the Home Farm manager as to the likely crop plan, were the farm still under conventional management.
- Org_Base: the actual rotation as it was following conversion in 2007. It does not include the more recent voluntary habitat interventions.

[18] This work was kindly conducted using a new model (currently awaiting publication) developed by Dr David Luscombe at the University of Exeter.

- Con_Hab: combines the Con_Base landscape with the habitat interventions that have been adopted on the Home Farm.
- Org_Hab: represents the current crop rotation and includes the voluntary habitat interventions that have been installed on the Home Farm over the past few years.

All features were mapped as polygons before being converted to raster files (15m x 15m grid cells) and then to ASCII Text files. ASCII text maps, with cell codes specific to each crop and non-crop habitat, were then fed directly into the BEE-STEWARD model to generate food source maps.

6.2.2.2 Flower density data

Flower density data (flowering units per m^2) are available for a suite of farmland non-crop (e.g. hedgerows) and crop habitats (e.g. oilseed rape) in the original Bumble-BEEHAVE model (Becher *et al.*, 2018). The existing flower density data for unimproved grassland, woodland, hedgerow and scrub habitats in the farm landscape were used. As the study was testing the role that organic cropping and habitat interventions play in pollinator population dynamics, empirical field data was collected on flowering density for all frequently grown conventional and organic crops and habitat interventions in the study landscape. Empirical data collection methods are outlined below.

Field and transect selection:

Field selection followed the same method described in Chapter 4. Crop plans and any habitat interventions were first discussed with the study farmers and mapped in GIS. The main crop in 2019, along with the proceeding and following crop, were recorded. Details of the previous and following crop were noted, as they were considered to impact on arable weed/flower densities. Frequently grown crop types were then short-listed (see Table 6.1). Grouping analysis using ArcGIS 10.5.1 (ESRI, Redlands, CA, USA) was conducted on fields of the desired cropping so that, wherever possible, fields shared a similar slope and aspect. Three fields for each crop type were selected (Figure 6.1). Habitat interventions were only applied on the organic farm and included grass margins, pollen and nectar margins, three pollen and nectar plots and un-cut strips on grass-clover leys. Three grass and three pollen and nectar margins were selected for transects and were mapped as polylines. The three pollen and nectar plots were mapped as polygons[19]. The Create Random Points tool was used in ArcGIS (ESRI, Redlands, CA, USA) to select a random point in each field and habitat plot and a 25m line mapped due east and west of each centre point creating a 50m transect. A random point was also assigned to each habitat margin and 25m was measured in each direction, following the orientation of the margin, to establish a 50m transect.

[19] Following the selection process two of the three pollen and nectar habitat blocks were compromised; one was re-sown in May 2019 with a different flower mix and the other re-sown in May 2019 as a wild bird cover mix. This was not communicated prior to the changes being made and as no other alternative pollen and nectar plots were established at the time the monitoring continued with the existing transect selection.

Table 6.1: Summary of main crops identified in the study area. For each crop/land use type, three replicate fields were selected and a 50m x 2m transect was randomly assigned. The main crop along with the previous winter and following winter crop/land use are shown

System	Main crop/land use type	Previous crop	Following crop	No. trans
Organic	Grass ley: Rye-grass white clover	Same	Same	3
	Cereal 1: Spring wheat	Grass ley	Overwinter stubbles	3
	Cereal 2: Spring triticale (into grass)	Stubbles	Stubble turnips	3
	Cereal 3: Spring triticale (into cover)	Stubble turnips	Grass ley	3
	Pasture: Rye-grass clover pasture	same	same	3
Conventional	Grass ley: rye-grass/some clover	Same	Same	3
	Maize 1	Stubble turnips	Winter wheat	3
	Maize 2	Grass ley	Winter wheat	3
	Cereal 1: Winter wheat/barley	Maize	Stubble turnips/mustard	3
	Cereal 2: Winter wheat	Maize	Grass ley	3
	Pasture: Rye-grass clover pasture	Same	Same	3
Habitat	Flower margin (pollen and nectar)	NA	NA	3
	Grass margin	NA	NA	3
	Ley margin: grass ley left uncut	NA	NA	-
	Flower plot: Pollen and nectar mix*	NA	NA	2
	Flower plot (wild bird cover)*	NA	NA	1
Total transects				42

*Changes in land management altered the species sown in the habitat blocks. Further details in Methods.

- Transect data was used from flower densities for the existing conventional and organic grass ley crops

Flower count surveys:

Survey transects were visited monthly, over five visits between the end of May/early June and the end of September/early October. Canes were installed at each transect end and all open flowering units were counted along the 50m x 2m transect. Flowering units were defined and counted following the same method in Baude *et al.*, (2016), with a flowering unit representing one or a number of flowers that can be visited by an insect without taking flight. Full details on how flowers from each species were counted can be found in Appendix D.1. Flower density (flower units per m^2) was calculated by dividing total flower count data by the total area of the survey transect (100m). Bumblebee species were noted during the flower count surveys to ensure that the bee species used in the model were present within the study landscape.

Calculating crop and habitat intervention flower densities:

The mean flower density for each plant species was calculated from the replicate transects for each survey month: May – September. Where the data revealed little difference in the flower counts of similar crop types, the flower count data were merged to simplify the number of different crop types in the model. In particular, Maize 1 was merged with Maize 2, conventional Cereal 1 with Cereal 2 and organic Pasture with conventional Pasture. The mean flower densities for those crop types were estimated from the merged data. For crops/land uses where flower density appeared to be unaffected by management, the average annual density of each flowering species was then calculated from the mean of the five survey visits. This method applied to conventional cereals, maize, pasture, organic cereal 3 (which was harvested but left as stubbles) and all habitat margins.

For four crop types there was clearly an obvious impact of management: organic clover silage leys, conventional grass leys, organic cereal 2 and organic cereal 3. For these crops, a 'mowing' function was developed in the BEE-STEWARD software to account for the impact of grass cutting or cereal harvesting on floral resources. The mowing function allows the specification of the date of cutting, the date of recovery and the relative abundance of flowering plants in the interim. Details on the calculation of mean flower density for these habitats in preparation for the 'mowing' function is explained in more detail in Appendix D.2.

Two of the three pollen and nectar habitat blocks were compromised by a change in management in May 2019. The uncompromised transect was used to calculate the average annual density of flower units for pollen and nectar plot habitats in BEE-STEWARD. Of the two compromised transect locations, one was dropped from the calculations for flower densities for pollen and nectar plot habitat. The other was re-categorised and included in BEE-STEWARD to calculate the flower densities for a different habitat flower plot type 'game cover mix'.

6.2.2.3 Flowering plant data

Pollen and nectar data for the insect-rewarding plants recorded across the survey transects was obtained from the literature. Flowering species that were recorded in very low densities were not included. Data for 35 flowering plants and four crops that represent important floral resources for bumblebees are already included within the Bumble-BEEHAVE model (Becher *et al.*, 2018). Pollen and nectar data were collated for a further 62 plant species, primarily using data from Hicks *et al.*, (2016) and Baude *et al.*, (2016)[20]. Where data were not available within these datasets, a wider range of literature was consulted and if still unavailable, pollen and nectar data were estimated using the most similar plant or group of plants. Flower phenology data were collected from Baude *et al.*, (2016) and where necessary, dates were updated to reflect field observations made in this study. Flowering plants were classified as being: 1) Important for bumblebees and/or 2) important for other pollinating insects. All flowering plants classified as being important for pollinating insects were included in calculations for total floral resource provision of pollen and nectar. Only plants classified as being important for bumblebees were retained for the BEE-STEWARD runs to determine population dynamics. A full list of flowering plants recorded in the study are included in Appendix D.1, including literature sources for pollen, nectar and any assumptions made during data collection. The final flower species lists used for bumblebee simulations and to determine landscape pollen and nectar are presented in Appendix D.3 and D.4 respectively.

6.2.3 Model simulations using the BEE-STEWARD software

Two models were generated using BEE-STEWARD, one to calculate pollen and nectar resource availability and one to run bumblebee population dynamics for each landscape scenario.

Pollen and nectar resource calculations were conducted by running the model with the mowing protocol switched on for six years with **all** insect-rewarding plants. Annual pollen (kg year^{-1}) and nectar sugar (kg year^{-1}) for each year of the rotation was then calculated by summing daily pollen and nectar calculations

[20] Data from Hicks *et al.,* (2016) and Baude *et al.,* (2016) are presented in different units (nectar sugar; µg day^{-1} and pollen; µL day^{-1}) to the format used in Bee-Steward (nectar; ml day^{-1} and pollen; g day^{-1}). They therefore required conversion to meet BEE-STEWARD requirements. Further details in Appendix D.5.

generated in the model. Pollen was converted from grams to kilograms and nectar was converted from litres to kilograms of sugar using the following equation:

$Nectar\ sugar =$ NectarLitres * 0.342.

where sucrose concentration of the nectar was assumed to be 1 mol/l, with a molar weight of 0.342 kg sucrose per mol.

BEE-STEWARD simulations were conducted separately for two common species of bumblebee: *Bombus terrestris* and *Bombus hortorum. B. terrestris* was chosen as a subject for this analysis as it is ubiquitous in the UK and has the most robust documentation on bee and colony behaviour and development for modelling. *B. hortorum* has a longer tongue length and slightly later emergence time in the model than *B. terrestris* and was selected to examine whether these features impact on the suitability of each landscape in supporting a separate bee species with a different ecology. Both species use the same nesting habitats, and typical of all bumblebees, require a continuous supply of pollen and nectar through the spring and summer months (Pywell *et al.*, 2005). Nectar is the critical energy source for adult bees, whereas pollen is used as a source of protein for larvae, freshly emerged workers and the queen (Konzmann and Lunau, 2014). Both species are known to be present in the case-study landscape and this was confirmed during the 2019 field work season.

The number of starting queens for both species was set at 400, based on an assessment of the carrying capacity of the study landscape and on the time it took for populations to stabilise (simulation tests reported in Appendix D.6). Tailoring the number of starting queens to the study landscape reduces the time it takes for the population to stabilise, reducing computational time and improving confidence in the likelihood that populations have reached equilibrium during the analysis period. Following these test simulations, it was decided that all models (i.e. for each landscape scenario and for each bee species) would be run for 24 years, allowing 12 years for the population to stabilise (two full cycles of the rotation) and then analysing the data thereafter (from 12 – 24 years). Each model was run 30 different times (i.e. 30 set seeds). Specified outputs included the number of adult bees, hibernating queens and nesting colonies, nest density and time to extinction. The number of nesting colonies is important as it represents a true measure of effective population size in bumblebees, each nest representing a breeding pair (Knight *et al.*, 2005).

6.2.4 Measuring yield and economic impacts

Data were collected on crop biomass, converted into saleable yield and a crop producer surplus (defined in Section 2.4.2) calculated for all rotational crops, including conventional cereal, grass and maize and organic cereals and silage grass. Data on pasture productivity were not collected and were omitted from the analysis and reporting of biomass yield (tonnes year^{1}) and producer surplus (£ year^{-1}) from each landscape scenario. It was beyond the resources of this study to include outputs from pasture (e.g. grazed forage/livestock/milk) and this will inevitably have resulted in underestimating total producer surplus across the study landscape under both organic and conventional scenarios. Further work would be required in the future to estimate productive outputs from pasture and improve calculations of total producer surplus. In the meantime, the assumption is made that the same pattern, when comparing between organic and conventional outputs, would be observed for pasture and the rotational crops measured here.

The methods for collecting biomass data and calculating producer surplus for each landscape are explained below.

6.2.4.1 Crop biomass measurements

The collection of data on crop yields is described in detail in Chapter 3. The method is briefly summarised below.

Yield sites

Forage biomass was collected from conventional cereal, grass and maize and organic cereals and silage leys in 2019 immediately ahead of harvest (or as close as possible). The same fields used for floral resource transects were used to calculate crop yields, with the exception of organic grass silage (where additional yield data were also collected from six fields). A summary of the data collected for each habitat is provided in Table 6.2. As conventional grass ley management altered during the season (the farmer managed the fields for haylage, not silage), industry average yield for conventional silage was used in landscape scale biomass calculations.

Table 6.2: Field types from which yield data was collected for forage biomass and producer surplus calculations

System	Main crop/land use type	No. yield fields	No. yield cut plots
Organic	Grass ley: Rye-grass white clover	9	54
	Cereal 1: Spring wheat	3	17
	Cereal 2: Spring triticale (into grass)	3	20
	Cereal 3: Spring triticale (into cover)	3	24
Conventional	Grass ley: rye-grass/some clover	-	-
	Maize 1	3	17
	Maize 2	3	36
	Cereal 1: Winter wheat/barley	2	14
	Cereal 2: Winter wheat	3	25

- Conventional grass ley fields were used alternatively in 2019 (for hay) and therefore industry standard data was used instead for crop yield figures

Most of the rotational crops used in the study landscape are used as forage for dairy animals, with the exception of some small areas of conventional combinable cereals. Yield data were therefore collected and reported as if it was being harvested as forage biomass (DM tonnes yr^{-1}).

Field sampling

Yield samples were collected from pre-determined points on an 84m x 84m square sampling grid, located in the field using a hand-held GPS (Nomad Trimble, Sunnyvale, CA, USA).

Samples were collected and processed following the methods in Chapter 3, Section 3.6.4.

Calculating forage biomass at a landscape scale

Forage biomass yield data were combined and simplified for maize, conventional cereals and organic cereals. Minimum, maximum and mean biomass yields (DM t ha^{-1}) were then calculated from field data for conventional maize and cereal crops and organic silage and cereal crops. Conventional silage productivity ranges were taken from the literature (AHDB, 2021). The area of each crop for each year of the rotation in each landscape scenario was calculated in ArcGIS. Crop areas (ha) were multiplied by minimum, maximum, mean and industry average biomass yields (t DM ha^{-1}) to give the total annual biomass output (DM tonnes) for each landscape scenario.

6.2.4.2 Producer surplus calculations

Producer surplus is the measure that economists use in cost-benefit analysis to summarise the benefits of productive activity in monetary terms. It is calculated as the total revenue made through the sale of goods minus the total variable costs of production. Whilst gross margins for crops are often reported in the industry handbooks (e.g. Nix, 2018; ORC, 2017) these measurements differ from producer surplus in so much as they do not include the full field costs of producing the crops: for example, land preparation, crop management and harvest costs. The measure of producer surplus used here is essentially a measure of operating profit, that is to say the revenues generated by the sale of the crop minus all field-based management costs incurred in the crops production. It combines saleable yield data, crop market values and production cost information. The methods for data collection for each are explained below.

Calculating saleable yield and crop revenues

Fresh weight biomass yields from field data were used to calculate organic silage and conventional maize saleable crop yields. As cereal crops are typically sold as grain and straw, a conversion factor was applied to cereal DM forage biomass to calculate these values. The following equation was used:

$$Grain\ yield = \frac{(BY - W)}{GCF}$$

where *BY* is the dry matter biomass yield (t ha^{-1}), *W* is the estimated dry matter weed content crop (estimated at 1.28 t ha^{-1} based on Bulson *et al.,* 1996 for organic crops and as 0 t ha^{-1} for conventional crops) and *GCF* is the grain conversion factor. A grain conversion factor of 2.31 was used for organic spring cereals, 2.51 for conventional wheat and 2.27 for conventional barley. All conversion factors were based on the mean ratio of grain to straw from a range of different agricultural studies presented in Scarlat, Martinov and Dallemand (2010). Straw yield was calculated based on the remaining DM biomass.

Total crop revenue (£ ha^{-1}) was calculated by multiplying the saleable yield (t ha^{-1}) by the projected market value using the John Nix Pocketbook 2019 edition (Nix, 2018) for conventional crops and The Organic Farm Management Handbook

(ORC, 2017) for organic crops. Conventional and organic grass silage prices were taken from online marketplace sources and data collected from local agricultural auctioneers.

Calculating cost of crop production

Farm management data for the establishment and harvest of the 2019 crop was used to estimate a crop Cost of Production (CoP, £ ha^{-1}). Costs for each management activity were either provided by the farm or they were estimated, based on local contractor costs or data in the Nix Farm Pocket Handbook (2018). All fertilizer and spray costs were estimated using Nix (2018). Further details can be found in Chapter 3, Section 3.6.5.

Calculating producer surplus at a landscape scale

Producer surplus was first calculated per ha (£ ha^{-1}) for each crop in each field by subtracting the total cost of crop production from the total crop revenue. The minimum, maximum and mean producer surplus (£ ha^{-1}) was then calculated for each crop across all study fields. Producer surplus (£ ha^{-1}) was also calculated using average yields from industry handbooks (Nix, 2018; ORC, 2017). Producer surplus (£ ha^{-1}) was multiplied by the area of each crop type, in each rotation year for each landscape scenario to give the total producer surplus (£ yr^{-1}).

6.2.5 Data analysis

The effect of land management type (organic, conventional and habitat) on flower abundance data and the effect of landscape scenario (Con_Base, Org_Base, Con_Hab and Org_Hab) on bumblebee colony abundance was tested using generalized linear models (GLMs). Land management type or landscape scenario were used as fixed effects. All models were fitted with a negative binomial error structure. Zero-inflated negative binomial models were used for flower count data and *B.terrestris* colony number data, following guidance in Blasco-Moreno *et al.*, (2019). All GLMs were run using the glmmTMB package (Brooks et al., (2017) in R (R Core Team, 2020). Model residuals were reviewed and models were checked for zero inflation and over dispersion to confirm appropriate error structure using the DHARMa Package (Hartig, 2021). *Post-hoc* analysis of land management type impacts on flower abundance were conducted in the emmeans package (Lenth, 2020). Significance was tested at $p<0.05$.

6.3 Results

6.3.1 Floral resources under different land uses

6.3.1.1 Flower count and density data

Significant differences were detected in the number of insect-rewarding flowering units between organic field sites, conventional field sites and habitat interventions. Flower numbers were significantly higher on organic farmland than on neighbouring conventional farmland (pairwise post-hoc analysis; p = <0.0001). Habitat interventions also had significantly higher flowering units than on neighbouring conventional farmland (pairwise post-hoc analysis; p = 0.002). No significant difference, however, was detected between the number of flowering units on organic farmland and habitat interventions (i.e. flower plots, margins and grass margins) (pairwise post-hoc analysis; p = 0.77).

Figure 6.2 provides a summary of the data for each surveyed crop and habitat intervention type. Insect-rewarding flowers were often absent or in very low densities on conventional maize (mean ± SD, 0.40 ± 0.68 flower units m^{-2}) and cereal cropland (0.05 ± 0.28 flower units m^{-2}). On neighbouring organic cereal crops there were higher densities of flowering plants (mean across three organic cereal types, 61.89 ± 110.20 flower units m^{-2}), though there was large variability between some organic field sites and during different times of the year. Flowering densities were typically low in conventional and organic pasture but higher in conventional ley and higher again in organic ley grassland, where they were dominated by flowering *Trifolium repens* (white clover). Flowering density was variable between habitat interventions, with low flower counts on grass margins (2.44 ± 3.04 flowers m^{-2}) and high counts in flower plots (73.19 flower units m^{-2}).

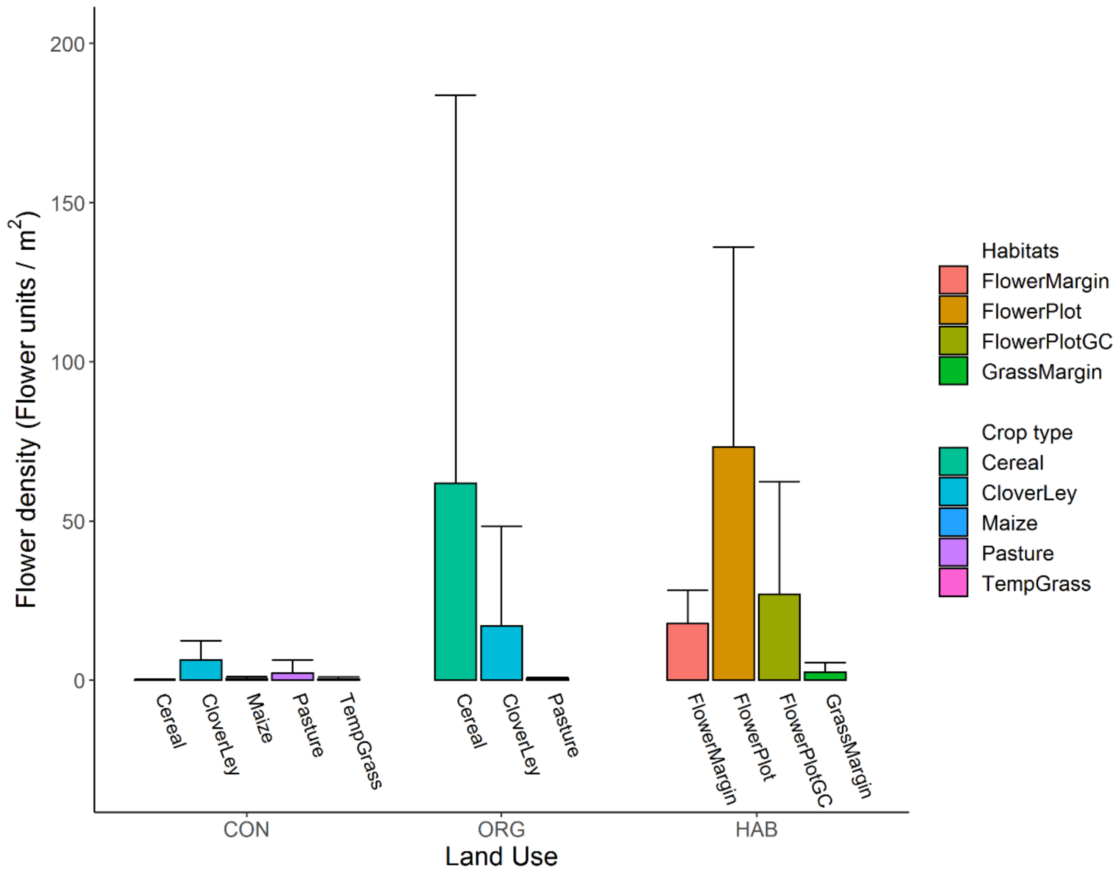


Figure 6.2: Flower density (flower units per m^2) bar plot for each of the surveyed crop and habitat types. Error bars show standard deviation. Conventional cropped land (CON) exhibits lower flower densities in nearly all comparable crop types to organic farmland (cropped land and grass leys). Marginally higher densities were observed in conventional pasture. Habitat types (HAB) show variable flower densities, significantly higher than those observed in conventional farmland but not on organic farmland (ORG) where flowering densities, particularly in cereal crops, were high.

6.3.1.2 BEE-STEWARD floral resource outputs

The outputs from annual pollen and nectar resource calculations made in the BEE-STEWARD software are shown in Figure 6.3. Figure 6.3 shows the mean annual pollen and nectar resources provided by all insect-rewarding plants across the 6-year rotation for conventional and organic scenarios with habitat interventions (Con_Hab and Org_Hab) and without (Con_Base and Org_Base).

The BEE-STEWARD software reports that all landscape scenarios are limited by nectar resources, suggesting that bumblebee development is more likely to be limited by nectar than pollen availability. Focusing on that resource, therefore, the model suggests that nectar resources (kg sugar yr^{-1}) are considerably higher across organic scenarios when compared to conventional scenarios, showing a 623% increase from the conventional scenario without interventions (Con_base) to the organic scenario without interventions (Org_base) and a 359% increase between the conventional scenarios with interventions (Con_Hab) to the organic scenario with interventions (Org_Hab). In contrast, it is evident that pollen resources are greater in the conventional landscapes. The higher total annual pollen availability can be explained by larger areas of cropped maize, accounting for 96% of the total pollen in the conventional scenario. Maize plants in the model generate a large amount of pollen but this is only available for a limited amount of time in mid to late July. Such a limited window of supply is unlikely to be particularly useful in sustaining bumblebee populations.

Figure 6.3 shows that habitat interventions increased pollen and nectar resources in both organic and conventional landscapes. In terms of proportional increase, those habitat interventions were more effective at increasing nectar availability on conventional farmland, with marginal increases observed on organic farmland. Nectar resources showed an increase of 59% between the conventional baseline to conventional habitat scenarios (Con_Base to Con_Hab), compared to only a 7.05% increase between the organic baseline and organic habitat scenarios (Org_Base and Org_Hab). Despite the addition of habitat interventions more than doubling the available nectar sugar in the conventional landscape, there is still a large gap in nectar resource availability when compared to even the baseline organic scenario.

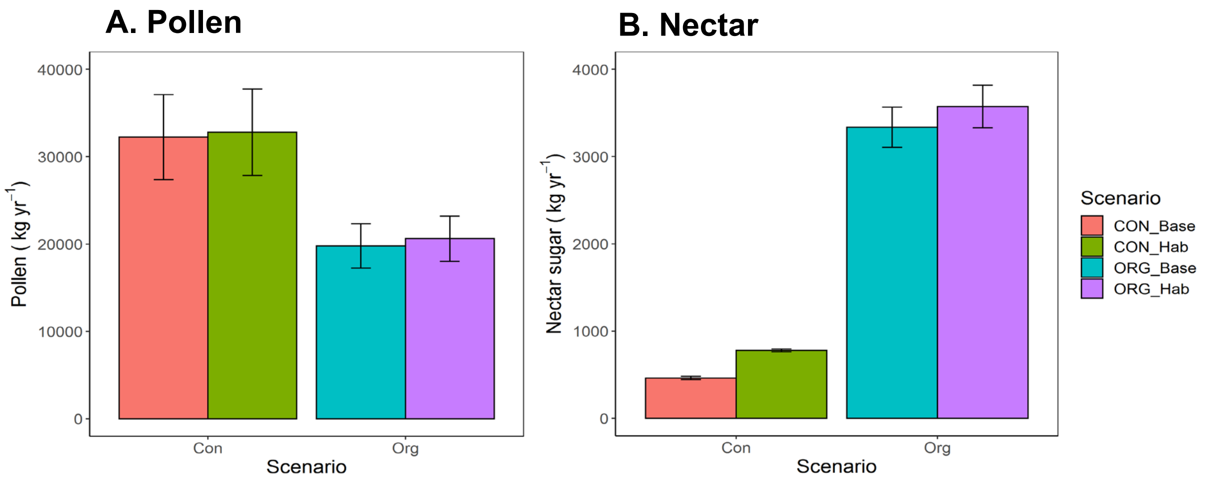


Figure 6.3: The availability of A.) Total annual pollen (kg yr-1) and B.) Nectar sugar (kg yr-1) across the six year rotation within the 3km x 3km landscape used in the Bee-Steward model. Due to slightly different areas of crops being grown each year in the rotation nectar and pollen resources fluctuate. Error bars show the standard deviation from the mean across the six years of the rotation. ORG_Base and CON_Base refer to baseline scenarios, without habitat interventions. ORG_Hab and CON_Hab refer to the baseline scenario plus habitat interventions. Considerably higher nectar sugar is produced in the organic compared to conventional scenarios. Whereas, pollen, primarily a function of maize growing, is higher in the conventional scenarios. Habitat interventions show marginal increases in resources in most scenarios, apart from for conventional nectar where there is a 59% increase in nectar sugar availability at the landscape scale through the addition of pollen and nectar habitats.

6.3.2. BEE-STEWARD Bumblebee population outputs

BEE-STEWARD outputs show that the landscape scenario had a significant impact on *B.hortorum* population stocks, with the organic baseline landscape supporting significantly higher numbers of *B.hortorum* colonies (p = <0.001) than the conventional landscape (Figure 6.4A). The addition of habitat interventions also significantly increased *B.hortorum* colony numbers in both conventional and organic landscapes (Con_Base to Con_Hab p = 0.002, and Org_Base to Org_Hab p = 0.009). Whilst the same general trends were observed for *B.terrestris*, there were no significant differences in colony numbers between landscape scenarios. One trend that was different between *B.terrestris* and *B.hortorum* outputs was for organic baseline and organic habitat scenario comparisons. Here, model outputs show a slight decrease in mean *B.terrestris* colony numbers due to habitat interventions (Table 6.3). However, the differences between the two scenarios were not significant.

All landscape scenarios showed greater suitability for *B.hortorum* with much higher reported nesting densities when compared to *B.terrestris.* For example, mean (± SD) peak colony density for *B.hortorum* in the organic baseline scenario was 59.9 ± 4.6 colonies km^{-2}, over ten times higher than densities for *B.terrestris,* 4.97 ± 3.20 colonies km^{-2} in the same landscape. Furthermore, *B.terrestris* populations went extinct in multiple runs across all landscape scenarios. A summary of model results are presented for both species in Table 6.3.

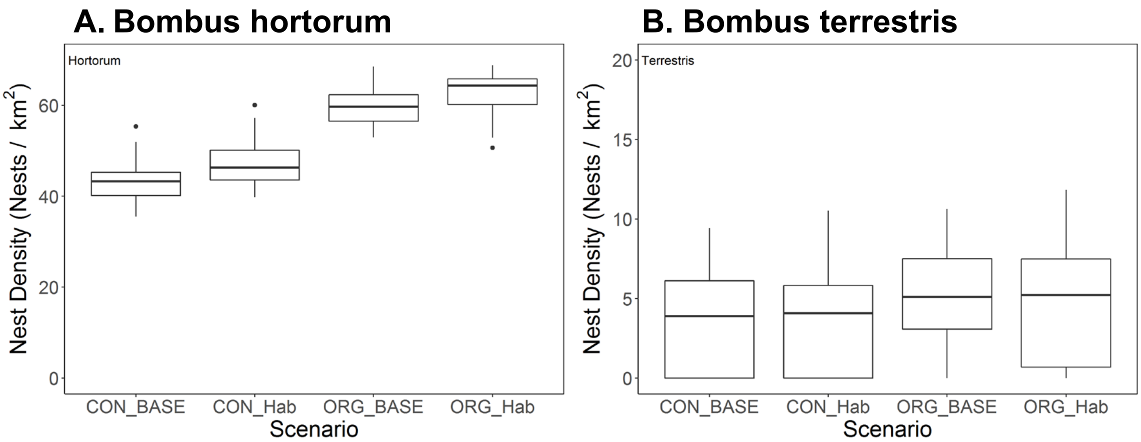


Figure 6.4: B.hortorum (A.) and B.terrestris (B.) colony nesting density in the four landscape scenarios across the 30 model runs. ORG_Base and CON_Base refer to baseline scenarios, without habitat interventions. ORG_Hab and CON_Hab refer to he baseline scenario plus habitat interventions. Significant differences were detected in colony numbers for Bombus hortorum between the Con_Base and Org_Base scenarios. Habitat interventions had a significant impact on increasing Bombus hortorum from Con_Base to Con_Hab and for Org_Base to Org_Hab. No significant differences were detected in Bombus terrestris colony numbers between landscape scenarios.

Table 6.3. shows that mean colony numbers, colony density, adult bees produced and, where applicable, the time to extinction were all higher in organic compared to conventional simulations. The model outputs suggest that landscape-scale conversion from a majority conventional (Con_Base) to a majority organic agriculture (Org_Base) increases mean colony density by 44% for *B.terrestris* and by 38% for *B.hortorum*. With more adult bees supported in the organic scenario without interventions (Org_Base), on average 85.66% more *B.terrestris* and 41.02% more *B.hortorum*. Changes in colony and bee numbers were relatively smaller following the addition of habitat interventions, with the addition of habitats having the greatest positive effect within the conventional landscape scenario. Here increases in mean colony density from convention scenario without intervention to the conventional scenario with interventions were 6.25%

for *B.terrestris* and 9.45% for *B.hortorum.* Interestingly, *B.terrestris* colonies declined (-7.44%) with the addition of habitat interventions in the organic landscape and *B.hortorum* colony density increased by 5.68%.

Table 6.3: Summary data of model outputs for B.terrestris and B.hortorum across the four landscape scenarios. Data are summarised as the mean number of colonies (at the peak during each model run) and adult bumblebees and the mean peak colony density from year 12 – 24, across all 30 model runs. The number of model runs where the bee populations survived are shown (surviving runs), along with the mean time to extinction for those runs where a bumblebee population was not sustained for the full model duration (24 years).

Characteristic	ORG_BASE N = 30[1]	ORG_Hab N = 30[1]	CON_BASE N = 30[1]	CON_Hab N = 30[1]
Bombus terrestris				
No. colonies (peak)	45 (29)	42 (34)	31 (28)	34 (28)
Colony Density (peak) (km^2)	4.97 (3.20)	4.6 (3.7)	3.45 (3.04)	3.68 (3.12)
No. bees	1,158,905 (1,058,480)	1,087,717 (1,052,775)	624,212 (736,889)	665,490 (716,258)
Time to extinction	5,282 (2,076)	5,220 (2,441)	5,032 (2,084)	4,805 (2,037)
No. surviving runs	18	14	8	11
Bombus hortorum				
No. colonies (peak)	546 (42)	577 (51)	396 (44)	433 (49)
Colony Density (peak)	59.9 (4.6)	63.3 (5.6)	43.4 (4.8)	47.5 (5.4)
No. bees	13,765,280 (1,232,258)	14,325,272 (1,226,075)	9,782,821 (958,092)	10,611,695 (1,006,910)
Time to extinction	NA	NA	NA	NA
No. surviving runs	30	30	30	30

[1]Mean (SD)

6.3.3 Provisioning ecosystem services; trade-offs and benefits

6.3.3.1 Forage biomass yields

The mean annual production of forage crop biomass (DM t yr^{-1}) are presented for each of the four landscapes across four productivity range scenarios[21] in Figure 6.5. As predicted, there is a clear trade-off in agricultural production when shifting

[21] The productivity ranges to calculate forage biomass are based on the following:
Low: Uses the lowest forage crop biomass figure observed from the study fields for each crop type; Mean: Uses the mean forage crop biomass figures observed from the study fields for each crop type; Max: Uses the highest forage crop biomass figure observed from the study fields for each crop type; IndStd: Uses the average forage crop biomass figures provided in organic and conventional industry handbooks

the majority of the landscape to organic agriculture. The gap in production is largest when organic and conventional yields are at their lowest (29% lower forage crop biomass production) and smallest when both organic and conventional yields were observed at their peak (5% lower forage crop biomass). Using average yields presented in industry handbooks (Nix, 2018; ORC, 2017) shows a 22% reduction in forage crop biomass production when shifting the majority of the landscape to organic agriculture. The addition of habitat interventions shows a reduction in forage yields of around 3% for both organic and conventional scenarios across all ranges.

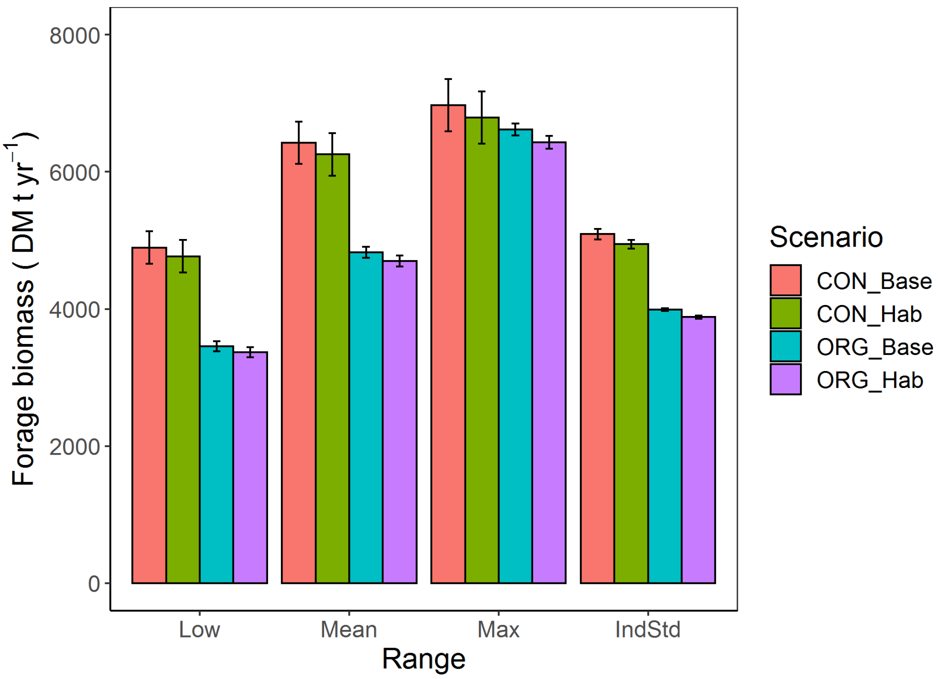


Figure 6.5: Forage biomass production (DM tonnes yr^{-1}) across the four landscape scenarios and for four ranges of productivity. Error bars show the standard deviation from the mean across the six years of the rotation. ORG_Base and CON_Base refer to baseline scenarios, without habitat interventions. ORG_Hab and CON_Hab refer to the baseline scenario with habitat interventions. Low, mean and max calculations are based on the lowest, average and maximum field yields for all sampled crops observed in organic and conventional study sites. IndStd forage biomass is calculated using average yield figures reported in industry handbooks for conventional (Nix, 2018) and organic (ORC, 2017) farms.

6.3.3.2 Producer surplus

Producer surplus (£ yr^{-1}), at the landscape scale, generally appears to increase or remain similar under a majority shift to organic agriculture across most productivity ranges[22] (Figure 6.6). Despite the lower yields observed in Figure 6.5, the organic landscape scenarios (Org_Base and Org_Hab) show a higher producer surplus than conventional landscape scenarios (Con_Base and Con_Hab) in mean and maximum productive ranges. The largest gap between producer surplus arises in the comparison of conventional baseline (Con_Base) to organic baseline (Org_Base) scenarios where organic crop producer surpluses were at their highest (max), showing a 61% increase in producer surplus at the landscape scale. Using industry standard yields (IndStd) and the crop costs of production, the producer surplus change was marginal, on average generating a 1.4% increase following a shift from the conventional baseline scenario (Con_Base, £181,106 yr^{-1}) to the organic baseline scenario (Org_Base, £183,708 yr^{-1}) . When using industry standard yields to calculate producer surplus, it was evident that some years the conventional baseline scenario and other years, the organic scenario had higher producer surplus. Whilst producer surplus increased under organic agriculture for mean, maximum and marginally for industry standard (IndStd) scenarios, this was not the case when organic crop yields were at their lowest. Here, the organic baseline scenario showed a 21% lower producer surplus than the conventional baseline scenario.

Saleable yield and crop producer surplus data used in the calculations are presented in Table 6.4. As a reminder, saleable yields refer to the crop as sold (i.e. cereal grain and fresh weight maize and grass silage), as opposed to the DM forage biomass used in the calculations for Figure 6.5.

Habitat interventions reduced producer surplus at a landscape scale by between 2.76-3.77% from the baseline scenarios across the difference ranges. Annual costs of habitat installation and management were relatively low, calculated at £1,586 yr^{-1} for organic farmland and £1,562 yr^{-1} for conventional farmland. Coupled with losses in production, the reduction in producer surplus from habitat installation ranged from £4,863 - £8,567 yr^{-1} in the conventional landscape and from £4,663 - £16,080 yr^{-1} in the organic landscape. Using industry standard

[22] The productivity ranges to calculate producer surplus are based on the following:
Low: Uses the lowest producer surplus figure observed from the study fields for each crop type
Mean: Uses the mean producer surplus figures observed from the study fields for each crop type
Max: Uses the highest producer surplus figure observed from the study fields for each crop type
IndStd: Is calculated using the average yields provided in organic and conventional industry handbooks to calculate crop revenue less the costs observed for the study fields for each crop type

yields, the mean annual loss was calculated at £6,090 yr^{-1} for the conventional landscape and £6,862 yr^{-1} for the organic landscape.

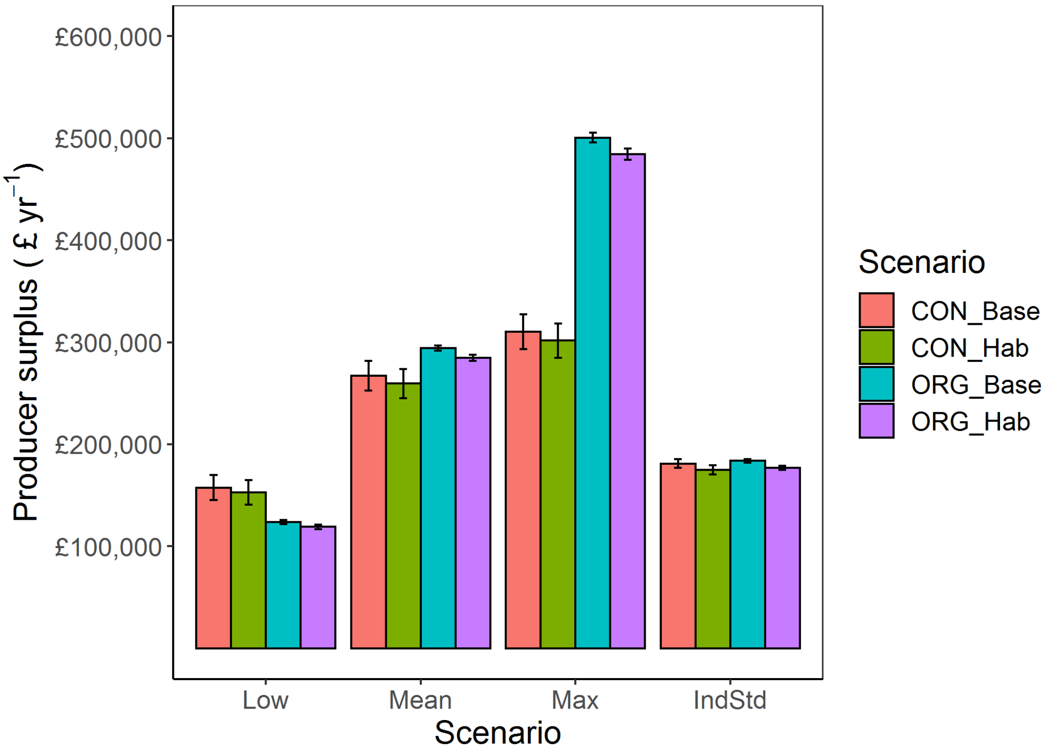


Figure 6.6: Producer surplus (£ yr^{-1}) across the four landscape scenarios and for four ranges of crop gross margins. Error bars show the standard deviation from the mean across the six years of the rotation. ORG_Base and CON_Base refer to baseline scenarios without habitat interventions. ORG_Hab and CON_Hab refer to the baseline scenario with habitat interventions. Low, mean and max calculations are based on the lowest, average and maximum field gross margins for all sampled crops observed in organic and conventional study sites. IndStd forage biomass is calculated using average yield figures reported in industry handbooks for conventional (Nix, 2018) and organic (ORC, 2017) farms.

Table 6.4: Summary of the yield and crop producer surplus data collected in this study and used in the calculation of producer surplus at the landscape scale. 2019 Low and 2019 Max data were generated from the lowest and highest yielding field and lowest and highest producer surplus at a field scale. 2019 Mean shows the mean across all sampled fields of that crop type and IndStd refers to average yield data taken from industry handbooks (Nix, 2018; ORC, 2017). IndStd yields were combined with actual crop costs of production to calculate IndStd producer surplus.

	Crop Price	**Saleable yield (t ha^{-1})**				**Gross margin (£ ha^{-1})**			
	(£ t^{-1})	**2019 Low**	**2019 Mean**	**2019 Max**	**IndStd**	**2019 Low**	**2019 Mean**	**2019 Max**	**IndStd**
Organic silage	£50.55	21.79	37.07	62.44	28	£258.45	£685.02	£1,300.71	£448.18
Organic cereal (grain)	£230.50	3.04	3.99	5.23	3.2	£241.29	£621.33	£1,028.81	£361.38
Organic cereal (straw)	£65.00	3.99	5.22	6.85	3.5	na	na	na	na
Conventional cereal (grain)	£147.60	7.72	9.29	10.51	8.2	£736.74	£1,056.88	£1,230.41	£543.47
Conventional cereal (straw)	£65.00	9.81	13.66	15.87	3.5	na	na	na	na
Conventional maize	£34.00	38.20	49.32	56.29	40	£586.23	£877.22	£1,079.14	£560.46
Conventional silage*	£36.42	36.00	48	48.00	47	£83.57	£ 280.61	£ 280.61	£264.19

* Taken from industry data (AHDB, 2020; Nix, 2018) (max used for comparisons due to high yielding year in 2019)

6.4 Discussion

The BEE-STEWARD outputs show that the landscape-scale shift to organic agriculture in this case study has the capacity to significantly increase bumblebee population stocks and generates a large amount of pollen and nectar which is likely to support a wide range of other pollinators. Despite a clear trade-off in yield, the case-study shows that improvements in bumblebee stocks can coincide with similar or even improved economic surplus from farm outputs. Habitat interventions had a less marked impact on bumblebee stocks and came with a clear trade-off in terms of yield and producer surplus. An expanded discussion on floral resource provision, bumblebee populations and trade-offs is provided below, before considering the cost-effectiveness of different strategies in delivering bumblebee improvements.

6.4.1 BEE-STEWARD outputs; Do pollen and nectar habitat interventions and/or a landscape-scale shift from conventional to organic agriculture enhance floral resources available to insect pollinators?

The higher floral abundance and the corresponding increase in nectar availability on organic land, particularly on cropped areas, was unsurprising given restrictions on the use of herbicides and as a result, higher densities of 'weeds' (Geppert *et al.*, 2020). Flower density was also expected to be higher on habitat intervention plots and margins, where a targeted mix of flowers are sown (Hardman *et al.*, 2016a; Geppert *et al.*, 2020). The finding is important, as higher floral resources have been closely associated with the state of pollinator abundance and diversity (Roulston and Goodell, 2011; Baude *et al.*, 2016; Sutter *et al.*, 2017).

Despite higher overall pollen provision in the conventional landscape, this can almost entirely be attributed to an abundance of maize pollen (96%). Landscapes dominated by maize and offering such a homogenous source of pollen have actually been found to adversely impact bumblebees (Hass *et al.*, 2019). The result supports other studies that have observed higher floral diversity and abundance on organic farmland (Tuck *et al.*, 2014) and this has been associated with higher pollinator abundance and diversity and pollination services (Holzschuh, Steffan-dewenter and Tscharntke, 2008; Hardman *et al.*, 2016a; Geppert *et al.*, 2020). Some of the observed weed species - such as *Tripleurospermum inodorum* (scentless mayweed) - were particularly abundant in some organic crops (e.g. organic cereals 1 and 2) and habitat interventions (flower margins and plots). Whilst *T. inodorum* is not utilised by bumblebees, it has been identified as being particularly important for other bee pollinators (Nichols, Goulson and Holland, 2019). It seems reasonable to suggest therefore, that in the case study area, the shift to organic agriculture could considerably increase wider pollinator abundance and diversity from the conventional baseline.

Whilst the addition of habitat interventions to both conventional and organic landscapes did increase floral resources, the magnitude of that increase was lower than when shifting from the conventional to the organic landscape scenario. The area of habitat interventions added to the case-study landscape were in-line with Countryside Stewardship (CS) Farm Wildlife and Pollinator Package (FWPP) advice (>1ha pollen and nectar resources per 100ha of arable/mixed farmland) (Dicks *et al.*, 2015) but it is clear that a greater area of pollen and nectar rich habitat interventions would need to be added to a conventional landscape to match the resource availability provided by the model organic landscape. Other studies have also observed that organic farmland has the capacity to provide floral resources over a far wider area, than smaller dense flower plots and margins (Geppert *et al.*, 2020). Relative increases in pollen and nectar were greatest on conventional farmland, compared to organic farmland. The finding makes sense, given that sowing pollen and nectar flower plots on organic farmland results in the loss of some weedy crop habitat, which already has some pollinator benefits. The same observation was made in the proportional response of bumblebee populations, which is discussed further in the next section.

6.4.2 BEE-STEWARD outputs: do pollen and nectar habitat interventions and/or a landscape-scale shift from conventional to organic agriculture enhance long-term bumblebee populations?

BEE-STEWARD outputs show that a shift to organic agriculture significantly increases *B. hortorum* populations, compared to the conventional landscape scenario. The addition of habitat interventions, in line with the CS FWPP, to baseline conventional and organic landscapes also had a smaller but still significant impact on *B. hortorum* populations. The results are in line with other studies that have found higher bumblebee abundance and colony development on organic compared to conventional farmland (Rundlöf, Nilsson and Smith, 2008; Geppert *et al.*, 2020) and on farmland with added habitat interventions (Pywell *et al.*, 2006, 2015; Carvell *et al.*, 2007; Wood *et al.*, 2015).

Whilst, the same patterns were generally seen for *B. terrestris,* it was interesting to observe that these differences were not significant and were small compared to other modelled (Becher *et al.*, 2018) and empirical studies reporting *B. terrestris* colony density (Wood *et al.*, 2015). Surprisingly, no significant difference in *B. terrestris* populations between the different landscape scenarios were observed, particularly when shifting to the organic landscape scenario where there was a 623% increase in nectar resources (Figure 6.3 in Results). Interestingly, Wood *et al.,* (2015), in their field observation based study, also found significant increases in *B. hortorum* colonies but no significant differences in *B. terrestris* colonies, as a response to agri-environment scheme management. Further testing of the model parameters would be required to fully understand model results for *B. terrestris* but it is probable that it is a consequence of differences in species tongue length and emergence dates. *B. terrestis* have a shorter tongue length than *B. hortorum* and therefore, cannot access plants with deeper corolla tube's, some of which - such as *Raphanus raphanistrum* (wild radish) - were particularly abundant in some organic crops (organic cereals 1 - 3) and habitat interventions (flower plots). Furthermore, *B. terrestris* tend to emerge earlier in the model (1st April ± 28 days) compared to *B. hortorum* (30th April ± 8 days), during a period when low floral resource availability was observed in the case-study landscape (see Appendix D.7 for annual pollen and nectar availability chart). Floral resources tend to increase in the model around the 30th April and into May as bramble and clovers start to come into flower so the scarcity of floral resources through April is reducing the establishment and resilience of modelled *B.terrestris* populations. Further work would be required to establish whether this is a genuine problem in the case-study landscape (i.e. is there a lack of early season forage) or an artefact of the model parameters.

The BEE-STEWARD results for *B. hortorum* are important as they provide strong evidence to show that organic farming and the addition of habitat interventions can, at the very least, increase populations of some bee species at the landscape scale. Wood *et al.,* (2015) highlight that to date a consistent problem in evaluating bumblebee response to agri-environment schemes (and the same could apply to bumblebee transects on organic farmland) has been that it is not clear whether high bumblebee abundance recorded on flower patches is a function of attracting lots of workers to flower rich areas or due to a population increase in the wider landscape. BEE-STEWARD offers the advantage of providing insight into the landscape scale population response. Both strategies clearly have the capacity to support greater bee populations, which are likely to be more resilient to change and deliver more pollination services. Changes in *B. hortorum* populations are clearly greater following a shift to organic agriculture (38% increase in colony density) when compared to the addition of small habitat patches on conventional farmland (9.45% increase in colony density). Coupled with evidence to suggest that organic agriculture could generate more producer surplus for the farmers provides a compelling case for its role in improving pollinator stocks.

As with proportional change in floral resources it was found that adding habitat interventions to conventional land had a greater impact than interventions on organic land. It supports Scheper *et al.*, (2013) suggestions concerning the impact of ecological contrasts, with the largest changes in pollinators generated by large contrast changes in resources. The study found that AES interventions were most effective when incorporated into "structurally simple, resource-poor landscapes dominated by arable fields". The finding suggests that adding habitat interventions to enhance floral resources for pollinators would be best targeted at conventional, not organic farmland.

It is important to highlight that the results here represent a single case-study landscape and a number of studies have identified the importance of landscape heterogeneity on the response of bumblebees to organic management (Rundlöf, Nilsson and Smith, 2008) and to habitat interventions (Carvell *et al.*, 2007; Geppert *et al.*, 2020). Bumblebee colony density has been found to closely align with surrounding semi-natural habitat (Knight *et al.*, 2009) and field size also impacts on wild bee abundance (Hass *et al.*, 2018). The case-study landscape had a mean field size of 5 ± 2.46 ha, an annually-tilled arable area of 24 % and a pasture area of 20%, which is closer to the (Rundlöf, Nilsson and Smith, 2008) classification of the heterogenous landscapes used in their study (although the mean field size and arable area was larger in this study than in theirs: 3.11 - 3.21ha and 15-16% respectively). The case-study average field size of 5ha is slightly higher than the Devon average (4ha; Britt *et al.*, 2011) but still significantly smaller than the homogenous landscapes of Cambridge (16ha; Robinson and

Sutherland, 2002). Semi-natural habitat areas (woodland, hedges, scrub and unimproved grassland) defined in the model were relatively low (8.24%) compared to Knight *et al.,* (2009) (19.4%) but within range of the study sites used by Carvell *et al.*, (2011) (4.4% - 33.4%; median 13.85%). These details are relevant as organic agriculture has been found to be most effective at supporting higher numbers of bumblebees in more homogenous landscapes (Holzschuh *et al.*, 2007; Rundlöf, Nilsson and Smith, 2008). Further work would be required to validate these finding in other areas but it could be the case that in more homogenous landscapes (larger fields, less semi-natural habitats) the contrast in bumblebee populations supported between organic and conventional with/or without habitat interventions could be greater.

6.4.3 The trade-offs and/or benefits of different land management scenarios

A common criticism levelled at organic systems is that ecological benefits typically come with the trade-off of reduced yield (Tuck *et al.*, 2014). The same was observed in this case-study, with a majority shift from conventional to organic agriculture reducing DM biomass (tonnes DM year^{-1}) output between 5 – 29% at the landscape scale. However, despite lower yields, premiums paid for organic products, and in some cases the lower cost of production, typically resulted in increased or very similar producer surplus at the landscape scale. Where organic yields were at mean, max or industry standard ranges, there was no observed trade-off between increased pollinator resources/bumblebee populations and the economic surplus from agricultural outputs for farmers.

In some instances, there is evidence that organic conversion can deliver a 'win-win', with increasing producer surplus alongside benefits in pollinator resources and bumblebee populations. However, this was based on yield data from 2019 only (for mean and max producer surplus ranges). Using the industry standard figures for yield (average yields from multiple years and farms) and the costs calculated in this study, the organic scenario generates a very similar annual producer surplus (£183,708.07 ± £1639.32) compared to the conventional (£181,105.98 ± £4415.25). As such, data using industry standard yields shows that organic conversion is more likely to create a 'break-even and win' situation, with the farmer observing limited changes in average economic surplus (without taking agri-environment scheme payments into account)) and there being significant increases in pollinator resources and bumblebee populations.

Maintaining a 'win-win' or 'break-even and win' scenario is contingent on good organic crop yields and the state of the organic market, with the profitability of organic farming typically dependent on the price premium paid for products (Crowder and Reganold, 2015; Röös *et al.*, 2018). It will also be impacted by the scale of government support for organic conversion and management (as discussed in Section 4.4). The impact of lower yields on profitability was observed in this study, with low yield ranges resulting in the contrary outcome where producer surplus is lower for the organic versus conventional scenario (Figure 6.6 in Results). The volatility in yields is recognised as having a large impact on the profitability of other organic farms (Smith *et al.*, 2019) and presents a significant risk for organic producers, being cited as a key reason why farmers have chosen not to convert to organic agriculture (Stochlic and Sierra, 2007; Łuczka and Kalinowski, 2020). In addition, there is a risk that expansion of organic agriculture, whilst improving pollinator stocks, could adversely impact organic farmers under circumstances where supply outstrips demand and organic crop prices and as such producer surplus declines. Crowder and Reganold (2015) however, identify that whilst organic agriculture's profitability relies on price premia (currently 29 – 32% higher than conventional products), these could still be reduced to 5 – 7% and organic farms would still be as profitable as conventional farms. Furthermore, demand for organic produce is in a state of growth within the UK, Europe and USA (Scott, 2020) and a number of other European countries already have a far greater proportion of their utilisable agricultural land (7 – 15%) under organic production (UK, 2.6%) (Scott, 2020). It seems reasonable, therefore, to argue that there is room for the expansion of productive organic agriculture in the UK whilst delivering benefits for both pollinators and producers.

In contrast to organic conversion, in this study taking land out of production to enable habitat interventions, whilst increasing *B. hortorum* populations, display a clear trade-off in terms of reduced yield and producer surplus. However, the simple calculation of the yield loss from such interventions (habitat area created*standard yield) could overstate yield reductions. For example, Pywell *et al.,* (2015) found that the addition of agri-environment scheme interventions can offset the loss in area by generating increased yield in remaining field areas of winter wheat, oil seed rape and field beans.

Trade-offs in producer surplus could also be offset by increases in other ES. It was beyond the scope of this study to conduct a cost-benefit analysis of the different scenarios and account for the value of other ES, including pollination. Valuation of pollination services is complex and was further complicated by the fact that the case-study landscape is dominated by wind pollinated crops. The yields of these crops are not dependant on pollinator abundance. However, it raises the question of the value of pollinators in such landscapes and further research is therefore advised on the commercial (crop production function) and social values of pollinators. This is further explored in Chapter 7.

6.4.4 What is the most cost-effective strategy to increase bumblebee populations?

As highlighted in the introduction, previous studies investigating ecological outputs of agri-environment scheme interventions rarely consider cost-effectiveness (Batáry *et al.*, 2015). Such cost-effectiveness calculations are important in developing future agri-environment schemes and understanding how changes in food markets (e.g. for organic produce) could impact land use and the delivery of ES. As detailed in Table 6.5, in this study the cost-effectiveness of two scenarios in generating changes to *B.hortorum* populations has been investigated. Considered in the context of payments for ecosystem services (PES) and assuming that more bumblebees equates to more pollinator services then Table 6.5 considers the most efficient way of achieving these changes taking into account 1.) current government payments and 2.) the costs borne by the farmer to intervene voluntarily.

Panel A of Table 6.5 shows the payments made by government to farmers to support organic conversion and the addition of habitat interventions based on current agri-environment schemes (Mid-Tier CSS). Panel B shows the costs to the farmer(s) of switching to organic agriculture or adding habitats without government support payments, using data collected in this study. The costs to the farmer include the costs of installing and maintaining habitats and the losses made in crop producer surplus (using industry standard yields) from taking areas out of production (for habitats) or switching to organic management. Panel C offers additional insight into the net benefit/cost to the farmer of each scenario, calculated as potential agri-environment scheme payment less the costs to the farmer (as detailed above).

Table 6.5: Proportional costs per uplift in annual mean number of colonies and adult bees based on the shift from CON_Base to CON_Hab, ORG_Base to ORG_Hab and from CON_Base to ORG_Base. The cost of generating changes in populations of B. hortorum are based on a) the payments made by government to incentivise the land manager to install habitat or shift to organic agriculture (the latter includes both conversion and management payments and is based on the average annual cost over a 20-year period); b) based on the annual costs incurred by the farmer when installing similar habitat interventions and the annual cost (or in this case profit – shown as a minus) of changing to organic; and c) the net cost to the farmer, calculated as the payments made under Countryside Stewardship less (A.) and the costs to the farmer (B.) Costs for table B are based on using industry average yields in the calculation of producer surplus.

A.) Applying the payments made under the Countryside Stewardship Scheme:			
	CON_BASE - CON_HAB	ORG_BASE - ORG_HAB	CON - ORG
Annual costs of intervention (per year)	£5,476.42	£5,476.42	£36,593.23
Change in colony number (per year)	28.02	19.29	139.61
Change in adult bee number (per year)	1039.50	730.42	5306.58
Cost per colony increase (per year)	**£195.45**	**£283.92**	**£262.12**
Cost per bee increase (per year)	**£5.27**	**£7.50**	**£6.90**
B.) Applying the costs to the farmer from this study:			
	CON_BASE - CON_HAB	ORG_BASE - ORG_HAB	CON - ORG
Annual costs of intervention (per year)	£6,089.60	£6,861.70	-£2,602.09
Change in colony number (per year)	28.02	19.29	139.61
Change in adult bee number (per year)	1039.50	730.42	5306.58
Cost per colony increase (per year)	**£217.33**	**£355.73**	**-£18.64**
Cost per bee increase (per year)	**£5.86**	**£9.39**	**-£0.49**
C.) Accounting for the costs/benefit to the farmer after Countryside Stewardship payments:			
Annual payments minus annual costs:	-£613.18	-£1,385.28	£39,195.31

Table 6.5, panel A shows that under current CSS payments the most cost-effective way for the government to deliver changes to *B.hortorum* populations is through the addition of habitat to conventional land. The cost per increase in modelled *B. hortorum* populations is similar across scenarios (£5.27 - £7.50 per bee; £195.45 - £283.92 per colony). Whilst organic conversion delivered a

significantly greater change in *B. hortorum* population (increase of 140 colonies), the addition of habitats to the conventional landscape offered a more cost-effective way to increase populations per colony and per bee. As a rough calculation, to match the modelled mean *B. hortorum*, colony numbers under the organic scenario (456 colonies) would require an additional 46.44ha of habitat interventions, with an estimated total annual payment of £27,213.59. Further model simulations would be required to validate these rough calculations. but these payments would be 34.47% lower than those associated with the conversion and management of organic land (£36,593.23). Accordingly, the data suggests that government payments might be better targeted at a sparing approach (i.e. sparing areas for habitats) rather than a sharing approach (i.e. supporting bees and cropping on the same area) when it comes specifically to improving pollinator abundance. This is perhaps unsurprising given that habitat interventions here are targeted to improve pollinators whereas organic agriculture provides general environmental benefits (Latacz-Lohmann and Renwick, 2002). It is therefore important to caveat this finding by noting that such policy decisions should also consider the wider ES that might arise from organic agriculture or habitat interventions.

Interestingly, it was observed in the field that organic and conventional pasture provided very similar floral resources. Based on the similarities in flower densities the same data were used to categorise both conventional and organic grass pasture in the model. Whilst organic pasture might have other environmental benefits (e.g. higher soil carbon storage (see Chapter 4), payments for organic management on pasture fields have, therefore, failed to deliver any improvement in pollinator floral resources. What is also clear from Table 6.5 Panel 1A is that the addition of habitat interventions on organic farmland were the least cost-effective, generating a lower change in *B. hortorum* colonies (as discussed in Section 4.2). The data suggest that habitat interventions are most cost-effective when targeted at conventional rather than organic farms, particularly when taking into account the fact that habitat intervention payments on organic farms are in addition to baseline conversion/management payments.

In contrast to Panel A, Table 6.5, Panel B and Panel C show that the most cost-effective means for the farmer to deliver changes to *B.hortorum* populations is through conversion to organic agriculture. Panel B presents an assessment of the annual costs/benefits borne by the farmer, in establishing the habitat interventions and of converting to organic agriculture. Alongside Panel C, it shows that the installation of habitats on conventional and organic land comes at a cost to the farmer (even with CSS payments), whereas conversion to organic agriculture shows a small mean annual profit to the farm (shown as a minus; £2,602.09 per year).

Whilst the annual costs of habitat interventions on conventional land (£6,089.60) are fairly similar to the annual CS scheme payments made (£5476.42 yr^{-1}), there is still a loss of £613.18 per year and the loss is even greater when adding habitat to organic land (£1,385.28). In theory, CSS payments are made on an 'income foregone' basis and it might be the case that the field management costs in this study are lower than those for other 'typical' UK farms used to calculate CSS payments. However, the finding of inadequate compensation for agri-environment scheme interventions has been reported by others, resulting in a lack of uptake of CSS (DEFRA, 2020; Little *et al.*, 2021). Little *et al.,* (2021) report that farmers and farming organisations have indicated that 'income foregone' based payments do not provide sufficient compensation and that the payment rates fail to consider regional variability or the skill of the farmer (being based on a 'typical farm'). Table 6.5, Panel B supports the assertions made by Little et al., (2021) and highlights the need to re-evaluate the current payment mechanisms to encourage farmers to deliver benefits to pollinators, alongside other ecosystem goods and services.

In contrast to habitat interventions, the pattern of producer surplus under organic conversion is different where it generates, on average, a small profit (not a loss) to the farmer alongside improvements in pollinator stocks. As discussed in Section 6.4.2, it is more reasonable to suggest that organic conversion would, therefore, deliver a very similar economic surplus to the farm (without taking into account CSS payments). Table 6.5, Panel B shows that the incentives provided by market returns on organic produce can sustain the organic farm but the profitability of conversion, when using industry standard yields at least, is likely to be similar to that of the conventional enterprise. The data suggests that organic agriculture supported through organic markets could deliver a strategy to enhance bumblebee populations that is both cost-effective for the farmer and cost-free for the government. However, given the risk based on wider variability in organic yields (discussed in Section 6.4.2), the relatively meagre increase in producer surplus (when using industry standard yields) it unlikely to offer sufficient incentive for widespread conversion. Indeed, significant increases in profitability (from £2,602.09 to £39,195.31) are only observed when adding government CSS payments (Table 6.5 Panel C). Latacz-Lohmann and Renwick (2002), in their review of organic subsidy support in the UK, argue that reliance on the organic market has typically resulted in sub-optimal delivery of ecosystem goods and services. Despite signs of growing demand and profitability, farmers have held back from organic conversion due to concerns over yield volatility (risks of pests and diseases), production costs, losses during transitional period, limited access to technical assistance, certification costs and importantly, either a lack of, or difficulties in accessing, organic price premiums and markets (Stochlic and Sierra, 2007; Łuczka and Kalinowski, 2020). Therefore, despite the data suggesting that organic agriculture supported by the organic market alone could be a cost-effective strategy to increase pollinator populations, it is unlikely that expansion will occur without government support. Further understanding of the future of organic markets and the necessary scale of government support is required to understand the potential of widespread organic conversion in the UK, as a strategy to deliver a cost-effective solution to enhance pollinators and benefit producers.

6.5 Conclusion

This study found that both the shift to organic agriculture and the addition of habitat interventions had the capacity to increase floral resources and bumblebee populations when applying the BEE-STEWARD software (which combines the Bumble-BEEHAVE and BEESCOUT models). In this case-study landscape it was found that a majority shift to organic agriculture increased floral resources available to insect pollinators and had the capacity to significantly increase *B.hortorum* populations. Whilst habitat interventions in-line with the areas required under the current Mid-Tier Countryside Stewardship Scheme for the Farm Wildlife and Pollinator Package also had the capacity to increase floral resources and significantly increase *B.hortorum* populations, changes were less pronounced than under the organic conversion scenario. Furthermore, they were more effective on conventional rather than organic land supporting the concept of ecological contrasts. Changes in *B.terrestris* species were not significant with the addition of habitat interventions or organic conversion. Further research and testing is required to understand if this is a genuine problem in the study landscape, investigating whether there is a lack of early season forage or if the results are an artefact of the model parameters. It would be valuable to also run the model for other bumblebee species with different ecological traits and across a wide range of different landscapes.

By incorporating data on crop yield, costs of production and cost revenues, it was possible to calculate producer surplus and establish the trade-offs or 'win-win' scenarios that might arise under organic conversion or adding habitat interventions. This was a valuable exercise and demonstrates that in the case-study landscape whilst organic conversion reduced crop yield, it had the potential to deliver increased or very similar economic surpluses for the farmer alongside the benefits to bumblebees. This is in contrast to the addition of habitat interventions on both conventional and organic land, with benefits in pollinator resources and bumblebee populations coming with the trade-off in farmer producer surplus. The continued delivery of a 'win-win' scenario under organic management is dependent on maintaining good organic crop yields. Such actions will be vulnerable to changing price premiums for organic produce. This case study provides a positive example of organic farming balancing healthy pollinator stocks with producer income. However, further validation work is required which extends the same accounting method across other farm landscapes and over multiple years.

When considering the most cost-effective strategy to enhance bumblebee populations, the study illustrates how organic agriculture, supported by organic price premiums, has the potential to be cost-effective for the farmer and, in principle, cost-free for the government. That is the organic farm appears to be profitable without government support, which could suggest that government subsidies could be dropped whilst still delivering enhancements in pollinators as a positive externality of the organic market. It is unlikely, however, given reports from the literature and observations made in this study (high yield volatility and only marginal profit increase), that expansion of organic agriculture in the UK will occur without government support. As being developed under the Farm to Fork strategy in the EU (European Commission, 2021), further research is required on organic yield volatility, organic markets and the scale of government support required to understand the capacity for widespread expansion of organic agriculture as a cost-effective strategy to deliver biodiversity benefits. It is important to note that adding targeted pollen and nectar habitat interventions had the potential to be more cost-effective than organic conversion under the current Countryside Stewardship Scheme payments. Taking into account PES, the data suggests that current government payments might be better targeted at a sparing approach, rather than a sharing approach, when it comes specifically to improving pollinator services. However, such policy decisions also need to consider wider ES that might arise from organic agriculture or habitat interventions. Incorporating these ES assessments into future work on organic and habitat intervention scenarios would be valuable to optimise future agri-environment schemes.

To conclude, it is recognised that the estimations of cost-effectiveness presented here are coarse and will depend on farm management cost data, the performance of different habitat interventions (i.e. flowering densities) and differences in weed densities on organic farms. However, the study provides one of only a few examples that have examined the cost-effectiveness of different agri-environment schemes in generating biodiversity benefits. It provides an example for other studies to examine the cost of on farm interventions and biodiversity benefits and contributes to the debate over future payment scales for AES.

Chapter 7: A systematic application of the natural capital approach at the farm scale: Is it a practical tool for routine land management decision-making?

7.1 Introduction

The NC approach holds out the promise of being the long-awaited panacea to sustainable land management decision-making and evaluation. It presents a structured framework for understanding the impact of land management on NC, the response in EF and the resulting change in value flows from ES. This approach to decision-making is commensurate with other sustainability objectives; supporting the principle that decisions should not be single-focused (say on profits from food production) but should consider the full range of benefits derived from land. Indeed, it embeds land management decision-making in the long-established practices of social cost-benefit analysis, providing a toolbox through which land managers can account for both the market (e.g. crops, timber and water) and non-market (e.g. climate regulation and flood alleviation) goods and services that are impacted by their decisions (Hanley *et al.*, 2015; Bateman and Mace, 2020; Ovando, 2021).

Given this potential, it is not surprising that there is growing advocacy for application of the NC approach in land management decision-making. Influential organisations such as the Natural Capital Committee and Natural Capital Coalition have actively encouraged its adoption at local and organisational scales (e.g. for individual farm or estate businesses) (Natural Capital Coalition, 2017; Natural Capital Committee, 2017). This is perhaps unsurprising given the critical role that farms and estates play in the preservation of the natural environment (Faccioli *et al.*, 2020). Since around 70% of the UK is managed as farmland (Connors, 2016) and in England roughly 91.5% of land is thought to be privately owned and managed (Shrubsole, 2019), meaningful change in the condition of the UK's NC will only be affected by realizing a fundamental change in the decision-making processes at the farm scale.

Despite positivity around the application of the NC approach, the vast majority of applications at the farm scale have been partial. Often, these partial applications focus solely on a detailed assessment of the physical condition of a farm’s NC, without attempting to estimate value flows (e.g. Smukler *et al.*, 2010; Gabriel *et al.*, 2013; Williams and Hedlund, 2013). Alternatively, applications may focus only on ecosystem valuation, simply proxying a farm's NC by assuming it resembles broadly similar habitat types for which approximate value flows have been estimated elsewhere (Silcock and Russ, 2018; Ovando, 2020) (although see Sandhu *et al.*, 2008; Porter *et al.*, 2009; Ghaley, Sandhu and Porter, 2015; Fan, Henriksen and Porter, 2016). This chapter presents details of one of the first attempts to implement a complete application of the NC approach at the farm scale in the UK. The study takes four ES pathways and applies the complete NC approach to a case-study farm estate contrasting a reality in which large parts of the farm estate converted to organic management from 2007 (referred to herein as the 'organic scenario'), to a counterfactual in which that land remained under conventional management (referred to herein as the 'conventional scenario').

This study demonstrates that it is possible to undertake an analysis of this nature but there are a number of significant challenges. In summary, these challenges include selecting meaningful metrics, accessing data and models and meeting the long-term data, resource, expertise and cost requirements involved in conducting this work at the farm scale. In some cases, the data required just does not exist, particularly when trying to value ES linked to biodiversity. The research raises a significant challenge to those advocating the application of the NC approach at the farm scale. It raises questions over whether, given the resources currently available it: 1) is practically possible for farm-managers to undertake the analyses required to do this properly; and 2) provides suitably robust information to inform field to farm management decision-making?

7.1.1 What is the natural capital approach?

The NC approach arises from the premise that stocks of NC (e.g. soil, air, water, ecosystems) can be replenished or degraded. The quality and quantity of these stocks (NC condition), along with various anthropogenic drivers and environmental processes (defined here as EFs), underpins the delivery of goods, services and disservices (ES) that directly impact on the welfare of individuals in society. A number of similar cascade frameworks have been presented that apply this thinking, conceptually linking NC condition to EF and the delivery of ES (Haines-young and Potschin, 2008; Dominati, Patterson and Mackay, 2010; Maseyk *et al.*, 2017). The components of these frameworks typically comprise three tiers: NC stocks (Tier 1), EF (Tier 2) (sometimes called processes) and ES (Tier 3). The ultimate goal of the NC approach is to utilise the necessary data from each tier to quantify the economic value of the multiple ES derived from NC. The intention is to conduct a holistic appraisal of the ES benefits and trade-offs that arise from a land management decision, including both market goods (e.g. food and water) and non-market services (e.g. climate regulation and recreation). Expressing these benefits and trade-offs in monetary terms (ES values), facilitates application of cost-benefit analysis as a decision-support tool; ES values can be examined in the same units as the financial costs and the economic returns of different land management scenarios. In doing so, advocates suggest that the NC approach can be utilised to select land management practices which are cost-effective and maximise the output of benefits to humans (Maseyk *et al.*, 2017; EFTEC, 2019; Bateman and Mace, 2020; Defra, 2020a).

7.1.2 How have land management decisions been evaluated in the past?

In the past natural scientists have typically measured properties from Tier 1 (NC condition) to characterise and compare environmental conditions or have tried to understand how ecological systems operate through measurements of Tier 2 (EF). It is at these tiers that change usually occurs due to alterations in land management. With respect to NC condition, metrics such as species richness have been used to evaluate changes in biodiversity (Kremen and Miles, 2012; Duncan, Thompson and Pettorelli, 2015), nutrient pollutant status (e.g. P or N) to assess water quality (Keeler *et al.*, 2012; Peukert *et al.*, 2014) and soil structure or nutrient levels (N, P, K) to evaluate soil condition (Peukert *et al.*, 2012; Rickson *et al.*, 2012; Glendell *et al.*, 2014; Greiner *et al.*, 2017). Likewise, measurements of EF have included pollination (Hardman *et al.*, 2016a), carbon sequestration (Poulton *et al.*, 2018) or nitrate leaching (Stopes *et al.*, 2002; Benoit *et al.*, 2015). A wide range of similar properties or functions have been identified by natural scientists as useful in providing quantitative evidence on the impact that different land management decisions might have on the environment, typically at fairly small spatial scales. However, a mechanistic understanding of how different land management alters NC condition and EF is still lacking for some management practices (Smith et al., 2017), with contradictory outcomes observed occasionally in response to changes (e.g. Snapp, Gentry and Harwood, 2010 compared to Williams and Hedlund, 2013). Furthermore, many of these current metrics do not necessarily provide the data required to assess ES values and the impacts to human well-being (Keeler *et al.*, 2012; Duncan, Thompson and Pettorelli, 2015; Smith *et al.*, 2017).

7.1.3 What are the challenges of applying the natural capital approach?

A full application of the NC approach at the farm scale does not, of course, stop at the identification of biophysical measurements of changing NC condition and EF. Rather those measurements must somehow be linked to impacts on ES flows that can be valued. The final step of the approach is not without difficulties. To start, these ES values are lacking for a range of potentially important ES, particularly those derived from biodiversity and the enjoyment of the natural environment (CCI, 2016; Faccioli *et al.*, 2020). Moreover, placing ES values on a number of services can be complex, particularly as the role that EF play in delivering ES can be highly spatially specific (Rodríguez *et al.*, 2006; Bartkowski *et al.*, 2020). Indeed, this spatial specificity can prevent the simple application of value transfer methods (e.g. applying values estimated in different spatial contexts by other studies), typically used in broad scale studies (e.g. Costanza *et al.*, 1997) at the farm scale.

Alongside valuation difficulties it is evident that local-scale applications of the NC approach are resource intensive, requiring data at a resolution that is not currently available in existing datasets (Smith *et al.*, 2017; Faccioli *et al.*, 2020). In addition, the holistic principles behind the NC approach ideally require the quantification of all flows of ES from NC that might be affected by a land management decision, from carbon storage to recreation. Accordingly, practical applications demand a multi-disciplinary approach that demands wide-ranging expertise, across multiple disciplines within the natural and social sciences. It has become widely accepted, therefore, that decision makers currently lack the tools or evidence needed to apply a NC approach to the assessment of management decisions (Guerry *et al.*, 2015; Maseyk *et al.*, 2017; Smith *et al.*, 2017). Indeed, in the recently published 'Enabling a Natural Capital Approach guidance' by the UK Government, it is acknowledged that applications using the available materials may be too "broad-brush" to inform spatially specific land management decisions, highlighting the need for a more detailed appraisal (Defra, 2020a). Such detailed appraisal is likely to require empirical data collection at an appropriate spatial scale and collaboration with a range of specialists and stakeholders.

7.1.4 Applications of the natural capital approach at local scales

Given these complications, most practical applications of the NC approach at the farm-management level have been partial: either only constructing a Tier 1 NC asset register or taking the subsequent Tier 2 step of quantifying or predicting a change(s) in EF (e.g. Silcock and Russ, 2018; Silcock *et al.*, 2018; Ovando, 2020, 2021). Those studies that have conducted an asset register of NC condition have typically focused on the quantity of NC and have not collected data on the quality of NC for most NC assets (e.g. EFTEC, 2018; Silcock and Russ, 2018; Silcock *et al.*, 2018) an undertaking which requires greater resource and expertise. NC condition (e.g. soil condition) can be critical in determining how farm management decisions will impact on EF and ultimately, ES value.

In lieu of practical ES valuation methods, some academic studies have proposed the use of indicators to infer ES delivery in agro-ecosystems, based on measurements of either NC condition or EF (Dale and Polasky, 2007; Williams and Hedlund, 2013). In principle, changes in indicators should signal changes in ES values. Understanding which metrics might be useful in signalling final ES value is, however, still limited and suitable metrics are likely to be spatially specific (Dale and Polasky, 2007). Furthermore, these indicators are not measured in units commensurate with the financial values used to assess the potential market and non-market impacts of farm-management decision-making.

A small number of local studies that have sought to value ES that flow from NC, at the farm (e.g. Dominati *et al.*, 2014) or national park scales (e.g. Faccioli *et al.*, 2019) have not necessarily used the NC approach to evaluate a land management decision. In contrast a small number of detailed academic studies, focusing typically at the field or plot scale, have incorporated measurements of NC condition, some EF and some ES in evaluating specific land management decisions or practices (Sandhu *et al.*, 2008; Porter *et al.*, 2009; Fan, Henriksen and Porter, 2016). These studies, primarily in Denmark and New Zealand, have conducted what is essentially the start-to-end application of the NC approach, however, they have tended to rely on relatively coarse assumptions in order to link NC condition to EF (e.g. earthworm abundance to soil formation) and used replacement cost methods as hard-to-justify proxies for the actual ES values (e.g. the price of top soil if no earthworms were present) (Sandhu *et al.*, 2008). Furthermore, these studies have focused on informing the literature on the merits of different agricultural practices, rather than how measurements of NC, EF and ES can be used by land managers to inform sustainable land management decision making. There remains a need, therefore, to develop a much better understanding of the data and methods needed to execute the NC approach at farm scales in a way that allows the meaningful application to land management decision-making. Until this is achieved the practical application of holistic NC frameworks at management-appropriate scales will be limited.

7.1.5 Study objectives

The aim of this study was to help address these gaps in understanding through a systematic application of the NC framework on the farm scale, quantifying NC condition, EF and ES value for four important ES types: soil carbon storage and climate regulation; crop growth and food provision; nitrate leaching and clean drinking water supply; and pollinator stocks and pollination services. In applying the method to a real farm, the study explores the practical challenges faced when attempting to apply the NC approach to farm-management decision-making. The study tackled the following research questions:

1. What are the data and science requirements of the NC approach when applied at the farm scale? Do these requirements make it practical for routine use in farm-management decision-making?
2. Given the costs and complexities of the full NC approach, can we rely simply on biophysical measurements of NC and EF to assess the likely scale of ES values delivered by farm management decisions?
3. When applying the NC approach, can conversion to organic agriculture deliver greater benefits to humans (ES value) than conventional agriculture?

7.2 Methods

7.2.1 The natural capital approach - methodological principles and ecosystem services

7.2.1.1 Adopting a natural capital-ecosystem services framework

The NC approach framework applied is presented in Figure 7.1. The principles behind the framework are that stocks of NC can be replenished or degraded - the quality and quantity of these stocks, along with various environmental processes, underpins the delivery of ES and this, in turn, has a direct impact on the welfare of individuals in society. Changes to the state of NC as a result of natural or anthropogenic drivers will have a direct consequence on EF and could, depending on spatial and temporal factors, impact the delivery of ES. The framework has been used to select measurements of NC, EF and ES that are considered within the literature to be connected. It has been adapted from Haines-young and Potschin (2008) and is very similar to other frameworks linking stocks, functions and services (e.g. Dominati, Patterson and Mackay, 2010; Keeler *et al.*, 2012)

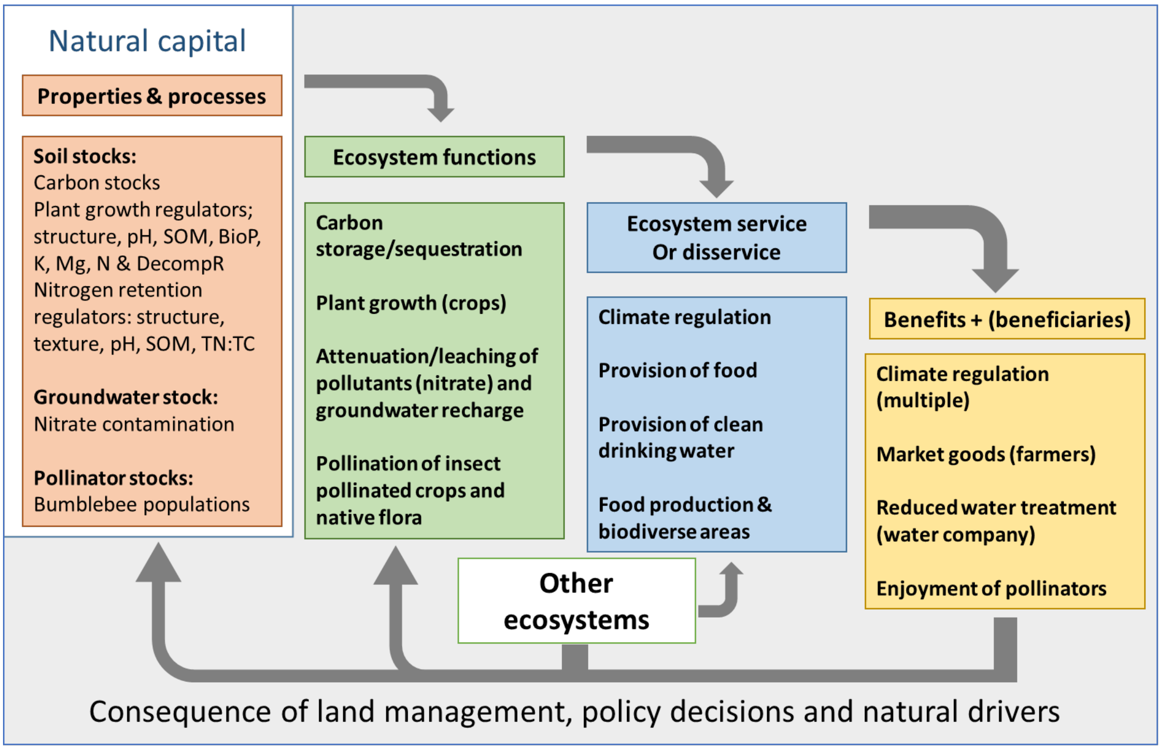


Figure 7.1: Natural capital approach framework applied in the study showing the component parts and how they theoretically link to deliver benefits to humans. Measurements applied in the study are shown and include: 1) soil carbon stocks relating the function of carbon dioxide storage/sequestration and the service of climate regulation; 2) soil NC properties that support plant growth relating to the function of crop growth and the provision of food; 3) soil NC properties associated with nitrogen retention relating to the function of nitrate leaching and the provision of clean drinking water 4) pollinator stocks relating to the pollination of crops and wild plants contributing to food production and biodiverse areas. Framework adapted from Haines-Young and Potschin (2008). Acronyms used: SOM (Soil organic matter), BioP (bioavailable phosphorus), K (potassium), Mg (magnesium), N (nitrogen), DecompR (decomposition of organic material) and TN:TC (nitrogen to carbon ratio).

Four ‘routes’ have been selected to follow through the cascade framework, in order to understand how well measurements at each stage of the framework (NC, EF or ES) are connected and the different conclusions that might be drawn when analysing data on each stage in isolation. Those selected are: soil carbon storage and climate regulation; crop growth and food provision; nitrate leaching and contamination of drinking water supply; and pollinator stocks and pollination services. It is recognised that this is not a comprehensive suite of ES or disservices that arise in agricultural landscapes. Rather than breadth of coverage, the purpose of the study was to examine in detail the steps through the cascade for particular service flows. The importance of the four selected service flows and rationale behind the measurements used in each stage of the NC framework are covered in Chapter 3, Section 3.4.

7.2.3 Introducing the case study

The study was conducted at Clinton Devon Estate in South West England. In 2007, a large part of the estate (the estate Home Farm, ca. 900ha) was converted to organic agriculture, primarily for dairy and arable production. The decision was made on financial grounds but there has been considerable interest since in understanding whether the transition has led to an improvement in estate NC and the enhanced delivery of ES on and off the farm. The framework presented above was applied to the context of the case-study, examining what conclusions could be drawn about the benefits/trade-offs of converting to organic agriculture when assessing different measurements of NC, EF and ES.

A full description of the case-study site can be found in Chapter 3, Section 3.1. In summary, the study has the advantage of presenting a number of different opportunities and challenges in the application of the NC approach to evaluate land management change. These are summarised below:

Change over time: Prior to organic conversion in 2007, the land was managed in a similar way to neighbouring conventional fields, which are on the same soil association (Bromsgrove association [541b]) and within the same landscape setting, experiencing the same climate. Using these conventional sites as a baseline of conditions prior to organic conversion enables an assessment of change in NC, EF and ES as a response to a shift in land management.

Link to water quality: The majority of the converted part of the estate drains to the Otter Sandstone or Budleigh Salterton Pebblebed Heaths Formation and these aquifer units represent a major groundwater drinking water resource in SW

England (Bearcock and Smedley, 2012). Water is abstracted from the groundwater aquifer from boreholes within the study area, treated and pumped for public drinking supply. In 2006, due to increasing concentrations and spikes in nitrate concentrations in abstracted raw groundwater, the water company installed a nitrate anion exchange plant at the local water works. Similar plants have had to be installed across the UK and internationally due to agricultural nitrate contamination (Waterworld, 2015; UK Water Projects, 2015; EFTEC, 2018; Southern Water, 2018). Communication with the local water company enabled an assessment of the costs of water treatment and how these might vary on account of changing nitrate concentrations arising from organic conversion.

Wind pollinated crops: The case-study landscape is dominated by wind-, not insect-, pollinated crops (e.g. maize, cereals and rye-grass clover leys). The yield of these crops is not influenced by pollinator populations and this represents a serious challenge in evaluating the value of pollinator services using common methods (e.g. crop dependency ratios or replacement costs). It is important to note that areas dominated by wind-pollinated crops are not unusual in the UK. For example, in England in 2019, cereal, temporary grass and pasture areas combined cover over 74% of the agricultural land base. In contrast, potential insect pollinated crops (e.g. oilseed, horticultural and other crops) covered just 13.87% (Defra, 2019).

Targeting these opportunities at Clinton Devon Estate the study aimed to apply the complete NC approach to the four selected ES pathways, contrasting a reality in which a large part of the farm estate was converted to organic management from 2007 (referred to as the ‘organic scenario’), to a counterfactual in which the land remained under conventional management (referred to as the ‘conventional scenario’).

7.2.4 Site selection and sampling

For detailed methods on site selection and sampling, please see Chapter 5, Section 5.2.2 for soils and Chapter 6, Section 6.2.1 for pollinators. A summary follows.

7.2.4.1 Soil properties, biomass production and nitrate leaching sites

Nine conventional (con) and nine organic (org) fields were selected for the collection of soil and crop biomass data. A similar number of fields have been used in other studies comparing organic and conventional ES delivery (Sandhu *et al.*, (2015); 10 org and 10 con fields). These chosen fields reflect the main rotational land uses at the case-study location. Three replicate fields for each system were selected that were within the grass ley phase of the rotation, the arable phase of the rotation and transitioning out of the grass phase and into arable. Three replicate sites were selected from each field, informed by pervious sampling results (see Chapter 4) for the measurements of soil properties and biomass yield. 54 soil samples were collected for the analysis of soil carbon stocks, fertility indicators and nutrient retention indicators between November - December 2018, following the methods outlined in Chapter 3 Section 3.6.

Nitrate leaching was determined on a smaller sub-set of six fields (three organic and three conventional) and were matched based on the stage each field was at in the rotation. Further details on site-selection are provided in Chapter 5, Section 5.2.5.5.

7.2.4.2 Pollinator sites

A 3x3 km grid centred on the organic case-study farm office was mapped to define the spatial extent of the BEE-STEWARD outputs. The main crop types were identified, including details of the main crop in 2019 and the proceeding and following crops. Three fields for each crop type (33 fields in total) were selected and a 50m x 2m transect was randomly assigned to each field. Transects were used to count flowering units from which to determine floral resource availability. Fields were selected so that they shared a similar slope and aspect.

7.2.5 Quantifying natural capital condition, ecosystem function and ecosystem service value

7.2.5.1 Soil carbon and climate change mitigation

7.2.5.1.1 Carbon stocks

Carbon stocks (t ha^{-1}) were calculated following Poeplau, Vos and Don (2017) as

$$C\ stock\ = SOC * BD * d$$

where SOC is soil organic carbon (%), BD is the BD of the soil (g cm^{-3}) (corrected for stone content) and *d* is the depth of the soil core (150 mm). BD and SOC were calculated following methods in Chapter 3, Section 3.6.1.

7.2.5.1.2 Greenhouse gas storage

Carbon stored within soil can be calculated in carbon dioxide equivalents (CO_2eq.) - a commonly used metric in measuring greenhouse gas emissions. Carbon stocks were converted to CO_2eq by multiplying carbon storage (t ha^{-1}) by 3.67, the atomic weight of carbon dioxide in relation to carbon (Paustian *et al.*, 2019a).

7.2.5.1.3 Valuation of avoided greenhouse gas emissions

The value of avoided emissions of carbon dioxide (CO_2eq.) was calculated by multiplying stored CO_2eq. by the projected non-traded price of carbon following the guidance of the UK Government Green Book (Hurst, 2019). The prices include a sensitivity analysis depending on climate outcomes and provide figures on a low, central and high price estimate which for 2019 were £34, £68 and £102 per tonne of CO_2eq. The total value for each price sensitivity range, (assuming all CO_2eq. remained stored within the soil) was annualised with a discount rate of 3.5% (following the UK Government Green Book; Hurst (2019)) over an infinite time period to give a final ES value for carbon sequestration in £ per ha yr^{-1}.

7.2.5.2 Plant growth and provisioning services

7.2.5.2.1 Soil fertility indicators

Soil samples were sent off for the identification of soil fertility indicators (lab accreditation; BS EN ISO/IEC 17025.): pH, SOM, bioavailable-P (BioP), Mg, K and particle size distribution (PSD). A sub-sample was retained for the analysis of total carbon (TC), total nitrogen (TN) and carbon to nitrogen ratio (TC:TN). Lab analysis methods are described further in Chapter 3, Section 3.6.1.

Organic matter decomposition rate (as an indicator of biological activity) was determined in the field using the standardised and globally applied Tea Bag Index (TBI) method (Keuskamp *et al.*, 2013). The method is described in full in Chapter 3, Section 3.6.2.

7.2.5.2.2 Biomass production

Biomass production was quantified at each field site for the main season crop, immediately ahead of harvest (or as close as possible) in 2019. The full method is described in Chapter 3, Section 3.6.4.

7.2.5.2.3 Valuation of provisioning services

Producer surplus was calculated by combing saleable yield data, crop market values and production cost information. The detailed method used to calculate producer surplus is covered in Chapter 6, Section 6.2.4 and is summarised below.

Saleable crop yields were calculated using data collected on biomass for each crop in order to determine crop revenue. Fresh weight biomass data was used to calculate organic silage and conventional hay, maize and fodder beet saleable crop yields. As cereal crops are typically sold as grain and straw, a conversion factor was applied to cereal DM forage biomass to calculate these values, as described in Chapter 6, Section 6.2.4[23].

Total crop revenue (£ ha^{-1}) was calculated by multiplying the saleable yield (t ha^{-1}) by the projected market value (Table 7.1). Conventional fodder beet and organic grass silage prices were taken from online marketplace sources and data collected from local agricultural auctioneers.

Table 7.1: Data used on standard yields and market prices taken from industry farm handbooks (or where specified alternative sources) and used in the calculation of crop revenues

Crop	**Market price (t)**		**Std. yield (t ha^{-1})**	**Price/Yield Source**
	Grain	**Straw**	**Grain (straw)**	
Spring triticale (organic)	£220 (feed)	£65	3.0 (3 - 4)	ORC (2017)
Spring wheat (organic)	£235 (feed)	£65	3.2 (3 - 4)	ORC (2017)
Silage (organic)	£50.55 (as bales)		28 (2 cuts)	Online sales/ORC (2017)
Hay (conventional)	£79		8.89 (42 t FW)	BHSMA (2019)/Nix (2018)
Maize (conventional)	£34		40	Nix (2018)
Fodder beet (conventional)	£25		70	Online sales/Nix (2018)
Silage (conventional)	£36.42 (as bales)		47 (11.75 t DM)	Online sales/Nix (2018)

[23] It is important to note here that in the organic system cereal straw and grain would usually be harvested as wholecrop cereal and used as dairy feed. This adds a layer of complexity when considering soil carbon cycling at the farm. Whilst straw is not typically incorporated into the soil on either organic or conventional field sites, on the organic farm this straw is harvested as wholecrop cereal and fed to cows. In turn the cows manure is returned to the field and plays a role in contributing to building soil carbon stocks at organic field sites. This is discussed in further detail in section 7.4.3.2.

Crop management data were gathered for each study field from the participating farmer to quantify the crop cost of production. The method is described in detail in Chapter 3, Section 3.6.5.

Producer surplus was calculated per ha (£ ha^{-1}) for 2019 for each crop in each field by subtracting the total cost of crop production from the total crop revenue. Producer surplus was also calculated per ha using data on industry standard yields (shown in Table 7.1 as "Std. Yields"). Using these data derived from average crop yields is important in considering how producer surplus might vary between different years (i.e. it accounts for variability in crop performance between years due to climate factors).

7.2.5.3 Nitrate leaching and clean drinking water provision

7.2.5.3.1 Nutrient retention indicators

Soil structure (using BD as an indicator), texture (based on particle size distribution), SOM and pH were used as indicators of nutrient retention. Detailed methods for analysis of each can be found in Chapter 3, Section 3.6.1. Particle size distribution provides relevant information on sand (2.00 – 0.063mm), silt (0.063 – 0.002mm) and clay (<0.002mm) components. Clay mineralogy is particularly important in determining nutrient retention. Whilst cation exchange capacity (CEC)[24] is an important measure of nutrient retention quantifying, it was beyond the resources of this study. CEC is strongly influenced by clay and soil carbon content (Calzolari *et al.*, 2016) which are both reported. Soil TN (%) and total nitrogen stocks (t ha^{-1}) are also included in the analysis of soil properties since higher nitrogen levels may increase the scale of leaching. TN stocks were calculated in the same way as carbon stocks: $Nitrogen\ stock\ = TN\ (\%) * BD * d$.

7.2.5.3.2 Nitrate leaching

Field measurements of nitrate leaching provides accurate data on the soil function of nutrient retention. Rates of nitrate leaching in freely draining soils play a critical role in determining groundwater aquifer contamination. Nitrate leaching

[24] Cation Exchange Capacity is the sum of the total exchangeable cations that a soil can adsorb (Brady and Weil, 2008). The CEC plays an important role in agricultural soils as it dictates the exchange of macro-and micro-nutrients with the soil solution, influencing both the efficient uptake of nutrients to plant roots and their potential to leach out of the soil (Brady and Weil, 2008).

was quantified following the method in Chapter 3, Section 3.6.3. The final output was calculated as the total nitrogen (as nitrate – NO_3 - N) leached in kg N ha^{-1} across each of the six study fields (3 organic, 3 conventional) and across each drainage season (2018 – 2019 and 2019 – 2020).

7.2.5.3.3 Valuation of clean drinking water supply

To value changes in nitrate contamination within the aquifer, data were collected on nitrate treatment costs at the nitrate anion exchange plant operated by the local water company. These included material costs (salt to recharge ion exchange cells), total power usage since 2006 and estimated labour hours per month. A brief overview is provided below and more detail on treatment protocols and costs are presented in Appendix E.1. Average annual salt usage was calculated from salt purchase records from 2012-2019 and multiplied by the salt price of £130 per tonne (price provided by water company). Total plant power usage (kwH) was divided by the plant operating years to calculate an average annual energy usage. Energy rates per kwH were taken from the UK Government Department for Business, Energy and Industrial dataset on the Non-Domestic Energy Prices per Quarter for 2019 (BIES, 2020). Annual plant power costs were calculated by multiplying average annual usage (kwH) by the average 2019 energy price for an 'Extra Large Business' (>150,000 MWH) of £0.114 per kwH. Labour employed in managing the anion exchange plant was estimated by company staff to be 1 – 2 days per month, which was costed over three different scenarios: 1 day (8 hours), 1.5 days (12 hours) and 2 days (16 hours). Labour costs were estimated using industry standards taken from the UK Government Office for National Statistics (ONS): Index of Labour Costs per Hour dataset (ONS, 2020). A mean 2019 labour cost per hour of £25.40 for the Electricity, Gas and Water Supply Industry (including Wages and Salaries, National Insurance Contributions, Employer Pension Contributions, Sickness, Maternity and Paternity Payments and Benefits in Kind) was multiplied by annual labour hours to give an estimated labour cost per year. Central labour cost estimates (at 1.5 days per month) were used in nitrate treatment cost saving scenarios.

The costs of constructing the nitrate anion exchange plant in 2006 were not available from current water company staff. Accordingly, cost estimates were drawn from information on similar plants installed in the UK. Summarising data on the installation costs of three different size plants (small, medium, large) a mean cost per m^3 of daily treatment capacity was estimated at £349.41 (SD £77.46). Taking that figure and multiplying by the plant maximum capacity of 3,600m^3 gave an installation cost for the study site plant of £1,257,876 which was rounded to £1.25 million. Installation costs were annualised over 20 years (the plant expected life time) using the government recommended discount rate of 3.5%.

Table 7.2 provides an overview of the costs of nitrate treatment at the local drinking water supply plant.

Table 7.2: Estimated costs of nitrate treatment at the nitrate anion exchange plant local to the case study area. Mid-range operational costs were calculated based on material costs, power costs and the central range estimate for labour costs. T = time over which the capital has been annualised (20 years) and r = the discount rate applied (3.5%).

	Cost	Comment
Capital Cost (£)	£1,250,000	Estimate
Annualised Capital Cost (£/year)	£87,951.35	r = 3.5%, T = 20
Operational Cost (£/year)	£23,437.69	Mid-range
Total Cost per year (£/year)	£111,389.03	Combined

To understand how changes in organic management could impact on nitrate treatment costs, it was necessary to link nitrate leaching inputs in the 'aquifer catchment' to the aquifer nitrate response. A simple model was developed to link potential nitrate leaching inputs (calculated in this study) to aquifer nitrate concentrations. The calculations are explained in detail in Appendix E.2 and a summary of assumptions follow below:

- The 'aquifer catchment' or recharge area - from which water drainage was considered to recharge the aquifer - was derived from the area of Otter Sandstone and Budleigh Salterton Pebble Beds formations that lie in the Lower Otter surface water catchment. Combined, these formations essentially act as one aquifer and share a common water table (Allen *et al.*, 1997).
- The 'aquifer capacity' - or how much the aquifer is likely to be storing at any one time - was taken as the average annual effective rainfall. Identified using MORECS data in Perl *et al.,* (2004) at 427mm. Perl *et al.,* (2004)

identify that the main recharge to the Otter Sandstone aquifer is through rainfall recharge and this aligns with the water company groundwater model data.

- The 'aquifer response time' - or the time it takes for the aquifer nitrate concentrations to respond to leached nitrate at the surface - is at least longer than 10 years, estimated initial response between 10 – 15 years (estimates based on data in Wang *et al.*, (2012) and Wang *et al.*, (2016)). It is an important assumption, as it means that the conversion of 895ha of agricultural land to organic agriculture in the aquifer area in 2007 is yet to have had a significant impact on aquifer nitrate concentrations. It is widely recognised that aquifers take time to respond to changes in nitrate leaching at the surface (Wang *et al.*, 2011) and in another Permo-Triassic Sandstone formation (Eden Valley), Wang *et al.*, (2013) modelled a mean travel time from the surface through the un-saturated zone of 12 years (range = 0 – 61 years).
- Nitrate leaching only occurs across the agricultural area within the aquifer catchment. Whilst it is recognised that other land uses (such as woodland or heathland) can leach deposited nitrogen, these losses are typically much lower than on agricultural land (Herrmann and Pott, 2005).
- Land use has remained largely similar in the aquifer catchment throughout the period of nitrate monitoring.

With these assumptions, it is possible to estimate the response of nitrate concentrations in the aquifer to current nitrate leaching rates and forecast the likely change in these following land management within the aquifer catchment area. It is then possible to estimate the change in treatment costs that those land management changes imply for the water company. Predicting future changes in nitrate treatment costs are necessary in lieu of long-term monitoring to quantify actual response over time.

7.2.5.4 Pollinator stocks, pollination and pollinator services

7.2.5.4.1 Pollinator stocks

The full method applied to model bumblebee populations is explained in Chapter 6, Section 6.2.

Simulations were conducted using the recently published BEE-STEWARD modelling software (Twiston-Davies, Becher and Osborne, 2021). The simulations were conducted separately for two common species of bumblebee: *Bombus terrestris* and *Bombus hortorum*. *B. terrestris* is ubiquitous in the UK and has the most robust documentation on bee and colony behaviour and development for modelling. *B. hortorum* has a longer tongue length and slightly later emergence time than *B. terrestris* and was selected to observe whether these features would have an impact on the suitability of each landscape in supporting species with a different ecology. Each model was run 30 different times to give a distribution of possible colony densities (along with additional data) under the conventional scenario (pre-organic conversion) and the organic scenario (post-organic conversion).

7.2.5.4.2 Pollination and pollinator services

The intention was to utilise the BEE-STEWARD software to identify visitation to crops that benefit from bumblebee pollination. Despite intentions, it became apparent that insect pollinated crops (e.g. field beans, oil seed rape, vegetable crops) are infrequently or never grown within the case-study area. The consequences of this are discussed later in this chapter as it is an important dilemma for NC and ES studies.

7.2.5.5 Farm-scale scenarios

As measurements of crop biomass yield, producer surplus, nitrate leaching, carbon stocks and floral resources (used to inform BEE-STEWARD) were taken at the field scale, it was necessary to scale up measurements to estimate changes across the whole farm. The two contrasting farm scenarios (organic scenario vs conventional scenario) were then used to compare nitrate leaching and changes across the aquifer, the response in bumblebee populations, changes in soil carbon stocks and yield and producer surplus outputs.

Measurements of mean nitrate leaching (kg N ha^{-1}) from organic and conventional field sites were scaled up over 895.20ha (i.e. X kg N ha^{-1} multiplied by 895.20 to estimate annual inputs). 895.20ha is the area of the converted organic farm covering the aquifer catchment. Floral resource data was also scaled up across the majority of the farm area and this process is explained in detail in Chapter 6, Section 6.2. Producer surplus and carbon outputs, however, are only considered for the rotationally managed part of the farm (396.36ha). It was beyond the resources of the study to measure pasture yields, livestock outputs and producer surplus from the remaining area of permanent grazed pasture (498.84ha). The rotationally managed part of the farm was divided into two areas: the area in the grass ley phase of the rotation (191.42ha) and the area in arable phase (204.94ha)[25]. Field data on carbon storage and crop provisioning services were scaled up to the rotational farm area by calculating the mean and standard deviation for CO_2eq storage, economic value of carbon stored, DM biomass and producer surplus under grass ley and arable field conditions. Mean values were then multiplied by the respective area of grass ley and arable across the farm and combined to give total outputs at the farm scale[26].

7.2.5.6 Statistical analysis

Soil fertility, nutrient retention indicators and nitrate leaching data were analysed for significant differences across the two land management scenarios using mean values for each field and applying Wilcoxon-rank/Mann Whitney-U tests in R (R Core Team, 2020). Carbon stocks, carbon values and biomass yields were analysed using linear or generalized linear mixed effects models which were selected depending on the data distribution. Distribution was characterised by plotting data distributions and the use of Shapiro-Wilk tests. Farm system (organic or conventional) was always used as a fixed effect and study field as a random effect. In carbon models' rotation stage (grass-arable, grass-grass, arable-arable) were included as fixed effects to control for potential variance attributed to the phase of the rotation. In biomass production models crop type (grass or arable) was included as a fixed effect to control for variance that could be caused by different crop type. Rotation stage and crop type were not included as random effects, as it is not considered prudent to include random effects that have less than five levels (Harrison, 2015). Producer surplus data were analysed using linear models, using mean data for each field. System and crop type were included as fixed effects. Analyses used the lme4 package in R (Bates *et al.,* 2015). The effect of farm system on bumblebee colony abundance was tested

[25] Areas were taken from actually cropping and ley areas across the organic farm in 2019 but remain fairly constant year on year throughout the rotation.
[26] It worth noting here that the conventional grass ley fields selected were managed for hay and haylage in 2019 rather than silage. Harvest and farm management data for hay was still collected and is presented at the field scale. However, when scaling up to the conventional farm-scale scenario industry figures for conventional silage yield and costs of production were used.

using generalized linear models (GLMs). Here, farm system was used as a fixed effect. All models were fitted with a negative binomial error structure. Zero-inflated negative binomial models were used for *B.terrestris* colony number data*,* following guidance in Blasco-Moreno *et al.*, (2019). All GLMs were run using the glmmTMB package (Brooks et al., (2017) in R (R Core Team, 2020).

Model residuals were reviewed and models were checked for zero inflation and over dispersion to confirm appropriate error structure using the DHARMa Package (Hartig, 2021). Significance was tested at $p<0.05$. All analyses were conducted in R (R Core Team, 2020).

7.3 Results

7.3.1 Carbon stocks and climate change mitigation

7.3.1.2 Natural capital, ecosystem function and ecosystem service value at the field scale: soil carbon

Data on the carbon stocks from conventional and organic field sites are presented in Figure 7.2. Plot C1 shows the measure of NC stock (SOC in tonnes ha^{-1}), C2 shows an indicator of function (sequestered CO2 equivalents in tonnes ha^{-1}) and C3 displays the value of the ES (stored carbon in £ ha yr^{-1}), showing the outcome from the three different carbon price scenarios. Mixed effect model outputs show that organic field sites had significantly higher carbon stocks ($p = 0.008$), significantly higher storage of CO_2eq ($p = 0.008$) and significantly higher ES value ($p = 0.008$). What is clear from Figure 7.2 is that measurements of carbon stocks, CO_2eq sequestration and value follow the same pattern through each tier of the NC framework, with the measurement of the NC condition directly relating to the measurement of ES value[27]. As such, for this ES pathway, conclusions made in the evaluation of soil carbon stocks for each system will be qualitatively identical to conclusions drawn from the evaluation of ES value data using non-traded carbon prices.

[27] Data from a larger soil dataset on the estate (across 225 data points, 16 conventional and 18 organic fields) shows the same pattern and same significantly higher carbon storage under organic field sites. Data from the full dataset is presented in Appendix E.3.

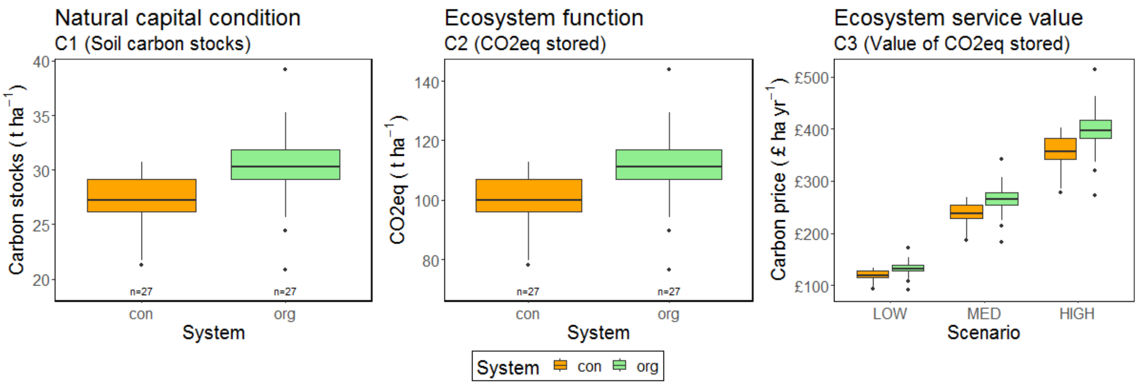


Figure 7.2: Three measurements of soil carbon across the NC approach, from soil carbon stocks (t ha^{-1}) (Plot C1), to stored CO_2eq (t ha^{-1}) (Plot C2), to the value of carbon sequestration (£ ha yr^{-1}) (Plot C3). The pattern of increasing carbon stocks, CO2eq and ecosystem service value under organic agricultural field sites is apparent throughout each plot. With soil carbon stocks directly relating to final ecosystem service value. Dashed lines show the average across conventional field sites with increases from this baseline, i.e. following organic conversion, showing enhanced soil carbon storage. Error bars show standard deviation.

7.3.1.3 Ecosystem service estimates at the farm scale: soil carbon

The potential social benefits derived from soil carbon storage have been scaled up to the farm level (Figure 7.3). Based on soil carbon levels at the arable phase of the rotation and under the grass phase of the rotation, the total CO_2eq stored have been estimated across the farm, considering a conventional baseline scenario and the new organic scenario. Under the organic scenario, the data show an average additional 3681.84 tonnes CO_2eq are stored. Over the 11 years since conversion, this equates to a sequestration rate from 2007 – 2018 of around 334.71 t CO_2eq yr^{-1}.

The additional value of the stored carbon is presented in Figure 7.3 (C5). The additional value from stored carbon under the organic farm scenario is shown in green (i.e. organic CO_2eq minus conventional CO_2eq) for each of the three non-traded carbon price scenarios. These economic benefits range from a mean of £4381.38 per year under the low-price scenario (£34 t CO_2eq^{-1}), to £13,143.69 under the highest price scenario (£102 t CO_2eq^{-1}). As a reminder, these figures relate to the net present value of carbon storage as of 2019 and assume no further annual sequestration.

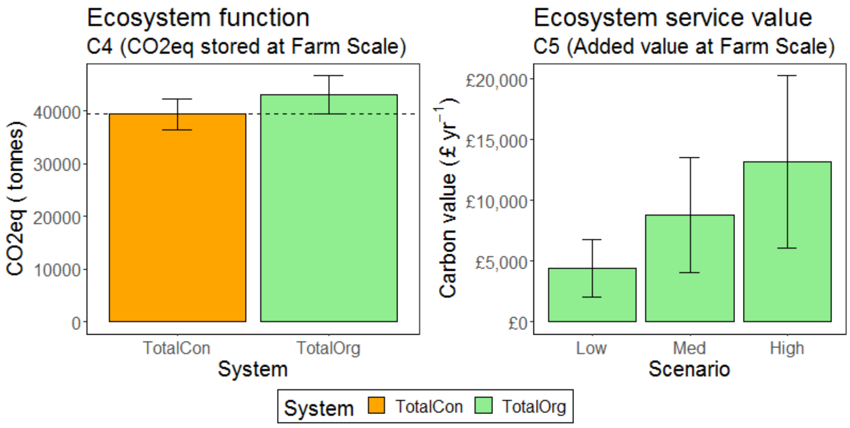


Figure 7.3: Soil carbon storage (as CO_2eq) (Plot C4) and the uplift in economic value of organic agriculture (above the conventional baseline) (Plot C5) at the farm scale. Storage of carbon at the grass and arable phases of the rotation were multiplied across the respective areas of the farm in 2019 to give the results. Under the current organic scenario there is greater carbon storage and this equates to an uplift in value (shown in £ yr^{-1}). Error bars show standard deviation and highlight the variability in carbon values depending on storage within both conventional and organic systems.

7.3.2 Soil fertility and provisioning services

7.3.2.1 Natural capital, ecosystem function and ecosystem service value at the field scale: provisioning services

The data relating to the provision of food and fibre at the field site scale are presented in Figure 7.4. Plot Y1 shows a comparison of the average re-scaled values for each fertility indicator across both systems, Y2 shows the biomass production outputs (tonnes DM ha^{-1}) for each system combining all stages of the rotation and Y3 shows producer surplus (£ ha yr^{-1}). All fertility indicators are scaled 0 – 1, with 1 representing the highest values and 0 the lowest observed in the data, apart from BD which is inverted (i.e. higher BD results in a lower value). Summary data for all fertility indicators is also shown in Table 7.3.

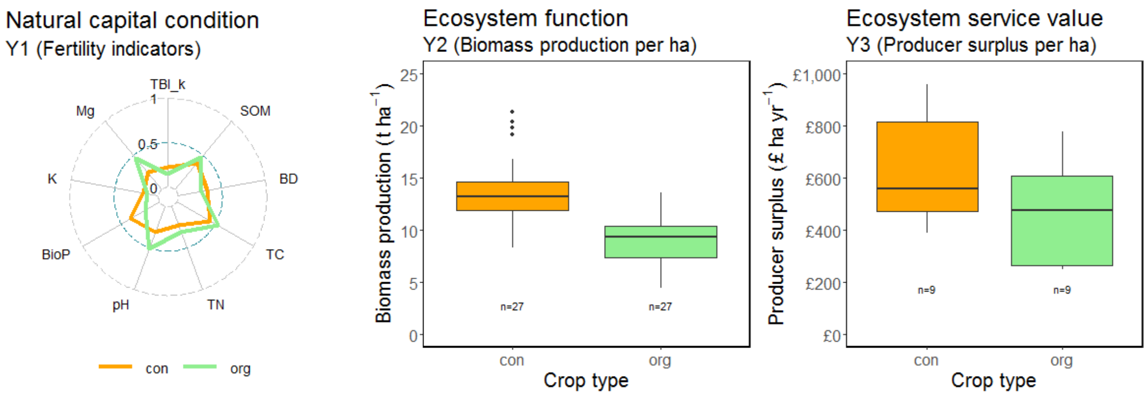


Figure 7.4: Measurements from each stage of the NC approach for provisioning services. Plot Y1 shows measurements of soil NC, displaying means of re-scaled (0 – 1) fertility indicators important in influencing plant growth. BD is inverted with high BD showing as a lower score (i.e. closer to zero). Y2 shows the biomass production (t DM ha^{-1}) including data on grass and arable crops and Y3 shows producer surplus (£ ha^{-1}). Significant differences are observed between DM Biomass production between the conventional (con) and organic (org) field sites but no significant differences are observed for producer surplus, where there is evidently more overlap in the data. Acronyms: TBI_k (decomposition rate), SOM (soil organic matter), BD (bulk density), TC (total carbon), TN (total nitrogen), BioP (bioavailable phosphorus), K (potassium) and Mg (magnesium).

In contrast to the pattern for carbon stocks, it is apparent from figure 7.4 that the measurements used for NC condition (fertility indicators), EF (crop biomass production) and ES value (producer surplus) show different relationships at each tier of the NC approach. Measurements of soil fertility indicators (Figure 7.4 Plot Y1 and Table 7.3) do not show that conventional or organic field sites are systematically 'better' or 'worse' in terms of soil NC condition. Whilst organic field sites had a higher mean SOC, TC and TN, only Mg and pH were significantly different to conventional field sites: p = 0.003 and 0.008 respectively. Whereas, conventional field sites had a higher mean bioavailable-P, K and biological activity (decomposition rate, TBI k) when compared to organic field sites but differences are not significant: p = 0.077, 0.2 and 0.2 respectively. The data do not provide clear evidence that NC condition is systematically improved in relation to producing crops under organic management as compared to conventional management.

In contrast, measurements of EF (crop biomass production) shown in Figure 7.4 (Plot Y2), show a clear difference between the two systems with conventional biomass yields significantly higher (p = 0.002) than organic biomass yields, despite there being no systematic difference across the measurements of soil fertility. Biomass production was consistently higher across all but one conventional field, which had a marginally lower biomass output under maize than the highest performing organic cereal fields. Measurements of EF (crop biomass production) also do not align directly with the measure of ES value (producer surplus). Whilst there is a similar pattern with higher mean producer surplus across conventional field sites (£628.80±200.48 ha^{-1}), compared to organic field sites (£465.30±195.46 ha^{-1}) there is no significant difference in producer surplus between the two systems (p = 0.133). There is considerable overlap in the range of values between the two with producer surplus across conventional fields ranging from £387.79 - £958.34 ha^{-1} for arable and £558.88 – £470.99 ha^{-1} for grass leys and organic field sites from £263.54 - £776.25 ha^{-1} for arable and £247.88 – £697.55 ha^{-1} for grass leys. In contrast to the pattern for biomass production, 66% of conventional fields (6 fields) had lower producer surplus than the highest performing organic cereal field (£776.25 ha^{-1}).

Therefore, the data indicates that there is no simple linear relationship between fertility indicators (NC), biomass production (EF) and producer surplus (ES value). Examination of the data comparing organic and conventional sites at each tier of the NC approach has the potential to result in different interpretation of either systems' capacity to deliver higher provisioning services[28]. Examining just the NC condition data, the conclusion would be that organic management is not systematically better than conventional management. In contrast, interpreting the EF data it would be straightforward to conclude that conventional management was significantly better than organic. It turns out that a detailed assessment of the actual value of food provisioning ES paints a more nuanced picture with a change to organic management delivering value loss in some locations and value gain in others.

[28] Data from a larger dataset on the estate (across 225 data points, 16 conventional and 18 organic fields) shows the same patterns with limited significant difference between an albeit smaller suit of fertility indicators, yet significantly higher biomass production under conventional fields. No significant difference is reported for producer surplus. Data from the full dataset is presented in Appendix E.3.

Table 7.3: Summarising soil data collected across conventional and organic field sites. Statistical analysis on a comparison of the means for each field for each parameter across the 9 conventional and 9 organic field sites. Acronyms: TBI_k (decomposition rate), SOM (soil organic matter), SOC (soil organic carbon) BD (bulk density), TC (total carbon), TN (total nitrogen), BioP (bioavailable phosphorus), K (potassium) and Mg (magnesium).

Soil property	**Conventional, N = 9**[1]	**Organic, N = 9**[1]	**p-value**[2]
BD (g cm^{-3})	1.33 (0.11)	1.38 (0.10)	0.4
SOM (%)	2.66 (0.40)	2.85 (0.39)	0.5
SOC (%)	1.38 (0.21)	1.48 (0.20)	0.5
TC (%)	1.37 (0.20)	1.48 (0.22)	0.2
TN (%)	0.131 (0.021)	0.150 (0.035)	0.2
C:N Ratio	10.55 (0.61)	10.12 (0.97)	0.5
pH	6.07 (0.39)	6.61 (0.30)	0.008**
BioP (mg l^{-1})	32 (16)	20 (8)	0.077
K (mg l^{-1})	124 (57)	99 (66)	0.2
Mg (mg l^{-1})	53 (12)	77 (17)	0.003**
N-potential	10.32 (1.42)	10.04 (1.21)	>0.9
Clay (%)	14.15 (2.45)	14.70 (1.87)	0.6
Sand (%)	67.5 (7.4)	68.8 (5.1)	0.8
Carbon stocks (t ha^{-1})	27.14 (1.94)	30.27 (2.67)	0.011*
Nitrogen stocks t ha^{-1})	2.57 (0.23)	3.08 (0.60)	0.019*
TBI_k	0.0223 (0.0021)	0.0204 (0.0028)	0.2

[1]Mean (SD)

[2]Wilcoxon rank sum test; Wilcoxon rank sum exact test

7.3.2.2 Ecosystem service estimates at the farm scale: provisioning services

Field crop yield data collecting in 2019, industry standard crop yield data from industry handbooks and cost of production data collected in 2019 were used to estimate outputs of crop biomass (EF) and producer surplus (ES) at the farm-scale. This was conducted across the arable (204.94ha) and temporary ley grassland (191.42ha) areas, for organic and conventional cropping scenarios. Figure 7.5 shows outputs of biomass producer surplus based on field crop yield data[29] (Y5) and producer surplus based on industry standard crop yield data (Y6)

[29] Apart from for conventional grass silage biomass production which was taken from industry standards

at the farm scale[30]. Industry standard data are used for comparison (Figure 7.4 Y6) and provide an insight into the likely producer surplus based on average yields (calculated over multiple years). Figure 7.5 shows a similar pattern to that seen in Figure 7.4, with the measure of EF (farm biomass output) different between organic biomass outputs at the farm scale and conventional biomass outputs at the farm scale. The organic farm scenario has a lower biomass output (36.15% lower). In contrast, producer surplus generated under organic management or conventional management is very similar. Whilst mean organic producer surplus is marginally lower than conventional producer surplus, when using both field data (£15,924.50 yr^{-1} or 8.18% lower) and industry standard data (£16,669.10 yr^{-1} or 9.96% lower), the standard deviation around the mean is such that it is difficult to discern with any certainty whether one system delivers greater provisioning ES value than the other. For example, using the industry standard yields, standard deviation is £16,105.80 and £25,301.21 for conventional and organic scenarios, respectively. The variability in crop yield performance and costs of production explain the variance between fields, with the organic scenario showing greater variability in producer surplus than conventional fields.

The estimations at the farm scale further highlight the disparity between measurements of biomass production and measurements of producer surplus. Differences in management costs and higher price premiums under organic management are the primary reason for the change in pattern when moving from measurements of farm biomass production (EF) to farm producer surplus (ES value).

[30] These estimates use the sub-set field data from this study; 9 conventional and 9 organic fields. This differs to the wider dataset used in Chapter 6 and as shown here gives a slightly different impression of producer surplus at the farm scale.

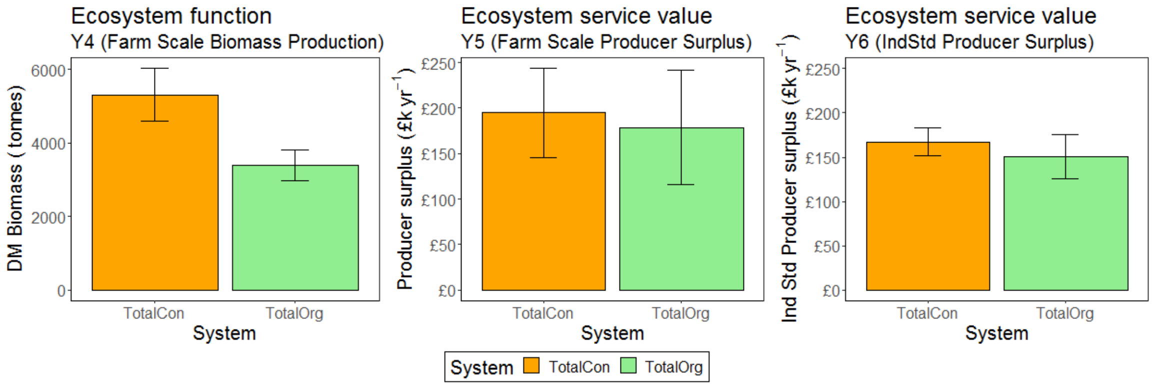


Figure 7.5: Estimated biomass production (Y4), producer surplus (Y5) and producer surplus using industry standard yields at the farm scale. Calculations are based on cumulative output of DM biomass/Producer surplus for grass and arable crops when multiplying them by the respective grass and arable areas on the farm. Error bars show standard deviation. Whilst differences in farm scale crop biomass are significantly different, there is considerable overlap in producer surplus when using primary crop yield data and industry standard crop yield data.

7.3.3 Nutrient retention and the provision of clean drinking water

7.3.3.1 Natural capital, ecosystem function and ecosystem service value from the field to the farm scale: nitrate leaching and clean drinking water supply

Figure 7.6 shows data relating to the field scale measurement of soil nutrient retention indicators (Plot N1), nitrate leaching (Plot N2) and the estimated savings in treatment costs at the farm scale under the conventional and organic scenarios (Plot N3). Nutrient retention indicators are scaled 0 – 1, with 1 representing the highest values and 0 the lowest values, apart from BD which is inverted. Summary data for each indictor are displayed in Table 7.1.

Interpretation of Figure 7.6 (Plot N1) and Table 7.3 suggests that soil properties that could influence nitrate retention or leaching are similar between organic and conventional field sites. Organic field sites had higher mean SOM, clay and TC - all of which contribute to increasing nutrient retention. However, differences were not significant. Organic field sites did have significantly higher nitrogen stocks ($p = 0.019$) and pH ($p = 0.008$) which could suggest greater risk of nitrogen losses. However, higher nitrogen stocks coincide with higher carbon stocks (Table 7.3) and a similar TC:TN ratio between conventional and organic sites. These factors likely suggest that higher soil nitrogen is bound to SOM, rather than being in water-soluble form and vulnerable to leaching. The data do not provide clear evidence that soil NC condition is systematically improved, in relation to retaining nitrate and delivering clean drinking water under organic management as compared to conventional management. Furthermore, as the results show, indicators do not appear useful predictors of EF (nitrate leaching, Plot N2) or ES value (measured as water treatment savings Plot N3). Both of which show a more defined difference in the potential ES output under organic versus conventional management.

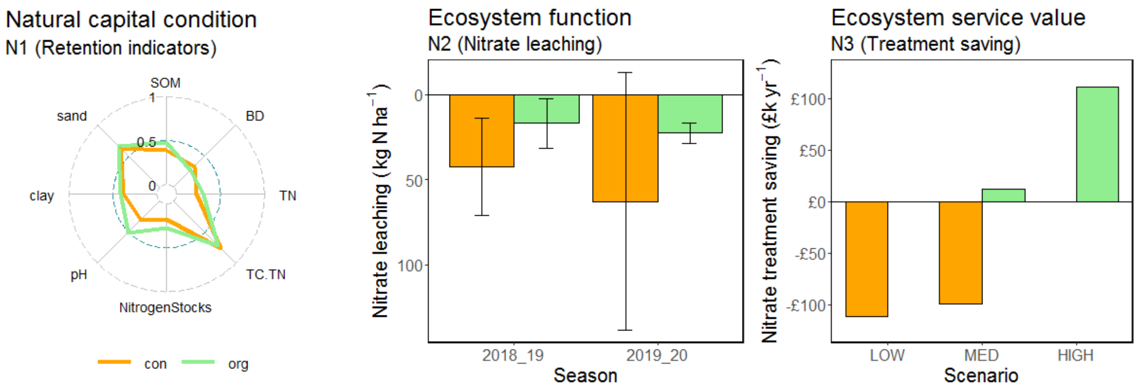


Figure 7.6: Measurements made at each stage of the NC approach for clean drinking water provision services/disservices. Y1 shows the measurement of soil condition indicators considered important in nutrient retention. Y2 shows nitrate leaching across the two sample seasons as a measure of EF. Y3 displays potential costs (minus) and cost savings (positive) associated with Low, Medium and High range scenarios at the water treatment plant scale (i.e. not per ha). Low assumes no response in aquifer nitrate following conversion to organic and no cost savings. Medium assumes reduction in aquifer nitrate concentrations and reversion to only baseline nitrate treatment costs. High assumes reduction in aquifer nitrate concentrations and sufficient water company confidence to decommission the plant.

Measurement of nitrate leaching (EF) show that mean nitrogen (as nitrate) loss across the two sample seasons is higher on conventional fields (mean 52.64 kg N ha^{-1}) than organic field sites (mean 19.85 kg N ha^{-1}) (Figure 7.6 Plot N2). Wilcoxon Rank Sum test results indicate that the difference between systems, however, is not significant ($p = 0.18$). The lack of significance is, in part, due to considerable variability between sites and the restricted sample size. Even so, the ceiling for nitrate leaching losses appears to be much higher for conventional fields, which were as great as 150 kg N ha^{-1}, four times greater than the highest losses under organic management of 33.88 kg N ha^{-1}.

Calculation of nitrate treatment savings (ES value) as a response to conversion (from conventional to organic agriculture), show that there could be a similar pattern to nitrate leaching results, with organic conversion offering the potential to reduce nitrate treatment costs (Figure 7.6 Plot N3). This outcome, however, is contingent on the exact aquifer response to the change in nitrate inputs and the future nitrate treatment strategy utilised by the water company. As explained in method Section 7.2.5.3.2, there is a delay in the aquifer response to nitrate leaching at the surface (initial response estimated at 10 – 15 years). Conversion to organic management in 2007 and associated reductions in nitrate leaching have not yet had a significant impact on the aquifer nitrate concentrations routinely measured by the water company. In lieu of being able to monitor the long-term changes in aquifer nitrate concentration and assess the changes in treatment practices, it is necessary to forecast the potential response to reductions in nitrate leaching. Figure 7.6 Plot N3, therefore, shows the cost savings associated with three potential outcomes that could arise due to reduced nitrate leaching. Table 7.4 shows that there is a strong likelihood that Medium and High range outcomes could occur in the future.

Figure 7.6 Plot N3 identifies no water treatment cost savings under the low range scenario[31], which assumes that, despite changes in nitrate leaching, there is little to no effect on aquifer nitrate concentration and treatment costs remain as usual. The treatment cost savings are moderate under the mid-range scenario[32] (£11,681.58 yr^{-1}), which shows the reduction in costs if the water company were to retain the treatment plant but operate only at baseline level. Under the high range scenario[33], savings are significant (£111,389.03 yr-1), with the reduction in nitrate concentration in the aquifer prompting the water company to decommission the nitrate treatment plant. It is important to note that the three ranges do not relate to different scales of nitrate leaching from conventional or organic land and the model that informs Plot N3 uses the same mean values taken from Plot N2 (52.64 kg N ha^{-1} for conventional and 19.85 kg N ha^{-1} for organic).

Whilst there is uncertainty in the exact aquifer response to reduced nitrate leaching, data presented in Table 7.4 show that organic conversion could have

---

[31] Low range ES value: Assumes no response in the aquifer nitrate concentration due to reduced nitrate leaching from organic agriculture.
[32] Mid range ES value: Aquifer nitrate concentrations fall below treatment thresholds but the water company out of caution retain the plant, which requires constant operation at a baseline level.
[33] High range ES value: Aquifer nitrate concentrations fall sufficiently below treatment thresholds, giving the water company confidence that they no longer require anion exchange treatment and decommission the plant. Management of spikes in nitrate concentrations is reverted to pre-anion exchange plant (2006) methods, blending water from multiple boreholes.

the capacity to achieve Medium and High range water treatment savings. That is that reducing leaching to a mean level of 19.85 kg N ha^{-1} across the organic land management area could reduce nitrate concentrations to below the water company nitrate treatment threshold. The information requirements of Table 7.4 provide an insight into the level of complexity in understanding how nitrate leaching (EF) links to nitrate treatment savings (ES value). It combines information on nitrate leaching, rainfall recharge, aquifer catchment and water company treatment thresholds.

Table 7.4: Data calculations used to estimate the aquifer response to nitrate leaching changes under conventional (baseline scenario) and organic scenarios. Calculation methods and data sources are shown. Expected business as usual concentrations are closely aligned with actual mean nitrate aquifer concentrations (measured by the water company in 2019). Organic conversion, assuming nitrate inputs of 19.85 kg N ha-1 reduces aquifer nitrate concentrations below the nitrate treatment trigger level of 37 mg NO3 l^{-1}.

Component	Value	Units	Calculation method
Aquifer information:			
Aquifer 'area'	81,947,054	m^2	Otter Sandstone and Budleigh PBs area
Rainfall recharge	0.427	m	MORECS data from Perl et al., (2004)
Aquifer 'capacity'	34,991,392	m^3	Aquifer area * rainfall recharge
Agricultural area of aquifer (AA)	5605.55	ha	Agricultural area (AA) from CEH LCM (2019)
Conventional scenario (BAU):			
Nitrogen input per ha	52.64	Kg N ha^{-1}	Field work data (FWD)
Nitrogen input over agri. area	295,076.41	Kg N	AA * nitrogen input per ha
Expected BAU conc. in the aquifer	0.008433	kg N m^{3-1}	Total N input / aquifer capacity
Expected BAU conc. in the aquifer*	37.27	mg NO_3 l^{-1}	Conversion from kg N m3 to mg NO3 l^{-1}
Organic scenario:			
Organic nitrogen input per ha	19.85	Kg N ha^{-1}	Field work data (FWD)
Organic agricultural area (OA)	895.32	ha	Farm data
Remaining conventional area (CA)	4710.24	ha	AA - organic agricultural area
Conventional total N input over area	247,947.00	Kg N	CA * con nitrogen input per ha
Organic total N input over area	17,772.01	Kg N	OA * org nitrogen input per ha
Total input from both	265,719.01	Kg N	Org nitrogen input + Con nitrogen input
Change in nitrate conc.	-3.71	mg NO_3 l^{-1}	New aquifer conc. - BAU aquifer conc.
Expected future conc. in the aquifer	33.56	mg NO_3 l^{-1}	Total N input / aquifer capacity (converted)
*Actual mean conc. in aquifer	37.16	mg NO_3 l^{-1}	Mean of all aquifer borehole samples in 2019
Nitrate treatment trigger level is at 37 mg NO_3 l^{-1}			

Table 7.4 presents a summary of the calculations used to link nitrogen inputs and the potential changes to aquifer nitrate concentrations. Full calculations and justification of input figures can be found in Appendix E.2. Reassuringly, there is good agreement between the estimated nitrate concentrations under a 'business as usual' conventional scenario (study estimation, 37.27 mg NO_3 l^{-1}), with the mean actual concentrations reported in water company aquifer samples in 2019 (water company data mean, 37.16 mg NO_3 l^{-1}). The agreement between the two figures suggests that calculations relating to inputs and aquifer capacity are reasonable. The calculations allow the link to be made between nitrate input and aquifer nitrate concentration and enables an assessment of what happens if nitrate inputs change (e.g. as a response to organic conversion). Table 7.4 shows that organic conversion at the case study site (over roughly 16% of the agricultural area draining to the aquifer) has the capacity to reduce aquifer nitrate concentrations to 33.56 mg NO_3 l^{-1}. This would reduce concentrations below the current nitrate treatment threshold (37 mg NO_3 l^{-1}) by 3.44 mg NO_3 l^{-1}. It is expected that there will be a lag in initial response of aquifer nitrate concentrations (10–15 years) following changes in nitrate inputs. It is likely that it will be longer before significant changes in nitrate concentrations occur. Unsurprisingly, the impact of organic conversion in 2007 is yet to show in current aquifer concentrations. It is reasonable to assume however, based on the data calculations and an understanding that aquifer nitrate concentrations lag behind land management change that cost savings in Figure 7.6 (Plot N3) could accrue in the future.

The results highlight the complexities of not only linking soil condition to nitrate leaching but also nitrate leaching to drinking water contamination and treatment. The relationship between nitrate leaching and the cost of water treatment will be contingent on a suite of different factors, requiring far more data than just measurements on properties or function. The indicators of NC condition, linked to nitrate retention, provide little indication of the likely response in nitrate leaching (EF). Whilst the measure of nitrate leaching (EF) is essential in understanding whether land management change might effect change in the provisioning of clean drinking water (ES), it is only one element in a highly complex calculation needed to determine the scale of that value. The value of the ES depends as much on decisions made in the economic and political world (i.e. regulated water company decisions), as it does to flows from the natural world.

7.3.4 Pollinator stocks and pollination services at the farm scale

The BEE-STEWARD outputs show an increase in pollinator stocks following conversion to organic agriculture. Table 7.5 details how mean colony numbers, colony density, adult bees produced and, where applicable, the time to extinction were all higher in organic compared to conventional simulations. The model outputs suggest that organic conversion at the case-study site significantly increases *B.hortorum* (38% increase in number of colonies, $p = <0.001$). Whilst there is a higher mean number of colonies and adult *B.terrestris* in the organic compared to the conventional scenario, the difference is not significantly different.

Both conventional and organic model scenarios showed better suitability for *B. hortorum* with higher reported nesting densities than *B. terrestris*. For example, mean (± SD) peak colony density for *B.hortorum* in the organic scenario was 59.9 ± 4.6 colonies km^{-2} - over ten times higher than densities for *B.terrestris* - and 4.97 ± 3.20 colonies km^{-2} in the same landscape. Furthermore, multiple *B. terrestris* model runs went extinct (i.e. not a single population was sustained for the 24-year model duration) across both landscape scenarios, although population extinction was far more common in the conventional scenario (only 8 surviving runs out of 30) than the organic scenario (18 surviving runs out of 30).

Table 7.5: Summary data of model outputs for B.terrestris and B.hortorum across the four landscape scenarios. Data summarise mean number of colonies (at the peak during each model run) and adult bumblebees and the mean peak colony density from year 12 – 24, across all 30 model runs. The number of model runs where the bee populations survived are shown (Surviving runs), along with the mean time to extinction for those runs where a bumblebee population was not sustained for the full model duration (24 years).

Characteristic	Organic, N = 30[1]	Conventional, N = 30[1]
Bombus terrestris		
No. colonies (peak)	45 (29)	31 (28)
Colony Density (peak) (km^2)	4.97 (3.20)	3.45 (3.04)
No. bees	1,158,905 (1,058,480)	624,212 (736,889)
Time to extinction	5,282 (2,076)	5,032 (2,084)
No. surviving runs	18	8
Bombus hortorum		
No. colonies (peak)	546 (42)	396 (44)
Colony Density (peak)	59.9 (4.6)	43.4 (4.8)
No. bees	13,765,280 (1,232,258)	9,782,821 (958,092)
Time to extinction	NA	NA
No. surviving runs	30	30

[1] Mean (SD)

The data on pollinators has been presented using the same framework as for carbon storage, provisioning services and clean drinking water supply (Figure 7.7.). Data on bumblebee populations (indicators of pollinator stocks) is presented for each system and for each species in plot B1, showing the increase in populations following organic conversion, as discussed above. In contrast, Figure 7.7 plots B2 and B3 show that despite higher population stocks there is no evidence of crop pollination and therefore no value can be attributed to crop pollination services within the study area. Crops that typically benefit from insect pollination (e.g. oil seed rape, field beans and horticultural crops) were not recorded within the study landscape (which is dominated by wind pollinated crops), preventing modelling of crop visitation and any estimation of the value of crop pollination services. Measurement of pollination of wild plants was not conducted and other service values of pollinators are unknown. Based on the available data, it was not feasible to identify whether the observed increases in pollinator stocks under organic management result in increased ES value.

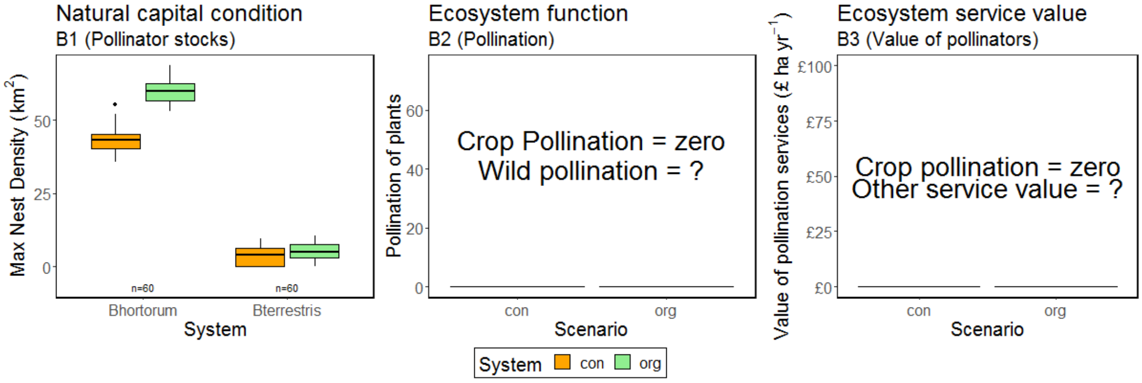


Figure 7.7: Measurements from each stage of the NC approach for pollinator stocks and services. B1 shows max nest densities (km^2) for the two species of bumblebee used as indicators of pollinator stocks. B2 shows the gap in available data on wild plant pollination (not collected in the study) and the lack of crop pollination (as a result of no crops being grown that typically benefit from insect pollination). B3 shows that unsurprisingly there is no value in increased crop pollination services and that other service values are unknown. Despite significantly higher B. hortorum and marginal higher B. terrestris populations under organic management, there is no evidence to support an increase in ES value.

7.4 Discussion

This study presents one of the first studies that has systematically applied the complete NC approach at the farm management scale using detailed field data. The methods and results presented here provide a valuable insight into the data and science requirements needed when conducting the NC approach at the farm management scale – demonstrating the wide range of expertise and resource needed to undertake an analysis of this kind. The results highlight that often biophysical measurements of NC and EF do not necessarily signal the likely scale of ES values delivered by farm management decisions. They show that proving NC condition has "improved" may tell you nothing about whether there is a change in the value of ES flows. The discussion is expanded on below regarding the data and science requirements needed in conducting the NC approach at the farm scale, asking the questions:

1) whether it is practical, given the challenges of collecting this data, for the routine use of the approach in farm-management decision-making?

2) whether, to avoid some of these challenges, it is possible to rely simply on biophysical measurements of NC and EF to assess the likely scale of ES values in response to land management change?

The discussion finishes by presenting the value of ecosystem goods and services measured at the farm-management scale. This showcases how completing the full NC approach can be useful in evaluating land management decisions and addressing the question: can conversion to organic agriculture deliver greater benefits to humans (ES value) than conventional agriculture?

7.4.1. Addressing research question 1: What are the data and science requirements of the NC approach when applied at the farm scale? Do these requirements make it practical for routine use in farm-management decision-making?

This section tackles research question one. Firstly, it identifies the data and science requirements of applying the NC approach for the four selected ES pathways, before highlighting the challenges associated with meeting these data and science requirements. Finally, it discusses whether, based on the findings here, the routine use of the NC approach is practical for farm-management decision-making and/or evaluation.

Figure 7.8 brings together a summary of the data required to generate the results in this study for each of the four ES flow pathways. Data that were successfully collected are shown in black and data that could not be collected are shown in red. A non-exhaustive list of beneficial data is also provided for metrics recognised as also being important indicators of NC condition or measurements EF and could be useful in future studies. The empirical measurements collected in the study are shown for NC condition, EF and ES, alongside the additional data required to understand either environmental processes linked to EF or the value of ES. Figure 7.8 is presented for two purposes: 1) to act as a resource that others can use to understand the data requirements associated with valuing each tier of the NC approach for these four ES flow pathways; and 2) to illustrate the resource intensive nature of analyses that wish to measure NC condition through to quantifying ES value at the farm scale.

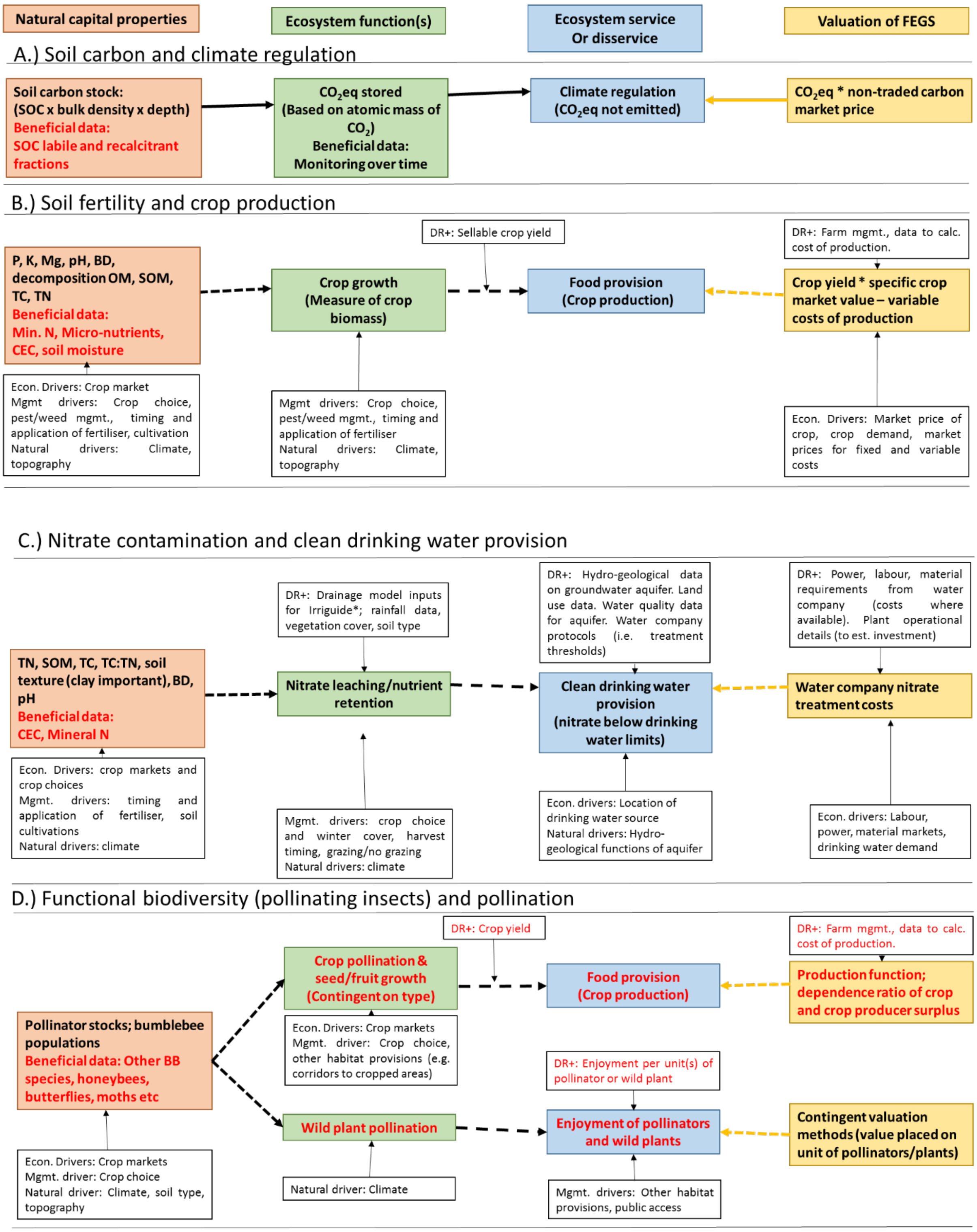


Figure 7.8: The information requirements for measuring NC condition, EF and ES value in the context of A) soil carbon storage and climate regulation; B) soil fertility and crop production; C) nitrate leaching and clean drinking water provision; and) pollinator stocks and pollination services. Block lines show where there is a clear link between NC condition, EF and ES value. Dashed lines highlight where further information and/or different information is required to understand EF and ES value. Red text shows measurements of NC condition, EF and ES value that were not conducted. Coloured boxes represent the core data attributed to NC condition, EF or ES, boxes above marked with DR+ show the additional data requirements needed to

understand EF and ES value and boxes below highlight other drivers aside from NC condition that drive EF and ES value.

Meeting the data and science requirements shown in Figure 7.8 presents a range of challenges. For some ES pathways, the full application of the NC approach is relatively straightforward. Measuring soil carbon and the valuation of the climate regulation service involves a direct transfer from an understanding of carbon stocks through to the value of carbon sequestered. In contrast, measurements of crop production, nitrate contamination and pollination services are more complex, requiring further information, data and expertise. Some of the generalisable data and science challenges encountered during this study, with examples, are discussed below. Further detailed discussion of the challenges specific to each of the four pathways traced through the NC approach in this study can be found in Appendix E.4.

Challenge 1: Selecting the right metrics There is no standardised suite of metrics for applying the NC approach at the farm scale and it is, therefore, necessary to go through the process of selecting appropriate metrics relevant to the study location. Even for relatively simple measurements - such as soil carbon stocks - some studies disagree over the optimum soil sampling depth for measuring changes in carbon storage (Baveye, Baveye and Gowdy, 2016). Some suggest the need to measure both labile and recalcitrant fractions of carbon in carbon storage calculations, distinguishing between those fractions that are more stable within the soil and those that more easily decompose (Yeluripati *et al.*, 2018). Whilst this is not generally the norm in other ES studies, it highlights that even apparently simple measurements of NC condition linked to EF and ES value are open to debate. Advances are required in establishing a standardised suite of metrics that can be applied through the NC approach to evaluate land management decisions.

Challenge 2: Meeting the resource requirements for collecting high-resolution data

Other studies have highlighted that local scale NC approaches have been hindered by the availability of data at a suitable resolution (Smith *et al.*, 2017; Faccioli *et al.*, 2020). The same was identified in this study, which found that even data that might be expected to be available (such as crop yield data) was not at a sufficient resolution[34] to be relevant at the management appropriate scale. Meeting these high-resolution data requirements, as shown throughout this study is undeniably resource intensive. The collection of crop yield data alone, a relatively simple measure in this study, required multiple visits to the field and, for some crops (grass silage), involved multiple visits to the same sites throughout the year to calculate cumulative yield. Furthermore, to understand producer surplus required collecting data on the farm management at each study field, costing each operation and calculating crop revenues. There have been advances in streamlining data collection across some of the metrics used in this study (e.g. automated tractor or combine yield calculators) but much of the data collection suitable for application at the farm management scale remains resource intensive.

Challenge 3: Collecting data over multiple years

Many of the measurements of ES applied in this study would benefit from monitoring over time. This includes monitoring soil carbon to improve the accuracy of the assessment of carbon sequestration, crop yields and costs of production. This would provide more robust estimations of climate regulation, producer surplus and aquifer nitrate concentrations. For example it would enable a transition from predicting water treatment savings to actually measuring them. Monitoring over multiple years would, however, significantly increase resource requirements.

Challenge 4: The cost of data collection and analysis

Given the resource intensity of applying the NC approach at the farm scale it is unsurprising that it is fairly costly. This is an obvious challenge in expanding the application to farm management decision-making. The perceived costs of more advanced soil testing (beyond standard measurement of P, pH and Mg) is already recognised as a barrier to farmers conducting soil health assessments (Briggs and Eclair-Heath, 2017). The costs in this study were most pronounced in the labour collecting and processing the primary data but also included paying for the installation of porous pots and sending soils off for analysis. All ES valuation was

[34] At the study farms where yield data, where it was collected at all, was restricted to arable crops and was only recorded at the field, field group (i.e. a number of fields all with the scale crop) or farm scale (i.e. a mean yield for a crop across the whole farm for that season). This is not uncommon across other farm holdings.

conducted in-house with support from an experienced environmental economist. Obtaining similar expertise privately, as provided by companies such as EFTEC (EFTEC, 2019), is likely to be expensive and beyond the resources available to most farmers.

Challenge 5: Missing data

The data for some measurements of EF - and particularly ES value - do not currently exist. It was beyond the resource capabilities of this study to collect some of these data. For example, in landscapes dominated by wind pollinated crops, valuation of pollination services is impossible by standard methods used for insect pollinated produce (e.g. production function methods). Whilst pollinators have been recognised as being important in pollinating wild plants such as hedgerow fruiting plants, critical in supporting farmland birds (Jacobs *et al.*, 2009), the value of this service is unknown. These wild plants and farmland birds alongside pollinating insects are all likely to be enjoyed by the people who access farmland for recreation. However, there is no transferable data on the value of public enjoyment of farm biodiversity. This study supports others that have noted this lack of data relating to the value of biodiversity. It is a significant stumbling block for many studies applying the NC approach, remaining a fundamental restriction in its holistic application (Faccioli *et al.*, 2020).

Challenge 6: Accessing data and models

Even data that does not require primary collection is not always publicly available. A good example of this is aquifer nitrate data, water treatment practices and information on the materials, power and labour used in nitrate treatment. This information is held by the local water company and required extensive engagement to gather. Even then further data processing was required because the water company did not compile exact costs of labour, power, materials and installation expenditure for the nitrate treatment plant. Similar access restrictions also apply to some of the models used in this study. Whilst access to the ADAS IRRIGUIDE model was kindly provided, with permission from ADAS, through the Wessex Water collaboration, it would typically require a licence agreement to access. Some of the other models, such as that used to map hedgerows for bumblebee landscape modelling, are also not currently publicly available.

Challenge 7: Meeting scientific expertise requirements

Meeting the data requirements shown in Figure 7.8 requires a range of scientific input and expertise. The lead PhD student required interdisciplinary skills in soil, water quality and biomass field sampling, soil and water lab analysis, arable plant and bumblebee identification and survey. They were supported by academic and industry colleagues with extensive expertise in agricultural land management, soil science, catchment hydrology, bumblebee ecology and environmental economics. In addition, further scientific support was required from external stakeholders. For example, undertaking the work on nitrate leaching and nitrate aquifer concentrations required support from water quality and groundwater experts at SWW, to understand aquifer dynamics, current data patterns and treatment protocols. Additionally, water quality experts at Wessex Water provided training in porous pot monitoring and undertook IRRIGUIDE drainage modelling. These expertise requirements are a challenge for the application of the NC approach at the farm scale, where it is evident that collaboration with a range of stakeholders that bring a suite of different scientific expertise is necessary to complete the process.

Given these challenges, is it practical to expand the complete application of the NC approach to decision-making at the farm-management scale?

The attempted quantification of four ES has highlighted the challenges associated with the application of the NC approach at local scales. The study supports another at the local organisational scale, which identified that for smaller spatial scale applications of the NC approach be informative, support is required to supply or collect fit-for-purpose data (Faccioli *et al.*, 2020). Without the collection of primary empirical data in this case-study, there would have been limited capacity to evaluate the effects of land management change. As highlighted previously, data collection is resource intensive and requires a range of expertise. Furthermore, this data collection was for just four ES flow pathways. Following the holistic principles of the NC approach, it would be necessary to at least attempt to incorporate information on the many other ES benefits and trade-offs that could arise following a land management decision (Bateman and Mace, 2020). Therefore, despite the aspirations of those encouraging adoption of the

NC approach, its application at the farm scale is currently largely impractical within the resources and expertise available to most farmers. To facilitate wider application will require advances in and open access to: tools, data and support from a range of specialists at management-appropriate scales.

Furthermore, there is currently limited incentive for land managers to apply the NC approach, aside from them understanding the private benefit flows from NC. At present, there are only limited opportunities in the UK through which land managers might be rewarded for the enhanced monitoring of NC and delivery of public ES. Whilst there are examples of reward schemes to support land managers achieving ES benefits - such as Australia’s Carbon Farming Scheme (Verschuuren, 2017) - these are currently few and far between. There is potential through the new UK Environmental Land Management Scheme (ELMS), which appears to be moving towards a payment for ecosystem goods and services structure (Bateman and Balmford, 2018), that there will be incentives to support land managers incorporating the NC approach into management decisions. Indeed, some farming estates such as Clinton Devon Estate and others are keen to advance their understanding on the application of the NC approach, in preparation for a new ELMS centred around payments for ecosystem goods and services. How this new scheme will unfold, however, remains unclear. If the UK government are serious about, “leading the world in using this approach [natural capital approach] as a tool in decision-making” (pg. 9 UK Government 25 year Environment Plan, Defra 2020) then it is clear that significant work will be required to support the data, resource and support needs of the NC approach within the new ELMS. In doing so, the new ELMS would be ground-breaking in supporting holistic agri-environmental practices that deliver a suite of ES.

7.4.2 Addressing research question 2: Given the costs and complexities of the full NC approach, can we rely simply on biophysical measurement of NC and EF to assess the likely scale of ES values delivered by farm management decisions?

The previous section outlines the depth of information required to understand NC condition, EF and ES value. Given the complexities associated, particularly with deriving ES values, it is recognised that there is a need for a set of metrics that can be used as indicators to signal how changes in land management will directly impact on ES value and human well-being (Dale and Polasky, 2007). Results presented in this study highlight how some relatively simple measurements of NC condition link well with final ES value (i.e. soil carbon stocks and climate regulation). In contrast, other measurements - such as crop biomass production and nitrate leaching - are important but do not directly align with producer surplus benefits of the costs of drinking water treatment. The connections between each tier of the NC approach, alongside which measurements are useful in signalling the scale of ES benefits, are discussed below for each of the four ES.

7.4.2.1 Measuring soil carbon and carbon sequestration

The results on carbon show a simple pattern through the NC framework, with measurements of carbon stocks facilitating a simple calculation of ES value, a fact that has been noted by other studies (Harris *et al.*, 2006; Keeler *et al.*, 2012; Duncan, Thompson and Pettorelli, 2015). In addition, measurements of soil carbon are recognised as offering additional value in terms of understanding soil health (Lal, 2016) and, as such, provide a valuable metric in the evaluation of land management decisions that impact soils.

7.4.2.2 Measuring soil fertility, crop biomass production and producer surplus

In contrast to measurement of carbon, results on soil NC condition offered limited obvious information on biomass yield and producer surplus with few significant differences in soil NC properties between the two systems. Whilst higher bio-available P in the conventional system could have supported improved plant growth, organic field sites were shown to have higher SOM and higher Mg, which are also important for plants. As Figure 7.8B shows there are other drivers influencing crop growth with crop selection being an obvious one (influenced by soil type and system but also dependent on economic drivers) and management practices (e.g. pest control and fertiliser applications). Even when comparing the same crop, studies have also struggled to find links between baseline soil conditions and yields (Miner *et al.*, 2020). Some of these measurements are still useful to land managers (e.g. P, K, Mg, pH), as they are critical in informing fertiliser and lime application rates. However, they were not useful metrics for understanding biomass production or signalling ES value.

Measuring crop biomass production or crop yield, which is common in multiple other ES studies (Smukler *et al.*, 2010; Snapp, Gentry and Harwood, 2010; Gabriel *et al.*, 2013; Williams and Hedlund, 2013; Pywell *et al.*, 2015), is important in understanding crop production. However, as shown in this study, crop biomass or yield is not directly related to ES value (measured as producer surplus). In effect, crop yield is not necessarily a good correlate with the value of crop provisioning services. This was interesting to observe, as it has been identified that farmers and farm advisors often pursue crop yields without thorough investigation of crop gross and net margins (Jarvis and Woolford, 2017). Differences in crop market prices due to premiums paid for organic crops and differences in costs of production meant that, despite significantly higher biomass production in the conventional scenario, there was no significant difference between producer surpluses. This is an important finding as many studies stop short of valuing ES and consider the provision of food based on the quantity of crop yield (e.g. Snapp, Gentry and Harwood, 2010; Gabriel *et al.*, 2013; Williams and Hedlund, 2013; Pywell *et al.*, 2015). Although this may have some relevance when evaluating land management change in the same farming system - for example, comparing different tillage operations on a conventional farm - it does not when comparing different systems. Using only crop yields does not allow a meaningful comparison between outputs of different crops and without considering differences in the costs of production it fails to capture information regarding the actual benefits to the producer.

In studies comparing ES from organic and conventional agriculture, the lower crop yields from organic sites are often considered as a 'trade-off' (e.g. Snapp, Gentry and Harwood, 2010; Gabriel *et al.*, 2013; Williams and Hedlund, 2013). However, this fails to capture information on how people value the crops derived from the two different systems. Given the price premiums paid, some individuals evidently value organic produce more, which is linked to perceived health, nutritional, environmental, taste and welfare benefits (Hoffmann and Wivstad, 2015). These price premiums are not exclusive to organic produce, with other systems (such as pasture fed livestock) also achieving higher prices (Stampa, Schipmann-Schwarze and Hamm, 2020). Future studies comparing ES from different farming systems, where one may obtain a price premium, should acknowledge these market differences when making assertions about provisioning ES without conducting valuation of these goods. In these instances, it is clear that to adequately understand differences in ES value outputs, it is necessary to undertake the complete NC approach, including the valuation of ES.

7.4.2.3 Measuring nitrate leaching and the delivery of clean drinking water

Analyses show that undertaking all stages of the NC approach is also necessary to understand how changes in land management impact on the delivery of clean drinking water. Soil NC condition offered a poor proxy for nitrate leaching. Whilst improved soil conditions such as SOM under organic field sites could have contributed to the enhanced retention of nitrogen (Harris *et al.*, 2006), it is more likely that management drivers played a greater role in nutrient losses - such as crop selection, grazing management and in the second season (2019 – 2020) additional applications of nitrogen fertiliser. Other data, however, such as measurements of soil mineral nitrogen ahead of the drainage season could improve understanding of the potential for losses of nitrogen (Webb, Harrison and Ellis, 2000) and should be considered in future studies. Ultimately, however, to understand the impacts to the delivery of clean water it is necessary to measure, or at least model (Environment Agency, 2021), nitrate leaching. These data are essential in connecting leaching with aquifer nitrate concentrations and understanding how change in leaching affects change in the aquifer.

Whilst measurements of leaching do suggest that one management type (conventional in this case) might be more likely to pollute drinking water supply, it is impossible to identify the scale of which this will affect treatment costs. The impact of nitrate leaching on drinking water provision is spatially specific (is there even a local drinking water supply to pollute?) and depends on the scale of changes in aquifer nitrate concentrations, in response to new land management. In this study it is evident that over the scale of the organic farm (ca.900ha), reducing the inputs of nitrate could generate threshold changes in aquifer nitrate concentrations, circumventing the need for costly treatment. It would be possible for these cost savings to be considered at the 'per ha' scale (i.e. costs of treatment divided by area of farm). However, changes in leaching over 1ha will not generate a proportional change in costs compared to changes over 900ha. This dependence on scale makes an understanding of nitrate losses and treatment costs at one location difficult to transfer to another. Indeed, different aquifers perform differently in terms of nitrate response to leaching (Wang *et al.*, 2016) and different water company protocols are likely to affect the costs of treatment. Additionally, results show that projections are temporally uncertain. Whilst changes in leaching might show an instant response to change in management, when that will start to generate changes in aquifer nitrate levels and changes in nitrate treatment is hard to predict. To this end, nitrate leaching provides valuable information about the potential environmental impacts of land management decisions. However, currently, it is not sufficient to advance understanding of the scale of impact to the delivery of clean drinking water or predict when this impact may manifest.

7.4.2.4 Measuring pollinator stocks and pollination services

Pollinators add additional complications to understanding the connections between NC stocks and ES value. It is clear that whilst it is possible to model some pollinator populations this does not provide any information about the value of pollination services in landscapes dominated by wind pollinated crops. This would require further information on how individuals value the changes in pollinator populations or how they value wild plants. Pollinator population stocks are likely to be a valuable metric for pollination services in landscapes that include insect pollinated crops (field beans, oil seed rape, horticultural crops) . However, until further information on the social enjoyment of pollinators and wild plants is advanced, it cannot be used to predict pollination services in large parts of the UK. That is not to say that these measurements are unimportant and they do advance understanding on the impact of land management decisions on farm wildlife. However, the compatibility of these measurements with the NC approach is restricted unless applied within landscapes with insect pollinated crops.

7.4.3 Addressing research question 3: When applying the NC approach, can conversion to organic agriculture deliver greater benefits to humans (ES value) than conventional agriculture?

This section highlights some of the ES accounting that can be done when completing a full application of the NC approach. The example presented demonstrates how valuation of ESs enables an assessment of the cumulative benefits or trade-offs that might arise under conversion to organic agriculture. The same approach could be used to compare multiple different agricultural land management scenarios. The section first explores whether organic conversion delivers greater ES benefits to humans and secondly, discusses whether the delivery of external ecosystem goods and services is cost-effective, in terms of agri-environment scheme payments. Finally, the discussion highlights some of the uncertainty in the accounting process.

The cumulative benefits derived from each system (scaled to the farm level) and a measure of the change following conversion to organic agriculture, are presented in Table 7.6. Three range scenarios for benefits are considered: 1) the Low range scenario uses the low carbon price data (£34 t CO_2eq^{-1}) and no reduction in water treatment costs; 2) the Central range scenario uses medium carbon price data (£68 t CO_2eq^{-1}) and the medium nitrate treatment cost savings; and 3) high range scenario using high carbon price data (£102 t CO_2eq^{-1}) and maximum nitrate leaching cost savings. All producer surplus data is based on industry standard yields to make figures more transferable across multiple years (i.e. accounting for yield variability). Table 7.6 also includes details of the agri-environment scheme payments available under the current Countryside Stewardship Scheme for organic conversion and management (annualised over 20 years) across just the rotationally managed part of the farm (used to calculate producer surplus and carbon stocks) and over the entire organic farm (used to determine the impact of reduced nitrate leaching).

Addressing the question posed at the start of this section, Table 7.6 shows that yes, conversion to organic agriculture in this case-study, can deliver greater benefits to humans (ES value) than conventional agriculture. However, this is contingent on the potential savings in drinking water treatment costs that could accrue due to changes in reduced nitrate leaching. As explained in results section 7.3.3.1, due to a lag in aquifer response to nitrate leaching, the impact of conversion to organic agriculture in 2007 has had to be predicted. Modelled results suggest that it is probable aquifer nitrate levels will reduce sufficiently below treatment threshold levels delivering moderate (£11,681.58 yr^{-1}) to significant cost saving benefits for the water company (£111,389.03 yr^{-1}). Even with moderate savings at the water treatment works (+£11,681.58 yr^{-1}: the central economic benefit scenario), organic agriculture delivers greater benefit for society which, alongside improvements in carbon stocks (+£8,763 yr^{-1}), offsets the loss in producer surplus (-£16,668.40). Under the high scenario, decommissioning of the nitrate water treatment works (returning to the pre-2006 water management system) would create, alongside improvements in carbon stocks, significant uplift in the benefits derived from organic conversion (+£107,864.33 yr^{-1}). Outcomes from Central and High range scenarios support other studies that show organic agriculture can increase carbon storage (Mondelaers, Aertsens and

Huylenbroeck, 2009; Snapp, Gentry and Harwood, 2010; Gattinger *et al.*, 2012; Tuomisto *et al.*, 2012) and lower mean nitrate leaching (Snapp, Gentry and Harwood, 2010; Tuomisto *et al.*, 2012; Benoit *et al.*, 2014; Biernat *et al.*, 2020). These scenarios align with other studies showing that organic agriculture can significantly increase ES value and reduce the costs of ecosystem disservices (Sandhu *et al.*, 2008; Sandhu, Wratten, Costanza and Pretty, 2015).

In contrast to High and Central scenarios, the Low range scenario does not show an ES value increase under organic management. The Low range scenario is included to show what could occur if nitrate aquifer concentration model predictions are incorrect and provides insight into the implications of organic conversion in a catchment without nitrate drinking water treatment issues. The scenario shows that the lower mean producer surplus (-£16,668.40 yr^{-1}) is not offset by CO_2eq stored (+£4,381.38 yr^{-1}) or savings in water treatment costs. Interestingly agri-environment payments under the Countryside Stewardship Scheme (CSS) for organic conversion and management, as shown in Table 7.6, would more than cover the losses to the farm from reduced crop producer surplus.

Table 7.6: Farm-scale annual economic benefits derived from the ES valued as part of this study: Soil carbon storage and climate regulation, producer surplus (using industry standard yields) and nitrate water treatment savings. Pollination services is included to highlight the absence of ES values. Conventional figures are presented as a proxy for baseline conditions, organic figures show outputs under current farming practice and change shows the differences between the two. Three economic benefit scenarios are presented; Low uses the low carbon price data (£34 t CO_2eq^{-1}) and no reduction in water treatment costs; Central uses medium carbon price data (£68 t CO_2eq^{-1}) and the medium nitrate treatment cost savings; High uses high carbon price data (£102 t CO_2eq^{-1}) and maximum nitrate leaching cost savings. The annual Countryside Stewardship Payments (annualised over 20 years) available for organic conversion/management are included below.

	Con	SD (Con)	Org	SD (Org)	Change
Low range scenario					
Climate regulation (CO2eq stored)	£ 46,837.49	£ 3,560.74	£ 51,218.87	£ 4,277.45	£ 4,381.38
Producer surplus (IndStd)	£ 167,282.20	£ 16,105.80	£ 150,613.80	£ 25,301.21	-£ 16,668.40
Water treatment saving (min)	£ -		£ -		£ -
Pollination services	£ -		£ -		£ -
Total	£ 214,119.69		£ 201,832.67		-£ 12,287.02
Central range scenario					
Climate regulation (CO2eq stored)	£ 93,675.19	£ 7,121.45	£ 102,438.19	£ 8,555.08	£ 8,763.00
Producer surplus (IndStd)	£ 167,282.20	£ 16,105.80	£ 150,613.80	£ 25,301.21	-£ 16,668.40
Water treatment saving (Med)	£ -		£ 11,681.58		£ 11,681.58
Pollination services	£ -		£ -		£ -
Total	£ 260,957.39		£ 264,733.57		£ 3,776.18
High range scenario					
Climate regulation (CO2eq stored)	£ 140,512.90	£ 10,682.51	£ 153,656.60	£ 12,832.07	£ 13,143.70
Producer surplus (IndStd)	£ 167,282.20	£ 16,105.80	£ 150,613.80	£ 25,301.21	-£ 16,668.40
Water treatment saving (Max)	£ -		£ 111,389.03		£ 111,389.03
Pollination services	£ -		£ -		£ -
Total	£ 307,795.10		£ 415,659.43		£ 107,864.33
Annual Countryside Stewardship Scheme payments for organic conversion and management:					
Over rotational farm area only (396.36ha) used for producer surplus/carbon calculations:					£ 32,699.70
Over entire farm area incl. pasture (895.32ha) used for nitrate leaching assessment:					£ 56,400.30

Taking data on CSS payments and comparing them to external ES values (i.e. everything but producer surplus) provides an insight into whether the public investment that supported the conversion to organic agriculture was cost-effective when adopting the ethos of “public money for public goods”. Taking average annual payments (incorporating conversion and area payments over 20 years) these public investments amount to £32,699.70 ha yr^{-1} over the rotationally managed part of the farm (396.36ha) and £56,400.30 over the whole farm area (including pasture used in nitrate leaching modelling; 895.32ha). Under Low and Central range scenarios, using the measurements of carbon, drinking water and pollinators applied in this study the scheme would not appear to be cost-effective[35]. Whilst CSS payments made across the farm offset losses to the farmer in producer surplus (-£16,668.40), there is not a significant increase in the external value of other ES. It is only under the High range scenario, with maximum water treatment cost savings, that the CSS payments would be cost-effective, delivering greater external benefits (£124,532.73 yr^{-1}) alongside improved profitability for the farmer. Here for every £1 spent annually on supporting organic management across the entire farm area, there would be an uplift in external ES benefit of £2.21 yr^{-1}.

The information on cost-effectiveness highlights the benefits of trying to account for market and non-market goods in the evaluation of land management decisions at a policy level. It provides 1) useful information on evaluating the effectiveness of agri-environment schemes; and 2) insight into support structures into the future. It is not the intention of this study to discuss these in detail but based on the data here there is a potential win-win for the environment and for producers following organic conversion supported by CSS payments. Despite this, organic agriculture remains undersubscribed in the UK compared to other European Countries (Scott, 2020) and it is recognised that land managers have been reluctant to convert to organic agriculture on account of concerns over yield volatility (risks of pests and diseases), production costs, losses during transitional period, limited access to technical assistance, certification costs and importantly, either a lack of or difficulties accessing organic price premiums and markets

[35] It is important to note here, however, that carbon storage and producer surplus is only calculated across the rotationally-managed part of the farm (based on data limitations for the pasture area). Accounting for these benefits/trade-offs could change the interpretation of whether low or medium range benefit scenarios would be cost-effective.

(Stochlic and Sierra, 2007; Łuczka and Kalinowski, 2020). The changing state of agri-environment schemes and land-based payments following Brexit to a 'public money for public goods' approach (Bateman and Balmford, 2018) could offer cost-effective payments to organic farmers to deliver many of the existing ES that they already support. Such payments, assuming demand for organic produce continues to remain in a state of growth (Scott, 2020), might offer a greater incentive to conventional farmers to convert to organic agriculture when existing fixed support payments (i.e. basic payment scheme) are scrapped. Consideration should also be given to the spatial targeting of payments made for organic conversion to maximise cost-effective use of public funds. As the findings highlight, the benefit flows from organic agriculture are spatially heterogeneous (e.g. depending on proximity of local drinking water supply and impacts to water quality) and strategic application of organic management will maximise the delivery of ES benefits.

7.4.3.2 Uncertainties associated with ecosystem service value calculations

Some uncertainty exists around the potential of organic conversions to deliver greater benefits to society and, in common with other NC or ES valuation studies (Sandhu *et al.*, 2008; Porter *et al.*, 2009; Fan, Henriksen and Porter, 2016; Faccioli *et al.*, 2020), the figures presented in Table 7.6 rely on numerous assumptions. These assumptions include the principle that mean carbon and producer surplus for arable and grass land use types can be scaled up across the rotational part of the farm, that carbon value and producer surplus are the same over the remaining pasture area and that nitrate leaching is uniform across the agricultural part of the aquifer catchment. There is sensitivity in the prices used to calculate producer surplus and slight changes in both conventional and organic markets and crop performance between the two systems could impact on producer surplus comparisons between years. Further monitoring on the directional changes in aquifer nitrate concentrations and water treatment costs over time, carbon sequestration and inter-annual variability in crop yields, crop prices and costs of crop production, would improve the certainty over the likely outcome of organic conversion.

Improving understanding on producer surplus would, for example, enable a more definite conclusion to be made about whether there is actually a trade-off in producer surplus when converting from conventional to organic agriculture. In common with the results presented in Chapter 6 for landscape scale producer surplus (using industry standard yields), the standard deviation from mean producer surplus is greater than the difference between organic and conventional scenarios. Indeed, results section 7.3.2.2 shows no significant difference between producer surplus from each system. The wide variability in yield, particularly organic yields, suggests that crop performance has a large impact on whether organic producer surplus is greater or lower than conventional producer surplus. Further work at the case-study site over multiple years and field sites would improve the confidence in which system delivers higher producer surplus but it seems reasonable to suggest based on the data here that differences are probably marginal and vary year on year. That is, organic conversion might not necessarily result in a trade-off in producer surplus. Indeed, a number of studies have identified that organic agriculture is often more profitable than conventional farms (Crowder and Reganold, 2015; Röös *et al.*, 2018; Smith *et al.*, 2019; Scott, 2020) (though this does vary between crop type; Scott, (2020)).

Even when making conclusions about producer surplus there are complexities around the assumptions made about the crops being sold off farm and how that might have implications on the carbon cycle on the land. The vast majority of the organic cereals at CDE are used as wholecrop cereals for dairy feed and these products are not commonly traded on markets. In this study the revenue generated from cereal crop production, therefore, had to be estimated on the basis that the grain and straw were actually sold on established markets and transported off farm (Table 7.1). A similar approach is used to value standing cereal crops in Nix (2018). In reality at CDE, whilst organic cereal grain and straw is completely removed from the field, it is then fed to cattle on the farm. The manure from these cattle is then used across the farm to build soil carbon. The assumption that grain and straw is sold and transported off farm is therefore complicated and there is a risk of double counting ES values; with estimates made to calculate producer surplus, including crop material technically meant to leave field sites, potentially overlapping with calculations on the value of soil carbon and climate regulation ES. This limitation highlights the importance of exploring the full carbon cycle across field sites when making assumptions about the value of carbon storage alongside other assumptions on crop sales. This was beyond the scope of this study but it is an important consideration for future research on quantifying the value of multiple ES from agriculture.

Finally, an important principle of the NC approach is that all potential benefits and trade-offs that could affect human well-being are considered. Table 7.6 presents an incomplete set of the ES that flow from agricultural land, failing to quantify others – such as the value of recreational enjoyment, biodiversity, surface water regulation and greenhouse gas emissions from farming practices. There is, therefore, the chance that incorporation of these values (if this was possible) could alter the interpretation of the results. Given the reported benefits of organic agriculture on biodiversity (Tuck *et al.*, 2014), however, it is likely that incorporation of these values will further validate the argument that organic agriculture can enhance ES delivery.

7.5 Conclusion

Whilst the NC approach holds out the promise of being the long-awaited panacea to sustainable land management decision-making, this study shows that currently its complete application is likely to be impractical for most farm managers. The study has highlighted that the information requirements needed to apply the NC approach at the farm-management scale are high. These needs are not met by existing datasets and currently require primary data collection to be of practical value to land managers making or evaluating decisions. Meeting these data requirements has various challenges including selecting meaningful metrics, accessing data and models and having the significant resources needed to cover long-term data, expertise and cost requirements. For the approach to be practically applied at the farm scale, multidisciplinary support networks will need to be developed which incorporate input from natural scientists, economists and other stakeholders, alongside advances in and open access to high-resolution data and tools.

Given the challenges associated with the complete application of the NC approach (i.e. incorporating measurements of NC condition and EF to understand ES value), it is attractive to consider whether the direction or scale of ES value change can be inferred using biophysical NC and EF indicators. This study found that it is often impossible to rely on biophysical measurements of NC or EF alone to inform the likely scale of ES values. Despite advocates supporting the monitoring of NC condition in response to land management change, the data presented here shows that proving NC condition has "improved" may tell you nothing about whether there is a positive or even any change in the value of ES flows. In most cases, it is necessary to collect additional data in order to understand the change in ES value and in some cases, particularly for ES linked to biodiversity, the data is currently not available (e.g. how people value seeing pollinators or the plants, birds and other farm wildlife that pollination supports).

Whilst the study has highlighted the complexities of collecting data at each tier of the NC approach, if the end goal is to understand ES value flows under a change in management there are some metrics that could streamline this process. These include measurments soil carbon (%) and BD to calculate soil carbon stocks and better understand climate regulation services and crop yields, revenues and the costs of production to measure benefits from food provisioning services. These metrics are relatively straightforward to measure and could be collected by farms to monitor the delivery of public and private benefits. Data on nitrate leaching alongside information on aquifer nitrate concentrations and water treatment costs allowed an understanding of the drinking water provisioning services but were considerably harder to collect. The BEE-STEWARD software did not advance understanding on the value of ES linked to pollinators in this case-study, though it would certainly be useful in the context of landscapes with insect pollinated crops.

Despite the challenges faced in completing an application of the full NC approach to farm management decisions, there are some clear advantages to pursuing such a goal. Collating the necessary data to understand the flows of ES values allows academics, policy makers and farmers to answer some interesting questions about different land management options. In this case-study, it allowed an evaluation of whether conversion to organic agriculture at Clinton Devon Estate had the capacity to deliver greater ES benefits to humans. Furthermore, it enabled an assessment of the cost-effectiveness of organic conversion in delivering external benefits, when considering agri-environment scheme payments. Whilst there are some uncertainties in this work, the study presents one of the first known examples of accounting for the change in multiple ES values in the UK when comparing organic and conventional agriculture at a farm scale. Showing that organic agriculture could have the capacity to enhance flows of ES and, under the highest benefit scenario for every £1 spent to support organic management, an additional £1.21 yr^{-1} would be delivered in external benefits. The research provides a foundation that can be further developed by practitioners to refine its application in the future.

Chapter 8: Synthesis and conclusions

This chapter synthesises the conclusions presented in chapters' 4 to 7. It summarises the key findings and the novel contributions that this research makes to building understanding on: 1) the farm scale application of the NC approach; and 2) the impact of different land management practices on NC, EF and ES.

In conducting this research the project has compiled a valuable suite of information for Clinton Devon Estate, Westcountry Rivers Trust and South West Water. Critically, it has established baseline NC conditions which the estate intend to use in future monitoring and it has identified that the estate's decision to convert the Home Farm to organic agriculture has enhanced a number of NC conditions and ES. It appears that whilst conversion to organic agriculture has reduced yields, it has not significantly impacted producer surplus, validating the estate's initial decision to convert to organic agriculture primarily on financial grounds. Furthermore, the research has provided valuable data on the potential reduction in nitrate contamination that could arise at South West Waters' treatment works and has enabled communication with local farmers on the importance of land management practices that reduce degradation of soil, water and biodiversity NC.

The key findings of the research are expanded below, with reference back to each of the four overarching objectives that were established with Clinton Devon Estate at the initial stage of this PhD. Additionally, this chapter considers the relevance of these findings to the government's recently published (2nd December 2021) Sustainable Farming Incentive (SFI) policy paper (Defra, 2021). The chapter finishes by summarising important directions for future research.

8.1 Objective one: To establish baseline natural capital conditions for soil, water and biodiversity natural capital at the farm scale

Baseline NC conditions were established in chapters 4, 6 and 7 for soil, groundwater and pollinator resources and modelled bumblebee populations. An important finding in establishing baseline NC conditions was that, in most cases existing data was either not publicly available or was at an insufficient resolution to be meaningful in farm scale decision-making. Chapter four highlights the issues with using currently available existing soil data to categorise baseline soil NC conditions, such as over or under estimating soil carbon stocks. Chapter 6 shows that despite some existing data (e.g. on standard farm habitat floral resources), field collected data (e.g. on floral resources in organic fields) was critical in informing a local understanding of pollinator resources and bumblebee populations. Chapter 7 demonstrates that whilst some useful baseline data is available (e.g. groundwater quality), it is often held privately, requiring agreement to access it.

In bringing this information together, this study presents one of the first farm scale NC studies where primary data has been combined with existing data to categorise baseline NC conditions. It is one of the first studies to apply the recently published BEE-STEWARD software (Twiston-Davies, Becher and Osborne, 2021) to estimate baseline pollinator resources and bumblebee populations at the farm scale. In doing so, it has provided additional floral resource data which can be used by future studies (i.e. on habitat plots and margins, organic arable crops and pasture). Improving the reference database used in the model will help academics, land managers and other stakeholders streamline applications of the software at other sites. The baseline NC conditions also present an important reference point for future monitoring.

A key finding from Chapter 4 is that whilst establishing baseline NC condition at the farm scale can be resource intensive, the data can be used to build an understanding of the drivers of NC condition. When combined with data on field management practices, it was possible to identify some key drivers of baseline soil NC condition. This is discussed further in the next section. However, it highlights the importance of baseline NC data not just for future monitoring but also for gaining rapid insight into what might be degrading or enhancing NC. The approach presented in chapter four can help guide other land managers, interested in establishing baseline soil NC conditions, in future decision-making and establishing research priorities.

It is important to acknowledge here that this study did not categorise baseline NC conditions for all forms of NC on the estate. Neither did it monitor NC across every field or farm on the estate. Addressing this challenge was beyond the resources available to the PhD which instead focused on a defined area of the estate which spanned one soil association, drained directly to the groundwater aquifer and covered the main spectrum of farm management practices on the estate. The intention was to provide the robust baseline data needed to underpin meaningful interpretation of NC condition and to be of use in addressing other research objectives. In doing so, it has provided both a foundation to tackle objectives two and three and has established metrics and methods that can be applied to other parts of the estate in the future.

8.2 Objective two: To build understanding on how land management practices and intensity impact natural capital condition and productive output

Clinton Devon Estate expressed a strong interest in understanding the impact of different land management practices, frequently applied across the estate, on NC condition. They were interested in the impact of farming intensity on their in-house and tenant farms and on the relationship between productive output and NC condition.

In tackling this objective, the study has provided valuable local evidence for the estate to communicate with tenants about the implications of intensive land management practices. The findings are likely to help guide further investigations into the benefits of reduced tillage and the use of longer-term grass leys in arable rotations. In addition, findings provide evidence to validate the need for policy incentives that sustain and enhance soil NC.

Chapter 4 shows that farm management intensity had a significant impact on soil carbon stocks, soil stability (N-potential (clay:SOC ratio)) and crop biomass yield. Increasing intensity coincided with decreasing soil carbon stocks and soil stability but increasing crop biomass production. Despite arguments made about the negative private impacts of degrading soil NC condition (i.e. reduced productivity), this was not observed in the study. No significant relationship was found between any of the measured soil properties and crop biomass yield. Disentangling the relationships between soil NC condition, inputs of manufactured capital and crop yield are undeniably complex but without evidence that degraded soil conditions (e.g. soil carbon) impact farmers, there appears to be limited private incentive for them to reduce intensive practices. The findings, therefore, validate the need for external mechanisms - such as the recently released SFI ‘Soil Standards’ - to incentivise farmers to protect elements of soil NC that deliver public benefits (e.g. carbon storage and climate regulation).

The SFI policy paper was released on the 2nd December 2021 and is the first of three new environmental land management schemes to be rolled out in England from 2022. The 'Soil Standards' are integral to the SFI, with introductory and intermediate payments becoming available to farmers to support the monitoring of SOM, establishment of soil management plans and the protection of soils over winter. It will be important to monitor whether these protection practices go far enough to protecting soil NC condition. Whilst protecting soils over the winter will likely reduce soil erosion risk, it will not necessarily improve the storage of soil carbon. The most intensive farm in this study (Farm 1 in Chapter 4), for example, consistently ensured winter cover with either autumn sown crops or a fast-grass winter ley, satisfying the requirements of the introductory soil standard. The soil carbon stocks at Farm 1, however, were the lowest across all farms with evidence of degraded soil carbon following the categorisation presented in Prout *et al.*, (2020). The findings suggest the need for an 'Advanced standard', which the government are working towards adding in 2023. They suggest that this will include support for no-tillage techniques which aligns well with the results presented in Chapter 4. There the research identified that the most likely drivers of soil carbon degradation are short-term rotations and frequent changes in cropping (necessitating primary tillage). Fields that had received less primary tillage operations over the six years prior to sampling and those that had been undisturbed for longer periods of time (e.g. long-term grass leys and pasture), stored more carbon and had a higher n-potential (clay:SOC ratio), suggesting greater soil stability. The findings support other literature on the benefits that arise from reduced tillage (Busari *et al.*, 2015; Büchi *et al.*, 2017; Haddaway *et al.*, 2017) and paddock grazed pasture (Whitehead, 2020).

In summary, the research highlights the importance of getting the new SFI policy right. Its success will be important in restoring and sustaining NC conditions and avoiding the detrimental impacts that could arise under further intensification of agriculture following the loss of the Basic Payment Scheme in 2028 (Helm, 2017; Arnott *et al.*, 2021).

8.3 Objective three: To explore the capacity for organic agriculture to balance food production, producer welfare and the enhancement of natural capital and ecosystem service delivery

Objective three is tackled in chapters 5, 6 and 7, all of which show that organic agriculture can deliver a number of environmental benefits (increased soil carbon stocks, increased floral resources and reduced nitrate leaching) and depending on the location of these, could result in a significant increase in the value of external ES. Organic crops had significantly lower crop yields but there were no significant differences detected in producer surplus; that is, whilst benefits in NC and ES came at the expense of crop production, this did not translate into a trade-off in producer welfare (measured as producer surplus).

It is important to acknowledge here that despite improvements in soil NC, pollinator resources and bumblebee populations, not all measurements of NC and EF were systematically improved under organic management. The findings from Chapter 5 also highlight how improvements in soil conditions (e.g. carbon storage and soil structure) could be made in both organic and conventional systems. Furthermore, the study highlights that the scale of the benefits derived are spatially specific and can be dependent on the extent of the area under organic management. For example, the level of those benefits can depend on the types of cropping in a landscape (e.g. insect or wind pollinated crops) and the impact to important NC assets (e.g. the presence or absence of local drinking water resources). Even so, in addressing objective three, this study provides one of the first UK studies to have quantified differences in NC condition, EF and ES values under organic compared to conventional management. In doing so, it has provided compelling evidence that spatially-targeted organic conversion could contribute to improving the flow of ES in the UK, whilst not adversely affecting producer welfare.

Chapters 6 and 7 identify that producer surplus was similar under organic and conventional management, an observation that is explained through the price premium commanded by organic produce and, in some cases the lower costs of organic production. Significant improvements in producer surplus under organic management, however, were only really observed when including subsidy payments from the existing Countryside Stewardship Scheme. Without this government support (as chapter 6 highlights), it is unlikely that farmers will take the risk of converting to organic production. The government have only recently acknowledged that they are reviewing how to reward farmers for delivering environmental benefits through organic management and/or conversion in the new SFI (Soil Association, 2021b). Findings presented here provide evidence that support for organic management (particularly, spatially-targeted support) or, at the very least, support for management practices typical in organic systems (e.g. longer term fertility building leys), should be included in the new SFI scheme.

8.4 Objective four: To undertake a complete application of the natural capital approach (from measurement of natural capital condition through to economic valuation of ecosystem services) using field-based data and, by so doing, build understanding as to how the approach might be implemented at the farm scale and assess whether it is suitable for routine land management decision-making at that scale.

Objective four was addressed in Chapter 7, building upon the data collected as part of chapters 5 and 6. In addressing this objective, it is the first known UK study to incorporate primary field data in the completion of the full NC approach (quantifying NC condition, EF and ES value) at the farm scale. In conducting this work it is also the first known study to attempt to link changes in nitrate leaching (in response to genuine land management change) to groundwater aquifer nitrate concentrations and the subsequent implications for water treatment costs.

The study adds to recent evidence (Faccioli *et al.*, 2020) highlighting that, despite interest in the application of the NC approach at local scales, the data, tools and resources are generally lacking to facilitate its routine use in land management decision-making. If the UK government are serious about realising their ambition of “leading the world in using this approach as a tool in decision-making” (HM Government, 2018, pg. 9) at management-appropriate scales, they will need to significantly improve the resources available to land managers. Resources need to include multidisciplinary support networks which incorporate input from natural scientists, economists and other stakeholders, as well as advances in and open access to high-resolution data and tools. Tools that could offer particular value are those that facilitate the rapid processing of bio-physical and farm management data into ES value data (e.g. converting SOC data or crop yield data into values for climate regulation and producer surplus).

The research has also contributed to learning on the implications of using 'indicators' to signal changes in ES in partial applications of a NC approach. In collating and analysing data at the NC, EF and ES value tiers of the NC approach, it is evident that conclusions made when interpreting NC indicators might differ both quantitatively and qualitatively from those when interpreting ES values. Concluding that NC condition has "improved" may not explain whether there is any change in the value of ES flows. Measurements of EF (such as nitrate leaching or crop growth) are important in working towards valuing ES but drawing inference based on EF alone does not necessarily lead to the same conclusions as when completing ES valuation. For example, the measurement of crop yield (a commonly used indicator of provisioning ES) does not provide all the information required to evaluate a change in producer welfare (i.e. benefits to farmers). A measure of the profit derived from the production and sale of the crop is required to achieve this. Therefore, whilst it is an attractive option to use bio-physical indicators of NC or EF to make predictions about ES value, this is often not a reliable method. The data supports other studies that have highlighted the need for both biophysical and economic metrics to evaluate ES flows from NC resources (Keeler *et al.*, 2012).

Despite the difficulties of using indicators in the NC approach, there are some metrics that are relatively easy to collect and provide a good foundational data-base for land managers. Notably: SOC and BD to estimate carbon stocks and climate regulation services[36]; and crop yield and cost of production data to calculate producer surplus. Whilst it was originally assumed that crop yield data would be collected as standard at the estate, such data was not available at management-appropriate scales (i.e. at the field or field group scale). Data at this scale is important to understand field and crop performance and evaluate how changes in field management are likely to alter changes in soil carbon and producer surplus. It is positive to see that under the new SFI, farmers will be required to measure SOM. Such measurements will provide a good baseline on

[36] Whilst some measurements of SOC and BD are relatively easy (compared to other metrics to assess NC and ES) there is still a debate over how these properties should best be quantified, with some researchers arguing for more resource intensive and robust methods of analysis. This is discussed in more detail in Section 7.4.1 Challenge 1 and Appendix E.4.

which to evaluate change in SOC but, as highlighted in Chapter 5, this will need to be combined with some measure of BD to estimate contributions to climate regulation ES.

These easy to measure ES should still be used with caution, however, as their exclusive use would fail to fulfil the holistic principles of the NC approach. For example, in Chapter 4 it was observed that the highest soil carbon stocks were under permanent organic grassland: a potential win for climate regulation services. In Chapter 6, however, it was observed that the same permanent grassland sites offered no significant difference in floral resource availability compared to conventional grassland sites and much lower floral resource availability compared to weedy organic arable fields. In this case, the two different management types (organic arable and organic pasture) support the maintenance of different NC stocks. Therefore, the focal point of any land management-based scheme needs to be carefully thought out to avoid such consequences. The finding highlights the importance of the holistic nature of the NC approach in avoiding biodiversity losses in pursuit of other public services (CCI, 2016).

8.5 Further work

In addressing the four objectives, this project has advanced understanding on the impact of different land management practices and on the application of the NC approach at management-appropriate scales. It has presented metrics and methods that could be taken forward in other applications of the NC approach. This study does not, however, represent an end point in this research arena. Further work is required to tackle the challenges faced in applying the NC approach and building understanding of how different land management practices impact on ES flows. A summary of further research suggestions is provided below (further details are found in the discussion/conclusion for each chapter).

Long-term monitoring of NC, EF and ES values at the case-study site would improve confidence in the conclusions made around the impacts of organic conversion and different land management intensities. Notably, further monitoring of soil carbon, crop yields (over a variety of growing years) and aquifer nitrate contamination and water treatment costs would improve understanding at the study site, on the role that conversion to organic agriculture has had on enhancing the delivery of ES. The outlook for on-going collection of some of this data looks promising with Clinton Devon Estate intending to continue to monitor soil carbon levels across the study fields, having joined the CASH (Carbon Assets for Soil Health) project being led by the Soil Association. Westcountry Rivers Trust are also continuing to monitor nitrate leaching in response to land management at the field sites installed in this study.

There is a need to build understanding on the application of the NC approach across a suite of different management scenarios, across different soil types and across landscapes with differing scales of heterogeneity. In the context of organic agriculture, this will help improve understanding on the scale of benefits that can be expected to be achieved under organic management in the UK. It will also advance understanding of the application of the NC approach and wider testing will help identify other spatially-specific ES flows and the metrics that can be used to quantify change in them. Other case-studies should therefore be developed that incorporate primary data on NC condition, EF and ES values across the UK. Encouragingly, two geographically-distinct farming estates have already expressed an interest in applying the NC approach framework presented in this thesis to explore the benefits and trade-offs of adopting regenerative agricultural practices.

Incorporating other measurements of NC, EF and ES value will be important in the future in order to fulfil the holistic requirements of the NC approach (Faccioli *et al.*, 2020), particularly those that are harder to measure: flood protection benefits, landscape values and biodiversity. Further research would be useful, for example, to refine understanding on how individuals use and value farmland landscapes in a given state. It would be interesting to establish whether individuals derive more welfare benefit from exploring the public rights of way that intersect an organic or conventional farmed landscape.

8.6 Concluding remarks

Despite the challenges of applying the NC approach at the farm scale, this study shows that the approach has some clear advantages in evaluating farm decisions. Building towards the final valuation of ES has enabled the comparison of the flows of ES benefits from organic compared to conventional management.

Incorporating these economic values alongside financial costs (as presented in Chapter 6 and 7) enables policy makers and land managers to evaluate the cost-effectiveness of different schemes. Using such cost and benefit data could become increasingly important for: 1) academics and policy makers to communicate with land managers about the private and external benefits of their actions; and 2) land managers to communicate the value of their NC assets and the costs of sustaining these to private and public funders. There does, however, remain a long way to go, with improved capacity for data collection and future developments in methods and tools to support applications at the farm scale. It's complete or even partial application is therefore unlikely to be practical for most land managers at present. Furthermore, its development into a powerful support tool will be contingent not only on the challenges being addressed but also on there being a clear incentive for land managers to incorporate measurements of NC and the value of ES in their decision-making frameworks. It will be interesting to observe whether the incentive become stronger under the new ELMS scheme and as interest from private organisations in investing in farm NC (e.g. carbon credit schemes) develops.

References:

ADAS (2005) 'ADAS Porous Pots Procedure (4th Edition)'.

ADAS (2021) 'Crop and Water Use: Irriguide Drainage Model'. Available at: https://adas.co.uk/services/crop-water-use/ (Accessed: 29 December 2021)

Adhikari, K. and Hartemink, A. E. (2016) 'Linking soils to ecosystem services - A global review', *Geoderma*. Elsevier B.V., 262, pp. 101–111. doi: 10.1016/j.geoderma.2015.08.009.

AHDB (2020) 'Nutrient Management Guide (RB209)', (February).

Akaike, H. (1973). 'Information theory and an extension of the maximum likelihood principle'. In B. N. Petrov & B. F. Csaki (Eds.), Second International Symposium on Information Theory. pp. 267–281. Academiai Kiado: Budapest.

Allen, D. J., Darling, W. G., Davies, J., Newell, A. J., Gooddy, D. C. and Collins, A. L. (2014) 'Groundwater conceptual models : implications for evaluating diffuse pollution mitigation measures', *Quarterly Journal of Engineering Geology and Hydrogeology*, 47, pp. 65–80. doi: http://dx.doi.org/10.1144/qjegh2013-043.

Andersson, E., Nykvist, B., Malinga, R., Jaramillo, F. and Lindborg, R. (2015) 'A social-ecological analysis of ecosystem services in two different farming systems', *Ambio*, 44(1), pp. 102–112. doi: 10.1007/s13280-014-0603-y.

Andersson, G. K. S., Rundlöf, M. and Smith, H. G. (2012) 'Organic Farming Improves Pollination Success in Strawberries', *PLoS ONE*, 7(2), p. e31599. doi: 10.1371/Citation.

Ansell, D., Freudenberger, D., Munro, N. and Gibbons, P. (2016) 'The cost-effectiveness of agri-environment schemes for biodiversity conservation: A quantitative review', *Agriculture, Ecosystems and Environment*. Elsevier B.V., 225, pp. 184–191. doi: 10.1016/j.agee.2016.04.008.

Anthony, S. G., Silgram, M., Collins, A. L., Fawcett, L. E., Anthony, S. G., Silgram, M., Collins, A. L., Fawcett, L. E., Anthony, S. G., Silgram, M., Collins, A. L. and Fawcett, L. E. (2009) 'Modelling nitrate river water quality for policy support Modelling nitrate river water quality for policy support', *International Journal of River Basin Management*, 7:3, pp. 259–275. doi: 10.1080/15715124.2009.9635388.

Archer, N. A. L., Bonell, M., Coles, N., MacDonald, A. M., Auton, C. A. and Stevenson, R. (2013) 'Soil characteristics and landcover relationships on soil hydraulic conductivity at a hillslope scale: A view towards local flood management', *Journal of Hydrology*. Elsevier B.V., 497, pp. 208–222. doi: 10.1016/j.jhydrol.2013.05.043.

Armengot, L. and José-maría, L. (2011) 'A novel index of land use intensity for organic and conventional farming of Mediterranean cereal fields', *Agronomy for Sustainable Development*, 31, pp. 699–707. doi: 10.1007/s13593-011-0042-0.

Arnott, D., Chadwick, D. R., Wynne-jones, S. and Jones, D. L. (2021) 'Vulnerability of British farms to post-Brexit subsidy removal, and implications for intensification, extensification and land sparing', *Land Use Policy*. Elsevier, 107, p. 104154. doi: 10.1016/j.landusepol.2019.104154.

Askari, M. S. and Holden, N. M. (2015) ‘Quantitative soil quality indexing of temperate arable management systems’, *Soil and Tillage Research*. Elsevier B.V., 150, pp. 57–67. doi: 10.1016/j.still.2015.01.010.

Awan, S. (2016) *AHDB Research Review No . 87 A review of the past , present and future of precision agriculture in the UK*. Available at: https://cereals.ahdb.org.uk/media/1126475/rr87.pdf.

Bailey, R.J. and Spackman, E. (1996). ‘A model for estimating soil moisture changes as an aid to irrigation scheduling, and crop water use studies: Operational details and description’. *Soil Use Management.* 12, 122–128

Baker, J. M., Ochsner, T. E., Venterea, R. T. and Griffis, T. J. (2007) ‘Tillage and soil carbon sequestration — What do we really know ?’, *Agriculture, Ecosystems and Environment,* 118, pp. 1–5. doi: 10.1016/j.agee.2006.05.014.

Ball, B. C., Bingham, I., Rees, R. M., Watson, C. a. and Litterick, a. (2005) ‘The role of crop rotations in determining soil structure and crop growth conditions’, *Canadian Journal of Soil Science*, 85(5), pp. 557–577. doi: 10.4141/S04-078.

Barbieri, P., Pellerin, S. and Nesme, T. (2017) ‘Comparing crop rotations between organic and conventional farming’, *Scientific Reports*. Springer US, 7(1), pp. 1–10. doi: 10.1038/s41598-017-14271-6.

Barrios, E. (2007) ‘Soil biota, ecosystem services and land productivity’, *Ecological Economics*, 64(2), pp. 269–285. doi: 10.1016/j.ecolecon.2007.03.004.

Bartkowski, B., Bartke, S., Helming, K., Paul, C., Techen, A. and Hansjürgens, B. (2020) ‘Potential of the economic valuation of soil-based ecosystem services to inform sustainable soil management and policy’, *PeerJ*, 8:e8749, pp. 1–31. doi: 10.7717/peerj.8749.

Batáry, P., Dicks, L. V., Kleijn, D. and Sutherland, W. J. (2015) ‘The role of agri-environment schemes in conservation and environmental management’, *Conservation Biology*, 29(4), pp. 1006–1016. doi: 10.1111/cobi.12536.

Bateman, I. J., Abson, D., Beaumont, N., Darnell, A., Fezzi, C., Hanleys, N., Kontoleon, A., Maddison, D., Morling, P., Morris, J., Mourato, S., Pascual, U., Perino, G., Sen, A., Tinch, D., Turner, K., Valatin, G., Andrews, B., Asara, V., *et al.* (2011) ‘Chapter 22: Economic values from ecosystems’, *Human Well-being, UK National Ecosystem Assessment: Technical Report*, pp. 1067–1152. Available at: http://eprints.lse.ac.uk/44653/.

Bateman, I. J. and Balmford, B. (2018) ‘Public funding for public goods: A post-Brexit perspective on principles for agricultural policy’, *Land Use Policy*. Elsevier, 79, pp. 293–300. doi: 10.1016/j.landusepol.2018.08.022.

Bateman, I. J., Harwood, A. R., Mace, G. M., Watson, R. T., Abson, D. J., Andrews, B., Binner, A., Crowe, A., Day, B. H., Dugdale, S., Fezzi, C., Foden, J., Hadley, D., Haines-Young, R., Hulme, M., Kontoleon, A., Lovett, A. A., Munday, P., Pascual, U., *et al.* (2013) ‘Bringing ecosystem services into economic decision-making: Land use in the United Kingdom’, *Science*, 341(6141), pp. 45–50. doi: 10.1126/science.1234379.

Bateman, I. J. and Mace, G. M. (2020) ‘The natural capital framework for sustainably efficient and equitable decision making’, *Nature Sustainability*. Springer US, 3(10), pp. 776–783. doi: 10.1038/s41893-020-0552-3.

Bateman, I. J., Mace, G. M., Fezzi, C., Atkinson, G. and Turner, K. (2011) 'Economic analysis for ecosystem service assessments', *Environmental and Resource Economics*, 48(2), pp. 177–218. doi: 10.1007/s10640-010-9418-x.

Batey, T. (2009) 'Soil compaction and soil management - A review', *Soil Use and Management*, 25, pp. 335–345. doi: 10.1111/j.1475-2743.2009.00236.x.

Baude, M., Kunin, W. E., Boatman, N. D., Conyers, S., Davies, N., Gillespie, M. A. K., Morton, R. D., Smart, S. M. and Memmott, J. (2016) 'Historical nectar assessment reveals the fall and rise of Britain in bloom', *Nature*, 530(7588), pp. 85–88. doi: 10.1038/nature16532.

Baveye, P. C., Baveye, J. and Gowdy, J. (2016) 'Soil "Ecosystem" Services and Natural Capital: Critical Appraisal of Research on Uncertain Ground', *Frontiers in Environmental Science*, 4, pp. 1–49. doi: 10.3389/fenvs.2016.00041.

Bearcock, J. and Smedley, P. (2012) 'Baseline groundwater chemistry: the Sherwood Sandstone of Devon and Somerset', p. 73. Available at: http://nora.nerc.ac.uk/id/eprint/17094.

Beaudoin, N., Saad, J. K., Van Laethem, C., Machet, J. M., Maucorps, J. and Mary, B. (2005) 'Nitrate leaching in intensive agriculture in Northern France: Effect of farming practices, soils and crop rotations', *Agriculture, Ecosystems and Environment*, 111, pp. 292–310. doi: 10.1016/j.agee.2005.06.006.

Becher, M. A., Grimm, V., Knapp, J., Horn, J., Twiston-Davies, G. and Osborne, J. L. (2016) 'BEESCOUT: A model of bee scouting behaviour and a software tool for characterizing nectar/pollen landscapes for BEEHAVE', *Ecological Modelling*. Elsevier B.V., 340, pp. 126–133. doi: 10.1016/j.ecolmodel.2016.09.013.

Becher, M. A., Grimm, V., Thorbek, P., Horn, J., Kennedy, P. J. and Osborne, J. L. (2014) 'BEEHAVE: A systems model of honeybee colony dynamics and foraging to explore multifactorial causes of colony failure', *Journal of Applied Ecology*, 51, pp. 470–482. doi: 10.1111/1365-2664.12222.

Becher, M. A., Twiston-Davies, G., Penny, T. D., Goulson, D., Rotheray, E. L. and Osborne, J. L. (2018) 'Bumble-BEEHAVE: A systems model for exploring multifactorial causes of bumblebee decline at individual, colony, population and community level', *Journal of Applied Ecology*, pp. 1–12. doi: 10.1111/1365-2664.13165.

Beedell, J. (2021) *The value of natural capital accounts to farms and estates*, *Strutt and Parker*. Available at: https://rural.struttandparker.com/article/the-value-of-natural-capital-accounts-to-farms-and-estates/ (Accessed: 21 December 2021).

Benoit, M., Garnier, J., Anglade, J. and Billen, G. (2014) 'Nitrate leaching from organic and conventional arable crop farms in the Seine Basin (France)', *Nutrient Cycling in Agroecosystems*, 100, pp. 285–299. doi: 10.1007/s10705-014-9650-9.

Benoit, M., Garnier, J., Billen, G., Tournebize, J., Gréhan, E. and Mary, B. (2015) 'Nitrous oxide emissions and nitrate leaching in an organic and a conventional cropping system (Seine basin, France)', *Agriculture, Ecosystems and Environment*. Elsevier B.V., 213, pp. 131–141. doi: 10.1016/j.agee.2015.07.030.

Berdeni, D., Turner, A., Grayson, R. P., Llanos, J., Holden, J., Helgason, T., Watt,

P. J. and Leake, J. R. (2021) 'Soil quality regeneration by grass-clover leys in arable rotations compared to permanent grassland : effects on wheat yield and resilience to drought and flooding . White Rose Research Online URL for this paper : Version : Pub', *Soil and Tillage Research*, 212, p. 105037. doi: 10.1016/j.still.2021.105037.

Bergkamp, G. and Cross, K. (2006) 'Groundwater and ecosystem services: towards their sustainable use', *International Symposium on Groundwater Sustainability (ISGWAS)*, pp. 177–193. doi: 10.1073/pnas.0707823105.

Berry, P. M., Sylvester-Bradley, R., Philipps, L., Hatch, D. J., Cuttle, S. P., Rayns, F. W. and Gosling, P. (2002) 'Is the productivity of organic farms restricted by the supply of available nitrogen?', *Soil Use and Management*, 18, pp. 248–255. doi: 10.1079/SUM2002129.

Beynon, S. A., Wainwright, W. A. and Christie, M. (2015) 'The application of an ecosystem services framework to estimate the economic value of dung beetles to the U.K. cattle industry', *Ecological Entomology*, 40(S1), pp. 124–135. doi: 10.1111/een.12240.

Bhogal, A., Bentley, C., Price, P. N. and Chambers, B. (2011) *The alleviation of grassland compaction by mechanical soil loosening (BD5001)*.

Bianchi, F. J. J. ., Booij, C. J. . and Tscharntke, T. (2006) 'Sustainable pest regulation in agricultural landscapes: a review on landscape composition, biodiversity and natural pest control', *Proceedings of the Royal Society B: Biological Sciences*, 273(1595), pp. 1715–1727. doi: 10.1098/rspb.2006.3530.

Biernat, L., Taube, F., Vogeler, I., Reinsch, T., Kluß, C. and Loges, R. (2020) 'Agriculture , Ecosystems and Environment Is organic agriculture in line with the EU-Nitrate directive? On-farm nitrate leaching from organic and conventional arable crop rotations', *Agriculture, Ecosystems and Environment*. Elsevier, 298, p. 106964. doi: 10.1016/j.agee.2020.106964.

BIES (2020). Department for Business, Energy and Industrial Strategy: Gas and electricity prices in the non-domestic sector. Available at: https://www.gov.uk/government/statistical-data-sets/gas-and-electricity-prices-in-the-non-domestic-sector (Accessed: 30 September 2020).

Bilotta, G. S., Brazier, R. E. and Haygarth, P. (2007) 'The Impacts of Grazing Animals on the Quality of Soils, Vegetation, and Surface Waters in Intensively Managed Grasslands', *Advances in Agronomy*, 94, pp. 237–280. doi: 10.1016/S0065-2113(06)94006-1.

Bilotta, G. S., Brazier, R. E., Haygarth, P. M., Macleod, C. J. A., Butler, P., Granger, S., Krueger, T., Freer, J. and Quinton, J. (2008) 'Rethinking the Contribution of Drained and Undrained Grasslands to Sediment-Related Water Quality Problems', *Journal of Environment Quality*, 37(3), p. 906. doi: 10.2134/jeq2007.0457.

Bilsborrow, P., Cooper, J., Tétard-jones, C., Srednicka-tober, D., Eyre, M., Schmidt, C., Shotton, P., Volakakis, N., Cakmak, I., Ozturk, L., Leifert, C. and Wilcockson, S. (2013) 'The effect of organic and conventional management on the yield and quality of wheat grown in a long-term field trial', *European Journal of Agronomy*, 51, pp. 71–60. doi: 10.1016/j.eja.2013.06.003.

Birkhofer, K., Bezemer, T. M., Bloem, J., Bonkowski, M., Christensen, S., Dubois, D., Ekelund, F., Fließbach, A., Gunst, L., Hedlund, K., Mikola, J., Robin, C., Seta, H., Ma, P., Putten, W. H. Van Der and Scheu, S. (2008) 'Soil Biology & Biochemistry Long-term organic farming fosters below and aboveground biota: Implications for soil quality , biological control and productivity', *Soil Biology & Biochemistry journal*, 40, pp. 2297–2308. doi: 10.1016/j.soilbio.2008.05.007.

BHSMA (2019). 'British Hay and Straw Merchants Association for Great Britain and its regions: Hay and straw sale prices'. Available at: https://ahdb.org.uk/dairy/hay-and-straw-prices.

Blasco-Moreno, A., Pérez-Casany, M., Puig, P., Morante, M. and Castells, E. (2019) 'What does a zero mean? Understanding false, random and structural zeros in ecology', *Methods in Ecology and Evolution*, 10(7), pp. 949–959. doi: 10.1111/2041-210X.13185.

Bliss, K. (2018) *Integrating leys in Arable Rotations. Project Report for Defra: Organic management techniques to improve sustainability of non-organic farming.*

Bommarco, R., Kleijn, D. and Potts, S. G. (2013) 'Ecological intensification: Harnessing ecosystem services for food security', *Trends in Ecology and Evolution*. Elsevier Current Trends, pp. 230–238. doi: 10.1016/j.tree.2012.10.012.

Bommarco, R., Lundin, O., Smith, H. G. and Rundlöf, M. (2012) 'Drastic historic shifts in bumble-bee community composition in Sweden', *Proceedings of the Royal Society B: Biological Sciences*, 279, pp. 309–315. doi: 10.1098/rspb.2011.0647.

Bone, J., Head, M., Barraclough, D., Archer, M., Scheib, C., Flight, D. and Voulvoulis, N. (2010) 'Soil quality assessment under emerging regulatory requirements', *Environment International*. Elsevier, 36(6), pp. 609–622. doi: 10.1016/j.envint.2010.04.010.

Boretti, A. and Rosa, L. (2019) 'Reassessing the projections of the World Water Development Report', *npj Clean Water*. Springer US, 2(15). doi: 10.1038/s41545-019-0039-9.

Brady, M. V., Hedlund, K., Cong, R. G., Hemerik, L., Hotes, S., Machado, S., Mattsson, L., Schulz, E. and Thomsen, I. K. (2015) 'Valuing supporting soil ecosystem services in agriculture: A natural capital approach', *Agronomy Journal*, 107(5), pp. 1809–1821. doi: 10.2134/agronj14.0597.

Brady, N.C. and Weil, R.R. (2008) 'The nature and properties of soil, 14 ed'. Prentice-Hall, Upper Saddle River, New Jersey.

Brauman, K. A., Daily, G. C., Duarte, T. K. and Mooney, H. A. (2007) 'The Nature and Value of Ecosystem Services: An Overview Highlighting Hydrologic Services', *Annual Review of Environment and Resources*, 32(1), pp. 67–98. doi: 10.1146/annurev.energy.32.031306.102758.

Brazier, R.E., Anderson, K.A., Quine, T., Quinton, J., Evans, M., Rickson, J, Bellamy, P., Rawlins, B., Ellis, M. (2011) *Developing a cost-effective framework for monitoring soil erosion in England and Wales: Final report to Defra for project SP1303*.

Briggs, S. and Eclair-Heath, G. (2017) 'Helping UK farmers to choose, use, and interpret soil test results to inform soil management decisions for soil health', *Aspects of Applied Biology: Crop Production in Southern Britain*, 134, pp. 161–168.

Britt, C., Tuffnell, N., Kirkham, F., Roberts, A. and Sparks, T. (2011) *Hedgerow Management: A survey of Land Managers' And Contractors' Practices and Attitudes: Research report for DEFRA project BD2117*.

Brooks, M.E., Kristensen, K., van Benthem, K.J., Magnusson, A., Berg, C.W., Nielsen, A., Skaug, H.J., Maechler, M., Bolker, B.M. (2007). 'glmmTMB Balances Speed and Flexibility Among Packages for Zero-inflated Generalized Linear Mixed Modeling'. The R Journal, 9(2), 378-400.

Brown, T. C., Bergstrom, J. C. and Loomis, J. B. (2007) 'Defining, valuing and providing ecosystem goods and services', *Natural Resources Journal*, 47, pp. 329–376. doi: 10.1525/sp.2007.54.1.23.

Büchi, L., Georges, F., Walder, F., Banerjee, S., Keller, T., Six, J., Heijden, M. Van Der and Charles, R. (2019) 'Potential of indicators to unveil the hidden side of cropping system classification: Differences and similarities in cropping practices between conventional, no-till and organic systems', *European Journal of Agronomy*. Elsevier, 109, p. 125920. doi: 10.1016/j.eja.2019.125920.

Büchi, L., Wendling, M., Amossé, C., Jeangros, B., Sinaj, S. and Charles, R. (2017) 'Long and short term changes in crop yield and soil properties induced by the reduction of soil tillage in a long term experiment in Switzerland', *Soil & Tillage Research*. Elsevier, 174, pp. 120–129. doi: 10.1016/j.still.2017.07.002.

Buchmann, S. L., & O'Rourke, M. K. (1991). Importance of pollen grain volumes for calculating bee diets. GRANA, 30(3-4), 591-595. https://doi.org/10.1080/00173139109427817

Bulson H A J, Welsh J P, Stopes C E & Woodward L. (1996). 'Agronomic viability and potential economic performance of three organic four year rotations without livestock, 1988-1995'. *Aspects of Applied Biology*, 47, Rotations & Cropping Systems. 277-286.

Bünemann, E. K., Bongiorno, G., Bai, Z., Creamer, R. E., De Deyn, G., de Goede, R., Fleskens, L., Geissen, V., Kuyper, T. W., Mäder, P., Pulleman, M., Sukkel, W., van Groenigen, J. W. and Brussaard, L. (2018) 'Soil quality – A critical review', *Soil Biology and Biochemistry*. Elsevier, 120, pp. 105–125. doi: 10.1016/j.soilbio.2018.01.030.

Burnham, K. P. and Anderson, D. R. (2002) *Model Selection and Multimodel Inference: a Practical Information-Theoretic Approach*. 2nd Ed. Berlin: Springer Berlin Heidelberg. doi: 10.1016/b978-0-12-801370-0.00011-3.

Busari, M. A., Kukal, S. S., Kaur, A., Bhatt, R. and Dulazi, A. A. (2015) 'Conservation tillage impacts on soil, crop and the environment', *International Soil and Water Conservation Research*. Elsevier, 3(2), pp. 119–129. doi: 10.1016/j.iswcr.2015.05.002.

Calzolari, C., Ungaro, F., Filippi, N., Guermandi, M., Malucelli, F., Marchi, N., Staffilani, F. and Tarocco, P. (2016) 'A methodological framework to assess the multiple contributions of soils to ecosystem services delivery at regional scale', *Geoderma*, 261, pp. 190–203. doi: 10.1016/j.geoderma.2015.07.013.

Cardoso, E. J. B. N., Vasconcellos, R. L. F., Bini, D., Miyauchi, M. Y. H., Santos, C. A. dos, Alves, P. R. L., Paula, A. M. De, Nakatani, A. S., Pereira, J. de M. and Nogueira, M. A. (2013) 'Soil health: looking for suitable indicators. What should be considered to assess the effects of use and management on soil health?', *Scientia Agricola*, 70(4), pp. 274–289. doi: 10.1590/S0103-90162013000400009.

Carolan, R. and Fornara, D. A. (2016) 'Soil carbon cycling and storage along a chronosequence of re-seeded grasslands: Do soil carbon stocks increase with grassland age?', *Agriculture, Ecosystems and Environment*. Elsevier B.V., 218, pp. 126–132. doi: 10.1016/j.agee.2015.11.021.

Carvalheiro, L. G., Kunin, W. E., Keil, P., Aguirre-Gutiérrez, J., Ellis, W. N., Fox, R., Groom, Q., Hennekens, S., Van Landuyt, W., Maes, D., Van de Meutter, F., Michez, D., Rasmont, P., Ode, B., Potts, S. G., Reemer, M., Roberts, S. P. M., Schaminée, J., Wallisdevries, M. F., *et al.* (2013) 'Species richness declines and biotic homogenisation have slowed down for NW-European pollinators and plants', *Ecology Letters*, 16(7), pp. 870–878. doi: 10.1111/ele.12121.

Carvell, C., Isaac, N., Jitlal, M., Peyton, J., Powney, G., Roy, D., Vanbergen, A., O'Connor, R., Jones, C., Kunin, W., Breeze, T., Garratt, M., Potts, S., Harvey, M., Ansine, J., Comont, R., Lee, P., Edwards, M., Roberts, S., *et al.* (2016) 'Design and Testing of a National Pollinator and Pollination Monitoring Framework.', (April), p. 65.

Carvell, C., Meek, W. R., Pywell, R. F., Goulson, D. and Nowakowski, M. (2007) 'Comparing the efficacy of agri-environment schemes to enhance bumble bee abundance and diversity on arable field margins', *Journal of Applied Ecology*, 44(1), pp. 29–40. doi: 10.1111/j.1365-2664.2006.01249.x.

Carvell, C., Osborne, J. L., Bourke, A. F. G., Freeman, S. N., Pywell, R. F. and Heard, M. S. (2011) 'Bumble bee species' responses to a targeted conservation measure depend on landscape context and habitat quality', *Ecological Applications*, 21(5), pp. 1760–1771. doi: 10.1890/10-0677.1.

Carvell, C., Roy, D. B., Smart, S. M., Pywell, R. F., Preston, C. D. and Goulson, D. (2006) 'Declines in forage availability for bumblebees at a national scale', *Biological Conservation*, 132(4), pp. 481–489. doi: 10.1016/j.biocon.2006.05.008.

CCI (2016) *Biodiversity at the heart of accounting for natural capital: the key to credibility (Cambridge Conservation Initiative)*. Available at: https://www.cambridgeconservation.org/wp-content/uploads/2016/07/CCI-Natural-Capital-Paper-July-2016_web-version.pdf.

CEH (2020). 'UKCEH Land Cover® plus: Crops map from the Centre for Ecology and Hydrology'. Downloaded from: https://digimap.edina.ac.uk (Accessed: 5 November 2020).

Chen, G., Weil, R. R. and Hill, R. L. (2014) 'Effects of compaction and cover crops on soil least limiting water range and air permeability', *Soil and Tillage Research*. Elsevier B.V., 136, pp. 61–69. doi: 10.1016/j.still.2013.09.004.

CLA (2021) 'A CLA Introduction to Natural Capital; Country Land and Bussiness Association'. London.

Calcagno, V. (2020). 'glmulti: Model Selection and Multimodel Inference Made Easy'. R package version 1.0.8. https://CRAN.R-project.org/package=glmulti

Comissão Europeia (2020) *Farm to Fork Strategy*. Available at: https://ec.europa.eu/food/sites/food/files/safety/docs/f2f_action-plan_2020_strategy-info_en.pdf.

Conant, R. T., Paustian, K. and Elliott, E. T. (2001) 'Grassland management and conversion into grassland: Effects on soil carbon', *Ecological Applications*, 11(2), pp. 343–355. doi: 10.1890/1051-0761(2001)011[0343:GMACIG]2.0.CO;2.

Connor, D. J. and Mínguez, M. I. (2012) 'Evolution not revolution of farming systems will best feed and green the world', *Global Food Security*, 1(2), pp. 106–113. doi: 10.1016/j.gfs.2012.10.004.

Connors, E. (2016) 'UK Natural Capital : Ecosystem accounts for farmland (Experimental Statistics)', pp. 1–22.

Cooper, J., Lindenthal, T., Reed, E. Y., Ho, S., Magid, D. J., Oberson, A. and Ma, A. L. P. (2018) 'Phosphorus availability on many organically managed farms in Europe', *Nutrient Cycling in Agroecosystems*, 110, pp. 227–239. doi: 10.1007/s10705-017-9894-2.

Cooper, R. J., Hama-Aziz, Z., Hiscock, K. M., Lovett, A. A., Dugdale, S. J., Sünnenberg, G., Noble, L., Beamish, J. and Hovesen, P. (2017) 'Assessing the farm-scale impacts of cover crops and non-inversion tillage regimes on nutrient losses from an arable catchment', *Agriculture, Ecosystems and Environment*. Elsevier B.V., 237, pp. 181–193. doi: 10.1016/j.agee.2016.12.034.

Costanza, R., D'Arge, R., de Groot, R., Farber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., O'Neill, R. V., Paruelo, J., Raskin, R. G., Sutton, P. and van den Belt, M. (1997) 'The value of the world's ecosystem services and natural capital', *Nature*, 387(6630), pp. 253–260. doi: 10.1038/387253a0.

Crowder, D. W. and Reganold, J. P. (2015) 'Financial competitiveness of organic agriculture on a global scale', *PNAS*, 112(24), pp. 7611–7616. doi: 10.1073/pnas.1423674112.

Dadson, S. J., Hall, J. W., Murgatroyd, A., Acreman, M., Bates, P., Beven, K., Heathwaite, L., Holden, J., Holman, I. P., Lane, S. N., O'Connell, E., Penning-Rowsell, E., Reynard, N., Sear, D., Thorne, C. and Wilby, R. (2017) 'A restatement of the natural science evidence concerning catchment-based "natural" flood management in the UK', *Proceedings of the Royal Society A: Mathematical, Physical and Engineering Science*, 473(2199), p. 20160706. doi: 10.1098/rspa.2016.0706.

Daigh, A. L. M. and Dejong-hughes, J. (2017) 'Fluffy soil syndrome : When tilled soil does not settle', 72(1), pp. 10–14. doi: 10.2489/jswc.72.1.10A.

Dale, V. H. and Polasky, S. (2007) 'Measures of the effects of agricultural practices on ecosystem services', *Ecological Economics*, 64(2), pp. 286–296. doi: 10.1016/j.ecolecon.2007.05.009.

Davies, D.H.K. and Welsh, J.P. (2002) 'Weed control in organic cereals and pulses'. In: Younie, D.; Taylor, B.R.; Welch, J.M. and Wilkinson, J.M. (Eds.) Organic cereals and pulses. *Papers presented at conferences held at the Heriot-Watt University, Edinburgh, and at Cranfield University Silsoe Campus,* *Bedfordshire, 6 and 9 November 2001.* Chalcombe Publications, chapter 5, pp. 77-114.

Defra (2019) *Farming Statistics Land Use , Livestock Populations and Agricultural Workforce At 1 June 2019 - England*.

Defra (2020a) *Enabling a Natural Capital Approach (ENCA)*. Available at: www.gov.uk/defra%0Ahttps://data.gov.uk/dataset/3930b9ca-26c3-489f-900f-6b9eec2602c6/enabling-a-natural-capital-approach.

Defra (2020b) 'Farming Statistics Final Crop Areas , Yields , Livestock Populations and Agricultural Workforce at 1 June 2020 , United Kingdom', (December), pp. 1–30.

Defra (2021) *Sustainable Farming Incentive: how the scheme will work in 2022*. Available at: https://www.gov.uk/government/publications/sustainable-farming-incentive-how-the-scheme-will-work-in-2022/sustainable-farming-incentive-how-the-scheme-will-work-in-2022 (Accessed: 15 December 2021).

DEFRA (2020) *Environmental Land Management tests and trials Quarterly evidence report*. Available at: www.gov.uk/government/publications.

DEFRA and National Statistics (2021) *Farm Accounts in England: Results from the Farm Business Survery 2019/20*. Available at: https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/962279/fbs_farmaccountsengland_18feb21.pdf.

Dickie, I., Evans, C., Smyth, M. and Artz, R. (2015) *Scoping the Natural Capital Accounts for Peatland: work package 3 of Report NR0165 for Defra*.

Dicks, L. V., Baude, M., Roberts, S. P. M., Phillips, J., Green, M. and Carvell, C. (2015) 'How much flower-rich habitat is enough for wild pollinators? Answering a key policy question with incomplete knowledge', *Ecological Entomology*, 40(S1), pp. 22–35. doi: 10.1111/een.12226.

Diekotter, T., Wamser, S., Wolters, V. and Birkhofer, K. (2010) 'Landscape and management effects on structure and function of soil arthropod communities in winter wheat', *Agriculture , Ecosystems and Environment*, 137, pp. 108–112. doi: 10.1016/j.agee.2010.01.008.

Dominati, E. J., Mackay, A. D., Bouma, J. and Green, S. (2016) 'An Ecosystems Approach to Quantify Soil Performance for Multiple Outcomes: The Future of Land Evaluation?', *Soil Science Society of America Journal*, 80(2), p. 438. doi: 10.2136/sssaj2015.07.0266.

Dominati, E., Mackay, A., Green, S. and Patterson, M. (2014) 'A soil change-based methodology for the quantification and valuation of ecosystem services from agro-ecosystems: A case study of pastoral agriculture in New Zealand', *Ecological Economics*. Elsevier B.V., 100, pp. 119–129. doi: 10.1016/j.ecolecon.2014.02.008.

Dominati, E., Patterson, M. and Mackay, A. (2010) 'A framework for classifying and quantifying the natural capital and ecosystem services of soils', *Ecological Economics*. Elsevier B.V., 69(9), pp. 1858–1868. doi: 10.1016/j.ecolecon.2010.05.002.

Domínguez, A., Camilo, J., Rosa, A. and Vanesa, R. (2014) 'Organic farming fosters agroecosystem functioning in Argentinian temperate soils : Evidence from litter decomposition and soil fauna', *Applied Soil Ecology*. Elsevier B.V., 83, pp. 170–176. doi: 10.1016/j.apsoil.2013.11.008.

Döring, T. F., Baddeley, J. A., Brown, R., Collins, R., Crowley, O., Cuttle, S., Howlett, S. A., Jones, H. E., Mccalman, H., Measures, M., Pearce, B. D., Pearce, H., Roderick, S., Stobart, R., Storkey, J., Tilston, E. L., Topp, K., Watson, C., Winkler, L. R., *et al.* (2013) *Using legume-based mixtures to enhance the nitrogen use efficiency and economic viability of cropping systems: Project Report No. 513*. Available at: http://www.sruc.ac.uk/download/downloads/id/1457/sac-sepa_conference_2010_proceedings#page=187%5Cnhttp://www.sruc.ac.uk/download/downloads/id/1457/sac-sepa%7B_%7Dconference%7B_%7D2010%7B_%7Dproceedings%7B#%7Dpage=187.

Duncan, C., Thompson, J. R. and Pettorelli, N. (2015) 'The quest for a mechanistic understanding of biodiversity – ecosystem services relationships', *Royal Society Publishing Proceedings B*, 282(20151348), pp. 1–10. doi: 10.1098/rspb.2015.1348.

Dungait, J. A. J., Cardenas, L. M., Blackwell, M. S. A., Wu, L., Withers, P. J. A., Chadwick, D. R., Bol, R., Murray, P. J., Macdonald, A. J., Whitmore, A. P. and Goulding, K. W. T. (2012) 'Advances in the understanding of nutrient dynamics and management in UK agriculture', *Science of the Total Environment*. Elsevier B.V., 434, pp. 39–50. doi: 10.1016/j.scitotenv.2012.04.029.

Eder, A., Salhofer, K. and Scheichel, E. (2021) 'Land tenure, soil conservation, and farm performance: An eco-efficiency analysis of Austrian crop farms', *Ecological Economics*. Elsevier, 180, p. 106861. doi: 10.1016/j.ecolecon.2020.106861.

EA (2018). The state of the environment: water resources. Available at: https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/709924/State_of_the_environment_water_resources_report.pdf (Accessed: 30 December 2021).

EFTEC (2018) *Cholderton Estate Natural Capital Account An Illustration of How Good Farming Pays*.

EFTEC (2019) *Natural capital accounting for land managers*.

ELD Initiative (2015) *The Economics of Land Degradation Initiative: Reaping economic and environmental benefits from sustainable land management*. Available at: http://www.eld-initiative.org/fileadmin/pdf/ELD-pm-report_05_web_300dpi.pdf.

Elliott, J., Firbank, L. G., Drake, B., Cao, Y. and Gooday, R. (2013) *Exploring the Concept of Sustainable Intensification*, *Land Use Policy Group*. Available at: www.lupg.org.uk%5Cnwww.snh.org.uk.

Ellis, N., Anderson, K. and Brazier, R. (2021) 'Mainstreaming natural flood management : A proposed research framework derived from a critical evaluation of current knowledge', *Progress in Physical Geography*, pp. 1–23. doi: 10.1177/0309133321997299.

Elsin, Y. K., Kramer, R. a. and Jenkins, W. A. (2010) 'Valuing Drinking Water

Provision as an Ecosystem Service in the Neuse River Basin', *Journal of Water Resources Planning and Management*, 136(4), pp. 474–482. doi: 10.1061/(ASCE)WR.1943-5452.0000058.

Emmett, B. A., Frogbrook, Z. L., Chamberlain, P. M., Giffiths, R., Pickup, R., Poskitt, J., Reynolds, B., Rowe, E., Spurgeon, D., Rowland, P., Wilson, J., Wood, C. M., Reynolds, B., Rowe, E., Rowland, P., Spurgeon, D., Wilson, J. and Wood, C. M. (2008) *CS Technical Report No.3/07: Soil Manual*.

Emmett, B. A., Reynolds, B., Chamberlain, P. M., Rowe, E., D., S., BrittainS.A., Frogbrook, Z., Hughes, S., Lawlor, A. J., Poskitt, J., Potter, E., Robinson, D. A., Scott, A., Wood, C. . and Woods, C. (2010) *CS Techincal Report No.9/07: Soils Report from 2007*. Available at: http://nora.nerc.ac.uk/9354/1/CS_UK_2007_TR9.pdf.

Environment Agency (2021) *The nitrate leaching tool: user guide. Chief Scientist's Group report. Project Number: SC160010*.

European Commission (2021) *Communication from the Commission to the European Parliament, the Council, the European Economic and Social Committee and the Committee of the Regions on an Action Plan for the Development of Organic Production. COM(2021) 141 final.*

Faccioli, M., Zonneveld, S., Tyler, C. and Day, B. (2020) *Local Natural Capital Accounting: does it deliver useful management information? A case study of Dartmoor and Exmoor National Parks*.

Fan, F., Henriksen, C. B. and Porter, J. (2016) 'Valuation of ecosystem services in organic cereal crop production systems with different management practices in relation to organic matter input', *Ecosystem Services*. Elsevier, 22, pp. 117–127. doi: 10.1016/j.ecoser.2016.10.007.

FAO (2015) *Natural Capital Impacts in Agriculture: Supporting Better Business Decision-Making*. Available at: http://www.fao.org/fileadmin/templates/nr/sustainability_pathways/docs/Natural_Capital_Impacts_in_Agriculture_final.pdf.

FAO (2017) *Soil Organic Carbon: the hidden potential. Food and Agriculture Organization of the United Nations.* Rome, Italy.

Feber, R. E., Firbank, L. G., Johnson, P. J. and MacDonald, D. W. (1997) 'The effects of organic farming on pest and non-pest butterfly\nabundance', *Agriculture, Ecosystems and Environment*, 64, pp. 133–139.

Fenichel, E. P., Abbott, J. K., Bayham, J., Boone, W., Haacker, E. M. K. and Pfeiffer, L. (2016) 'Measuring the value of groundwater and other forms of natural capital', *PNAS*, 113(9), pp. 2382–2387. doi: 10.1073/pnas.1513779113.

Ferrini, S., Schaafsma, M. and Bateman, I. (2014) 'Revealed and stated preference valuation and transfer: A within-sample comparison of water quality improvement values', *Water Resources Research*, 50, pp. 4746–4759. doi: 10.1002/2013WR014905.

Fezzi, C., Bateman, I., Askew, T., Munday, P., Pascual, U., Sen, A. and Harwood, A. (2014) 'Valuing Provisioning Ecosystem Services in Agriculture : United Kingdom', *Environmental Resource Economics*, 57, pp. 197–214. doi: 10.1007/s10640-013-9663-x.

Fezzi, C., Harwood, A. R., Lovett, A. A. and Bateman, I. J. (2015) 'S1 The environmental impact of climate change adaptation on land use and water quality', *Nature Climate Change*, 5(3), pp. 255–260. doi: 10.1038/nclimate2525.

Firbank, L. G., Elliott, J., Drake, B., Cao, Y. and Gooday, R. (2013) 'Evidence of sustainable intensification among British farms', *Agriculture, Ecosystems and Environment*. Elsevier B.V., 173, pp. 58–65. doi: 10.1016/j.agee.2013.04.010.

Forcone, A. *et al.* (2011) 'Botanical composition and protein content of pollen collected by Apis mellifera L. in the north-west of Santa Cruz (Argentinean Patagonia)', *Grana*, 50(1), pp. 30–39. doi: 10.1080/00173134.2011.552191.

Gabriel, D., Sait, S. M., Kunin, W. E. and Benton, T. G. (2013) 'Food production vs. biodiversity: Comparing organic and conventional agriculture', *Journal of Applied Ecology*, 50(2), pp. 355–364. doi: 10.1111/1365-2664.12035.

Garratt, M. P. D., Coston, D. J., Truslove, C. L., Lappage, M. G., Polce, C., Dean, R., Biesmeijer, J. C. and Potts, S. G. (2014) 'The identity of crop pollinators helps target conservation for improved ecosystem services', *Biological Conservation*. Elsevier Ltd, 169, pp. 128–135. doi: 10.1016/j.biocon.2013.11.001.

Gattinger, A., Muller, A., Haeni, M., Skinner, C., Fliessbach, A. and Buchmann, N. (2012) 'Enhanced top soil carbon stocks under organic farming', *PNAS*, 109(44), pp. 18226–18231. doi: 10.1073/pnas.1209429109.

Geppert, C., Hass, A., Földesi, R., Donkó, B., Akter, A., Tscharntke, T. and Batáry, P. (2020) 'Agri-environment schemes enhance pollinator richness and abundance but bumblebee reproduction depends on field size', *Journal of Applied Ecology*, 57(9), pp. 1818–1828. doi: 10.1111/1365-2664.13682.

Ghaley, Bhim B, Porter, J. R. and Sandhu, H. S. (2014) 'Soil-based ecosystem services : a synthesis of nutrient cycling and carbon sequestration assessment methods', *International Journal of Biodiversity Science, Ecosystem Services & Management*. Taylor & Francis, 10(3), pp. 177–186. doi: 10.1080/21513732.2014.926990.

Ghaley, B. B., Sandhu, H. S. and Porter, J. R. (2015) 'Relationship between C:N/C:O stoichiometry and ecosystem services in managed production systems', *PLoS ONE*, 10(4), pp. 1–14. doi: 10.1371/journal.pone.0123869.

Ghaley, Bhim Bahadur, Vesterdal, L. and Porter, J. R. (2014) 'Quantification and valuation of ecosystem services in diverse production systems for informed decision-making', *Environmental Science and Policy*. Elsevier Ltd, 39, pp. 139–149. doi: 10.1016/j.envsci.2013.08.004.

Gibbs, P. E. and Talavera, S. (2001) 'Breeding System Studies with Three Species of Anagallis (Primulaceae): Selfincompatibility and Reduced Female Fertility in A. monelli L.', 88(1), pp. 139–144.

Glendell, M. and Brazier, R. E. (2014) 'Accelerated export of sediment and carbon from a landscape under intensive agriculture', *Science of the Total Environment*. Elsevier B.V., 476–477, pp. 643–656. doi: 10.1016/j.scitotenv.2014.01.057.

Glendell, Miriam, Granger, S. J., Bol, R. and Brazier, R. E. (2014) 'Quantifying the spatial variability of soil physical and chemical properties in relation to mitigation of diffuse water pollution', *Geoderma*. Elsevier B.V., 214–215, pp. 25–41. doi: 10.1016/j.geoderma.2013.10.008.

Global Food Security (2014) *Agriculture's impacts on water quality: Farming and Water 1*. Available at: http://www.foodsecurity.ac.uk/assets/pdfs/farming-availability-water-report.pdf.

Gomiero, T. (2013) 'Alternative Land Management Strategies and Their Impact on Soil Conservation', *Agriculture*, 3, pp. 464–483. doi: 10.3390/agriculture3030464.

Gomiero, T., Pimentel, D. and Paoletti, M. G. (2011) 'Environmental Impact of Different Agricultural Management Practices: Conventional vs. Organic Agriculture', *Critical Reviews in Plant Sciences*, 30, pp. 95–124. doi: 10.1080/07352689.2011.554355.

Gosling, P. and Shepherd, M. (2005) 'Long-term changes in soil fertility in organic arable farming systems in England, with particular reference to phosphorus and potassium', *Agriculture , Ecosystems and Environment*, 105, pp. 425–432. doi: 10.1016/j.agee.2004.03.007.

Goulding, K. W. T., Poulton, P. R., Webster, C. P. and Howe, M. T. (2000) 'Nitrate leaching from the broadbalk wheat experiment, Rothamsted, UK, as influenced by fertilizer and manure inputs and the weather', *Soil Use and Management*, 16(4), pp. 244–250. doi: 10.1111/j.1475-2743.2000.tb00203.x.

Goulson, D., Rayner, P., Dawson, B. and Darvill, B. (2011) 'Translating research into action; bumblebee conservation as a case study', *Journal of Applied Ecology*, 48(1), pp. 3–8. doi: 10.1111/j.1365-2664.2010.01929.x.

Grand-Clement, E., Anderson, K., Smith, D., Luscombe, D., Gatis, N., Ross, M. and Brazier, R. E. (2013) 'Evaluating ecosystem goods and services after restoration of marginal upland peatlands in South-West England', *Journal of Applied Ecology*, 50(2), pp. 324–334. doi: 10.1111/1365-2664.12039.

Graves, A., Morris, J. Deeks, L. K., Rickson, J., Kibblewhite, M. G., Harris, J. A. and Farewell, T. S. (2011) *The Total Costs of Soils Degradation in England and Wales. Defra project CTE0946: Final Report submitted to Defra*.

Graves, A. R., Morris, J., Deeks, L. K., Rickson, R. J., Kibblewhite, M. G., Harris, J. A., Farewell, T. S. and Truckle, I. (2015) 'The total costs of soil degradation in England and Wales', *Ecological Economics*. Elsevier B.V., 119, pp. 399–413. doi: 10.1016/j.ecolecon.2015.07.026.

Gregory, A. S., Ritz, K., Mcgrath, S. P., Quinton, J. N., Goulding, K. W. T., Jones, R. J. A., Harris, J. A., Bol, R., Wallace, P., Pilgrim, E. S. and Whitmore, A. P. (2015) 'A review of the impacts of degradation threats on soil properties in the UK', *Soil Use and Management*, 31(July 2015), pp. 1–15. doi: 10.1111/sum.12212.

Greiner, L., Keller, A., Grêt-Regamey, A. and Papritz, A. (2017) 'Soil function assessment: review of methods for quantifying the contributions of soils to ecosystem services', *Land Use Policy*. Elsevier, 67, pp. 224–237. doi: http://dx.doi.org/10.1016/j.landusepol.2017.06.025.

Griffiths, B. S., Römbke, J., Schmelz, R. M., Scheffczyk, A., Faber, J. H., Bloem, J., Pérès, G., Cluzeau, D., Chabbi, A., Suhadolc, M., Sousa, J. P., Martins Da Silva, P., Carvalho, F., Mendes, S., Morais, P., Francisco, R., Pereira, C., Bonkowski, M., Geisen, S., *et al.* (2016) 'Selecting cost effective and policy-relevant biological indicators for European monitoring of soil biodiversity and ecosystem function', *Ecological Indicators*. Elsevier Ltd, 69, pp. 213–223. doi: 10.1016/j.ecolind.2016.04.023.

Grizzetti, B., Lanzanova, D., Liquete, C., Reynaud, A. and Cardoso, A. C. (2016) 'Assessing water ecosystem services for water resource management', *Environmental Science and Policy*. Elsevier Ltd, 61, pp. 194–203. doi: 10.1016/j.envsci.2016.04.008.

Van Groenigen, J. W., Van Kessel, C., Hungate, B. A., Oenema, O., Powlson, D. S. and Van Groenigen, K. J. (2017) 'Sequestering Soil Organic Carbon: A Nitrogen Dilemma', *Environmental Science and Technology*, 51, pp. 4738–4739. doi: 10.1021/acs.est.7b01427.

Van Groenigen, J. W., Velthof, G. L., Oenema, O., Van Groenigen, K. J. and Van Kessel, C. (2010) 'Towards an agronomic assessment of N2O emissions: A case study for arable crops', *European Journal of Soil Science*, 61(6), pp. 903–913. doi: 10.1111/j.1365-2389.2009.01217.x.

Grueber, C. E., Nakagawa, S., Laws, R. J. and Jamieson, I. G. (2011) 'Multimodel inference in ecology and evolution: Challenges and solutions', *Journal of Evolutionary Biology*, 24(4), pp. 699–711. doi: 10.1111/j.1420-9101.2010.02210.x.

Guerry, A. D., Polasky, S., Lubchenco, J., Chaplin-Kramer, R., Daily, G. C., Griffin, R., Ruckelshaus, M., Bateman, I. J., Duraiappah, A., Elmqvist, T., Feldman, M. W., Folke, C., Hoekstra, J., Kareiva, P. M., Keeler, B. L., Li, S., McKenzie, E., Ouyang, Z., Reyers, B., *et al.* (2015) 'Natural capital and ecosystem services informing decisions: From promise to practice', *PNAS*, 112(24), pp. 7348–7355. doi: 10.1073/pnas.1503751112.

Hackbart, V. C. S., de Lima, G. T. N. P. and dos Santos, R. F. (2017) 'Theory and practice of water ecosystem services valuation: Where are we going?', *Ecosystem Services*. Elsevier, 23, pp. 218–227. doi: 10.1016/j.ecoser.2016.12.010.

Haddaway, N. R., Hedlund, K., Jackson, L. E., Kätterer, T., Lugato, E., Thomsen, I. K., Jørgensen, H. B. and Isberg, P. E. (2017) *How does tillage intensity affect soil organic carbon? A systematic review*, *Environmental Evidence*. BioMed Central. doi: 10.1186/s13750-017-0108-9.

Haines-young, R. and Potschin, M. (2008) *Englands Terrestrial Ecosystem Services and the Rationale for an Ecosystem Approach. Full Technical Report to Defra (Project Code NR0107)*.

Hanley, N., Breeze, T. D., Ellis, C. and Goulson, D. (2015) 'Measuring the economic value of pollination services: Principles, evidence and knowledge gaps', *Ecosystem Services*. Elsevier, 14, pp. 124–132. doi: 10.1016/j.ecoser.2014.09.013.

Hanley, N., Ellis, C. and Breeze, T. (2013) *(UK) Accounting for the value of pollination services: For the Valuation of Natural Capital Accounting workshop, London, November 11th 2013.*

Hardman, C. J., Norris, K., Nevard, T. D., Hughes, B. and Potts, S. G. (2016a) 'Delivery of floral resources and pollination services on farmland under three different wildlife-friendly schemes', *Agriculture, Ecosystems and Environment*. Elsevier B.V., 220, pp. 142–151. doi: 10.1016/j.agee.2016.01.015.

Hardman, C. J., Norris, K., Nevard, T. D., Hughes, B. and Potts, S. G. (2016b) 'Delivery of floral resources and pollination services on farmland under three different wildlife-friendly schemes', *Agriculture, Ecosystems and Environment*. Elsevier B.V., 220, pp. 142–151. doi: 10.1016/j.agee.2016.01.015.

Harris, D., Boxworth, A., Crabtree, B., Consulting, C. J. C., King, J., Boxworth, A., Newell-price, P. and Gleadthorpe, A. (2006) *Economic Valuation of Soil Functions Phase 1 : Literature Review and Method Development. Prepared for Defra.*

Harris, L. (2021) 'Natural capital : What it is and how to value it on your farm', *Farmers Weekly*, pp. 1–10. Available at: https://www.fwi.co.uk/business/payments-schemes/environmental-schemes/natural-capital-on-farms-what-it-is-and-how-to-value-it.

Harrison, X. A. (2015) 'A comparison of observation-level random e ff ect and Beta-Binomial models for modelling overdispersion in Binomial data in ecology & evolution', *PeerJ*, 3(e1114). doi: 10.7717/peerj.1114.

Hartig, F. (2021). 'DHARMa: Residual Diagnostics for Hierarchical (Multi-Level / Mixed) Regression Models'. R package version 0.4.1. https://CRAN.R-project.org/package=DHARMa

Hass, A. L., Brachmann, L., Batáry, P., Clough, Y., Behling, H. and Tscharntke, T. (2019) 'Maize- dominated landscapes reduce bumblebee colony growth through pollen diversity loss', *Journal of Applied Ecology*, 56, pp. 294–304. doi: 10.1111/1365-2664.13296.

Hass, A. L., Kormann, U. G., Tscharntke, T., Clough, Y., Baillod, A. B., Sirami, C., Fahrig, L., Martin, J. L., Baudry, J., Bertrand, C., Bosch, J., Brotons, L., Bure, F., Georges, R., Giralt, D., Marcos-García, M., Ricarte, A., Siriwardena, G. and Batáry, P. (2018) 'Landscape configurational heterogeneity by small-scale agriculture, not crop diversity, maintains pollinators and plant reproduction in western Europe', *Proceedings of the Royal Society B*, 285(20172242). doi: 10.1098/rspb.2017.2242.

Hatt, S., Lopes, T., Boeraeve, F., Chen, J. and Francis, F. (2017) 'Pest regulation and support of natural enemies in agriculture: Experimental evidence of within field wildflower strips', *Ecological Engineering*. Elsevier B.V., 98, pp. 240–245. doi: 10.1016/j.ecoleng.2016.10.080.

Haygarth, P. M., Withers, P. J. A. and Hutchins, M. (2002) *Theoretical and Practical Effectiveness of Phosphorus and Associated Nutrient/Sediment Mitigation Measures in England and Wales*.

Hayhow DB, Burns F, Eaton MA, Al Fulaij N, August TA, Babey L, Bacon L, B., C, Boswell J, Boughey KL, Brereton T, Brookman E, Brooks DR, Bullock DJ, B. O., Collis M, Corbet L, Cornish N, De Massimi S, Densham J, Dunn E, Elliott S, G. T., Godber J, Hamilton S, Havery S, Hawkins S, Henney J, Holmes K, H. N., Isaac NJB, Johns D, Macadam CR, Mathews F, Nicolet P, Noble DG, O., CL, Powney GD, Richardson P, Roy DB, Sims D, Smart S, Stevenson K, S. R. and Walker KJ, Webb JR, Webb TJ, W. R. and G. R. (2016) 'State of Nature 2016', *The State of Nature Partnership*, pp. 1–85. Available at:

https://www.rspb.org.uk/globalassets/downloads/documents/conservation-projects/state-of-nature/state-of-nature-uk-report-2016.pdf.

Helm, D. (2017) ‘Agriculture after Brexit’, *Oxford Review of Economic Policy*, 33, pp. S124–S133. doi: 10.1093/oxrep/grx010.

Henrys, P. A., Keith, A. M., Robinson, D. A. and Emmett, B. A. (2012) *Model estimates of topsoil carbon [Countryside Survey]. NERC Environmental Information Data Centre.* doi: https://doi.org/10.5285/9e4451f8-23d3-40dc-9302-73e30ad3dd76.

Herrmann, M. and Pott, R. (2005) ‘Leaching of nitrate and ammonium in heathland and forest ecosystems in Northwest Germany under the influence of enhanced nitrogen deposition’, *Plant and Soil*, 273, pp. 129–137. doi: 10.1007/s11104-004-7246-x.

Herzog, F., Steiner, B., Bailey, D., Baudry, J., Billeter, R., Bukácek, R., De Blust, G., De Cock, R., Dirksen, J., Dormann, C. F., De Filippi, R., Frossard, E., Liira, J., Schmidt, T., Stöckli, R., Thenail, C., Van Wingerden, W. and Bugter, R. (2006) ‘Assessing the intensity of temperate European agriculture at the landscape scale’, *European Journal of Agronomy*, 24(2), pp. 165–181. doi: 10.1016/j.eja.2005.07.006.

Hewitt, A., Dominati, E., Webb, T. and Cuthill, T. (2015) ‘Soil natural capital quantification by the stock adequacy method’, *Geoderma*. Elsevier B.V., 241–242, pp. 107–114. doi: 10.1016/j.geoderma.2014.11.014.

Hewitt, A. E., Hedley, C. and Rosser, B. (2010) ‘Evaluation of soil natural capital in two soilscapes’, *19th World Congress of Soil Science, Soil Solutions for a Changing World*, pp. 17–20. Available at: http://www.ldd.go.th/swcst/Report/soil/.%5Csymposium/.%5Cpdf/1538.pdf.

Hicks, D. M., Ouvrard, P., Baldock, K. C. R., Baude, M., Goddard, A., Kunin, W. E., Mitschunas, N., Memmott, J., Morse, H., Nikolitsi, M., Osgathorpe, L. M., Potts, S. G. and Robertson, K. M. (2016) ‘Food for Pollinators: Quantifying the Nectar and Pollen Resources of Urban Flower Meadows’, *PLoS ONE*, 11(6), p. e0158117. doi: 10.1371/journal.pone.0158117.

HM Government (2018) *A Green Future: Our 25 Year plan to improve the environment*. Available at: https://www.gov.uk/government/publications/25-year-environment-plan.

Hoehn, P., Tscharntke, T., Tylianakis, J. M. and Steffan-Dewenter, I. (2008) ‘Functional group diversity of bee pollinators increases crop yield’, *Proceedings of the Royal Society B*, 275, pp. 2283–2291. doi: 10.1098/rspb.2008.0405.

Hoffmann, R. and Wivstad, M. (2015) *Why do (don’t) we buy organic food and do we get what we bargain for? SLU, EPOK - Centre for Organic Food and Farming*. Uppsala. Available at: www.slu.se/epok/english.

Holland, J. M., Smith, B. M., Storkey, J., Lutman, P. J. W. and Aebischer, N. J. (2015) ‘Managing habitats on English farmland for insect pollinator conservation’, *Biological Conservation*. Elsevier Ltd, 182, pp. 215–222. doi: 10.1016/j.biocon.2014.12.009.

Holzschuh, A., Steffan-Dewenter, I., Kleijn, D. and Tscharntke, T. (2007) ‘Diversity of flower-visiting bees in cereal fields: Effects of farming system, landscape composition and regional context', *Journal of Applied Ecology*, 44(1), pp. 41–49. doi: 10.1111/j.1365-2664.2006.01259.x.

Holzschuh, A., Steffan-dewenter, I. and Tscharntke, T. (2008) 'Agricultural landscapes with organic crops support higher pollinator diversity', *Oikos*, 117, pp. 354–361. doi: 10.1111/j.2007.0030-1299.16303.x.

Honisch, M., Hellmeier, C. and Weiss, K. (2002) 'Response of surface and subsurface water quality to land use changes', *Geoderma*, 105, pp. 277–298. doi: 10.1016/S0016-7061(01)00108-2.

Horn, J., Becher, M. A., Johst, K., Kennedy, P. J., Osborne, J. L., Radchuk, V. and Grimm, V. (2021) 'Honey bee colony performance affected by crop diversity and farmland structure: a modeling framework', *Ecological Applications*, 31(1), pp. 1–22. doi: 10.1002/eap.2216.

Horn, J., Becher, M. A., Kennedy, P. J., Osborne, J. L. and Grimm, V. (2016) 'Multiple stressors: using the honeybee model BEEHAVE to explore how spatial and temporal forage stress affects colony resilience', *Oikos*, 125(7), pp. 1001–1016. doi: 10.1111/oik.02636.

Horrocks, C. A., Dungait, J. A. J., Cardenas, L. M. and Heal, K. V. (2014) 'Does extensification lead to enhanced provision of ecosystems services from soils in UK agriculture?', *Land Use Policy*. Elsevier Ltd, 38, pp. 123–128. doi: 10.1016/j.landusepol.2013.10.023.

Howard, B., Neumann, J. and Riodan, O. (2016) 'Tool Assessor Supporting practical assessment of natural capital in land-use decision making', *JNCC Report 584. JNCC, Peterborough.* Available at: http://jncc.defra.gov.uk/pdf/584_web.pdf.

Howden, N. J. K., Burt, T. P., Worrall, F., Mathias, S. A. and Whelan, M. J. (2013) 'Farming for Water Quality: Balancing Food Security and Nitrate Pollution in UK River Basins', *Annals of the Association of American Geographers*, 103(2), pp. 397–407. doi: 10.1080/00045608.2013.754672.

Hu, Y., Xiang, D., Veresoglou, S. D., Chen, F., Chen, Y., Hao, Z., Zhang, X. and Chen, B. (2014) 'Soil organic carbon and soil structure are driving microbial abundance and community composition across the arid and semi-arid grasslands in northern China', *Soil Biology and Biochemistry*. Elsevier Ltd, 77, pp. 51–57. doi: 10.1016/j.soilbio.2014.06.014.

Humphries, R. N. and Brazier, R. E. (2018) 'Exploring the case for a national-scale soil conservation and soil condition framework for evaluating and reporting on environmental and land use policies', *Soil Use and Management*, 34(1), pp. 134–146. doi: 10.1111/sum.12400.

Hurst, Martin (2019). 'The Green Book: Central Government Guidance on Appraisal and Evaluation'. *Journal of Mega Infrastructure & Sustainable Development,* 1:1, 101-103, DOI: 10.1080/24724718.2019.1607713

Iizumi, T. and Wagai, R. (2019) 'Leveraging drought risk reduction for sustainable food, soil and climate via soil organic carbon sequestration', *Scientific Reports*. Springer US, 9(1), pp. 1–8. doi: 10.1038/s41598-019-55835-y.

Jabłoński, B. & Koltowski, Zbigniew. (2005). Nectar secretion and honey potential of honey-plants growing under Poland's conditions-part XV. Journal of Apicultural Science. 49. 59-63.

Jacobs, A., Poeplau, C., Weiser, C. and Don, A. F. Á. A. (2020) ‘Exports and inputs of organic carbon on agricultural soils in Germany’, *Nutrient Cycling in Agroecosystems*. Springer Netherlands, 5. doi: 10.1007/s10705-020-10087-5.

Jacobs, J. H., Clark, S. J., Denholm, I., Goulson, D., Stoate, C. and Osborne, J. L. (2009) ‘Pollination biology of fruit-bearing hedgerow plants and the role of flower-visiting insects in fruit-set’, *Annals of Botany*, 104(7), pp. 1397–1404. doi: 10.1093/aob/mcp236.

Jarvis, P. and Woolford, A. (2017) *Economic and ecological benefits of reduced tillage in the UK*. Available at: https://www.agricology.co.uk/sites/default/files/Economic and ecological benefits of reduced tillage in the Uk - Final.pdf.

Jenkinson, D. S., Fox, R. H. and Rayner, J. H. (1985) ‘Interactions between fertilizer nitrogen and soil nitrogen-the so-called “ priming ” effect’, *Journal of Soil Science*, 36, pp. 425–444.

Jensen, J. L., Christensen, B. T., Schjønning, P., Watts, C. W., & Munkholm, L. J. (2018) ‘Converting loss-on-ignition to organic carbon content in arable topsoil : pitfalls and proposed procedure’, *European Journal of Soil Science*, 69, pp. 604–612. doi: 10.1111/ejss.12558.

Johnston, A. E., Poulton, P. R., Coleman, K., Macdonald, A. J. and White, R. P. (2017) ‘Changes in soil organic matter over 70 years in continuous arable and ley–arable rotations on a sandy loam soil in England’, *European Journal of Soil Science*, 68, pp. 305–316. doi: 10.1111/ejss.12415.

Jones, J. ., Murphy, J. ., Collins, A. ., Sear, D. ., Naden, P. . and Armitage, P. . (2012) ‘The impact of fine sediment on Macro-Invertebrates’, *River Research and Applications*, 28, pp. 1055–1071. doi: 10.1002/rra.

Judd, A. and Lonsdale, J. A. (2021) ‘Applying systems thinking: The Ecosystem Approach and Natural Capital Approach – Convergent or divergent concepts in marine management?’, *Marine Policy*. Elsevier Ltd, 129, p. 104517. doi: 10.1016/j.marpol.2021.104517.

Kane, D. A., Bradford, M. A., Fuller, E., Oldfield, E. E. and Wood, S. A. (2021) ‘Soil organic matter protects US maize yields and lowers crop insurance payouts under drought’, *Environmental Research Letters*, 16(4). doi: 10.1088/1748-9326/abe492.

Keeler, B. L., Polasky, S., Brauman, K. A., Johnson, K. A., Finlay, J. C., O’Neill, A., Kovacs, K. and Dalzell, B. (2012) ‘Linking water quality and well-being for improved assessment and valuation of ecosystem services’, *PNAS*, 109(45), pp. 18619–18624. doi: 10.1073/pnas.1215991109.

Keith, A. M. and Robinson, D. A. (2012) ‘Earthworms as Natural Capital: Ecosystem Service Providers in Agricultural Soils’, *Economology Journal*, II, pp. 91–99.

Kemp, P., Sear, D., Collins, A., Naden, P. and Jones, I. (2011) 'The impacts of fine sediment on riverine fish', *Hydrological Processes*, 25(11), pp. 1800–1821. doi: 10.1002/hyp.7940.

Kennedy, C. M., Lonsdorf, E., Neel, M. C., Williams, N. M., Ricketts, T. H., Winfree, R., Bommarco, R., Brittain, C., Burley, A. L., Cariveau, D., Carvalheiro, L. G., Chacoff, N. P., Cunningham, S. A., Danforth, B. N., Dudenhöffer, J. H., Elle, E., Gaines, H. R., Garibaldi, L. A., Gratton, C., *et al.* (2013) 'A global quantitative synthesis of local and landscape effects on wild bee pollinators in agroecosystems', *Ecology Letters*, 16(5), pp. 584–599. doi: 10.1111/ele.12082.

Keuskamp, J. A., Dingemans, B. J. J., Lehtinen, T., Sarneel, J. M. and Hefting, M. M. (2013) 'Tea Bag Index: A novel approach to collect uniform decomposition data across ecosystems', *Methods in Ecology and Evolution*, 4(11), pp. 1070–1075. doi: 10.1111/2041-210X.12097.

Khan, J. and Din, F. (2015) *UK Natural Capital – Freshwater Ecosystem Assets and Services Accounts*. Available at: http://webarchive.nationalarchives.gov.uk/20160106100254/http://www.ons.gov.uk/ons/dcp171766_398822.pdf.

Kieboom, E., Silcock, P. and Russ, C. (2018) *Trial of the Natural Capital Protocol for land-based businesses Den Farm Natural Capital Assessment Reviewed by*.

Kirchmann, H., Kätterer, T., Bergström, L., Börjesson, G. and Bolinder, M. A. (2016) 'Flaws and criteria for design and evaluation of comparative organic and conventional cropping systems', *Field Crops Research*. Elsevier B.V., 186, pp. 99–106. doi: 10.1016/j.fcr.2015.11.006.

Klein, A. M., Vaissière, B. E., Cane, J. H., Steffan-Dewenter, I., Cunningham, S. A., Kremen, C. and Tscharntke, T. (2007) 'Importance of pollinators in changing landscapes for world crops', *Proceedings of the Royal Society B: Biological Sciences*, 274(1608), pp. 303–313. doi: 10.1098/rspb.2006.3721.

Knapp, J. L., Becher, M. A., Rankin, C. C., Twiston-Davies, G. and Osborne, J. L. (2019) 'Bombus terrestris in a mass-flowering pollinator-dependent crop: A mutualistic relationship?', *Ecology and Evolution*, 9(1), pp. 609–618. doi: 10.1002/ece3.4784.

Knapp, J. L. and Osborne, J. L. (2017) 'Courgette Production: Pollination Demand, Supply, and Value', *Journal of Economic Entomology*, 110(5), pp. 1973–1979. doi: 10.1093/jee/tox184.

Knight, M. E., Martin, A. P., Bishop, S., Osborne, J. L., Hale, R. J., Sanderson, R. A. and Goulson, D. (2005) 'An interspecific comparison of foraging range and nest density of four bumblebee (Bombus) species', *Molecular Ecology*, 14, pp. 1811–1820. doi: 10.1111/j.1365-294X.2005.02540.x.

Knight, M. E., Osborne, J. L., Sanderson, R. A., Hale, R. J., Martin, A. P. and Goulson, D. (2009) 'Bumblebee nest density and the scale of available forage', *Insect Conservation and Diversity*, 2, pp. 116–124. doi: 10.1111/j.1752-4598.2009.00049.x.

Knudsen, M. T., Kristensen, I. S., Berntsen, J., Petersen, B. M. and Kristensen, E. S. (2006) 'Estimated N leaching losses for organic and conventional farming in Denmark', *The Journal of Agricultural Science*, 144(2), pp. 135–149. doi:

10.1017/S0021859605005812.

Konzmann, S. and Lunau, K. (2014) ‘Divergent rules for pollen and nectar foraging bumblebees - A laboratory study with artificial flowers offering diluted nectar substitute and pollen surrogate’, *PLoS ONE*, 9(3). doi: 10.1371/journal.pone.0091900.

Kopittke, P. M., Menzies, N. W., Wang, P., Mckenna, B. A. and Lombi, E. (2019) ‘Soil and the intensification of agriculture for global food security’, *Environment International*. Elsevier, 132, p. 105078. doi: 10.1016/j.envint.2019.105078.

Kremen, C. and Miles, A. (2012) ‘Ecosystem Services in Biologically Diversified versus Conventional Farming Systems: Benefits, Externalities, and Trade-Offs’, *Ecology and Society*, 17(4), p. 40. doi: 10.5751/ES-05035-170440.

Lal, Rattan (2015) ‘Restoring soil quality to mitigate soil degradation’, *Sustainability*, 7, pp. 5875–5895. doi: 10.3390/su7055875.

Lal, R. (2015) ‘Sequestering carbon and increasing productivity by conservation agriculture’, *Journal of Soil and Water Conservation*, 70(3), pp. 55A-62A. doi: 10.2489/jswc.70.3.55A.

Lal, R. (2016) ‘Soil health and carbon management’, *Food and Energy Security*, 5(4), pp. 212–222. doi: 10.1002/fes3.96.

Lampkin, N. H., Pearce, B. D., Leake, A. R., Creissen, H., Gerrard, C. L., Girling, R., Lloyd, S., Padel, S., Smith, J., Smith, L. G., Vieweger, A. and Wolfe, M. S. (2016) *The Role of Agroecology in Sustainable Intensification. Report for the Land Use Policy Group. Organic Research Centre, Elm Farm and Game and Wildlife Conservation Trust.* Available at: http://www.snh.gov.uk/docs/A1652615.pdf.

Landis, D. A., Gardiner, M. M., van der Werf, W. and Swinton, S. M. (2008) ‘Increasing corn for biofuel production reduces biocontrol services in agricultural landscapes’, *Proceedings of the National Academy of Sciences*, 105(51), pp. 20552–20557. doi: 10.1073/pnas.0804951106.

Landis (2018). ‘0541b Bromsgrove Soil Association’. Available at: https://www.landis.org.uk/services/soilsguide/mapunit.cfm?mu=54102 (Accessed: 18 July 2018).

Latacz-Lohmann, K. and Renwick, A. (2002) ‘An economic evaluation of the Organic Farming Scheme’, *UK Organic Research 2002: Proceedings COR Conference, 26-28th March, Aberystwyth, 311-312*, pp. 1–122.

Leach, K., Palomo, G., Waterfield, W., Zaralis, K. and Padel, S. (2014) ‘Diverse swards and mob grazing for dairy farm productivity: A UK case study’, *4th ISOFAR Scientific Conference. ‘Building Organic Bridges’ at the Organic World Congress*, pp. 1155–1158. Available at: http://www.solidairy.eu/wp-content/uploads/Diverse-swards-and-mob-grazing-for-dairy-farm-productivity-A-UK-case-study..pdf.

Leifeld, J., Angers, D. A., Chenu, C., Fuhrer, J., Kätterer, T. and Powlson, D. S. (2013) ‘Organic farming gives no climate change benefit through soil carbon sequestration’, *Proceedings of the National Academy of Sciences of the United States of America*, 110(11), p. E984. doi: 10.1073/pnas.1220724110.

Lenth, R. (2020). ‘emmeans: Estimated Marginal Means, aka Least-Squares Means’. R package version 1.5.2-1. https://CRAN.R-project.org/package=emmeans

Little, R., Lyon, J., Rose, D., Tsoivalis, J., Escoffie, J., Burns, C. and Hartley, S. (2021) *Written evidence submitted to Parliment (ELM0053)*.

Loaiza-Puerta, V., Pujol, E. I., Wittwer, R., Heijden, M. Van Der and Six, J. (2018) ‘Soil & Tillage Research Improvement of soil structure through organic crop management , conservation tillage and grass-clover ley’, *Soil & Tillage Research*. Elsevier, 180(February), pp. 1–9. doi: 10.1016/j.still.2018.02.007.

Long, J.A. (2020). ‘jtools: Analysis and Presentation of Social Scientific Data’. R package version 2.1.0, <URL:https://cran.r-project.org/package=jtools>.

Lord, E. I. and Sheperd, M. A. (1993) ‘Developments in the use of porous ceramic for measuring nitrate leaching’, *Journal of Soil Science*, 44, pp. 435–449.

Lord, E., Shepherd, M., Silgram, M., Goodlass, G., Gooday, R., Anthony, S. G., Davison, P. and Hodgkinson, R. (2007) *Investigating the effectiveness of NVZ Action Programme measures: Development of a strategy for England. Report for Defra Project No. NIT18.*

Lori, M., Symnaczik, S., Mader, P., Deyn, G. De and Gattinger, A. (2017) ‘Organic farming enhances soil microbial abundance and activity — A meta-analysis and meta-regression’, *PLoS ONE*, 12(7), p. e0180442.

Lotter, D. W. (2003) ‘Organic Agriculture’, *Journal of Sustainable Agriculture*, 21(4), pp. 59–128. doi: https://doi.org/10.1300/J064v21n04_06.

Loveland, P. and Webb, J. (2003) ‘Is there a critical level of organic matter in the agricultural soils of temperate regions: A review’, *Soil and Tillage Research*, 70(1), pp. 1–18. doi: 10.1016/S0167-1987(02)00139-3.

Łuczka, W. and Kalinowski, S. (2020) ‘Barriers to the development of organic farming: A polish case study’, *Agriculture*, 10(536). doi: doi:10.3390/agriculture10110536.

Luscombe, D. J., Anderson, K., Gatis, N., Brazier, R.E. (2022). *Rapid, reproducible, landscape-scale mapping of Trees, Hedgerows and Woodland (THaW), using LiDAR and SAR data.* Manuscript in preparation.

Machmuller, M. B., Kramer, M. G., Cyle, T. K., Hill, N., Hancock, D. and Thompson, A. (2015) ‘Emerging land use practices rapidly increase soil organic matter’, *Nature Communications*, 6, p. 6995. doi: 10.1038/ncomms7995.

Mäder, P., Hahn, D., Dubois, D., Gunst, L., Alföldi, T., Bergmann, H., Oehme, M., Amadò, R., Schneider, H., Graf, U., Velimirov, A., Fließbach, A. and Niggli, U. (2007) ‘Wheat quality in organic and conventional farming: results of a 21 year field experiment’, *Journal of the Science of Food and Agriculture*. John Wiley & Sons, Ltd., 87(10), pp. 1826–1835. doi: 10.1002/jsfa.2866.

Maillard, E. and Angers, D. . (2014) ‘Animal manure application and soil organic carbon stocks : a meta-analysis’, pp. 666–679. doi: 10.1111/gcb.12438.

Makó, A., Kocsis, M., Barna, G. and Tóth, G. (2017) *Mapping the storing and filtering capacity of European soils; EUR 28392*. doi: doi:10.2788/49218.

Man, M., Deen, B., Dunfield, K. E., Wagner-Riddle, C. and Simpson, M. J. (2021) 'Altered soil organic matter composition and degradation after a decade of nitrogen fertilization in a temperate agroecosystem', *Agriculture, Ecosystems and Environment*. Elsevier B.V., 310, p. 107305. doi: 10.1016/j.agee.2021.107305.

Manning, P., Slade, E. M., Beynon, S. A. and Lewis, O. T. (2016) 'Functionally rich dung beetle assemblages are required to provide multiple ecosystem services', *Agriculture, Ecosystems and Environment*. Elsevier B.V., 218, pp. 87–94. doi: 10.1016/j.agee.2015.11.007.

Mansion-Vaquié, A., Ferrante, M., Cook, S. M., Pell, J. K. and Lövei, G. L. (2017) 'Manipulating field margins to increase predation intensity in fields of winter wheat (Triticum aestivum)', *Journal of Applied Entomology*, 141(8), pp. 600–611. doi: 10.1111/jen.12385.

Marshall, M. R., Ballard, C. E., Frogbrook, Z. L., Solloway, I., Mcintyre, N., Reynolds, B. and Wheater, H. S. (2014) 'The impact of rural land management changes on soil hydraulic properties and runoff processes: Results from experimental plots in upland UK', *Hydrological Processes*, 28(4), pp. 2617–2629. doi: 10.1002/hyp.9826.

Martinez-Garcia, L. B., Korthals, G. W., Brussaard, L., Mainardi, G. and Deyn, G. B. De (2021) 'Litter quality drives nitrogen release , and agricultural management (organic vs . conventional) drives carbon loss during litter decomposition in agro-ecosystems', *Soil Biology and Biochemistry*. Elsevier Ltd, 153, p. 108115. doi: 10.1016/j.soilbio.2020.108115.

Maseyk, F. J. F., Mackay, A. D., Possingham, H. P., Dominati, E. J. and Buckley, Y. M. (2017) 'Managing Natural Capital Stocks for the Provision of Ecosystem Services', *Conservation Letters*, 10(2), pp. 211–220. doi: 10.1111/conl.12242.

McBratney, A. B., Morgan, C. L. S. and Jarrett, L. E. (2017) 'The Value of Soil's Contributions to Ecosystem Services. In: Field D.J., Morgan C.L.S., McBratney A.B. (eds) Global Soil Security.', *Progress in Soil Science*. Cham: Springer, pp. 227–235. doi: 10.1007/978-3-319-43394-3_20.

McDaniel, M. D., Tiemann L.K. and Grandy, A. S. (2014) 'Does agricultural crop diversity enhance soil microbial biomass and organic matter dynamics ? A meta-analysis', *Ecological Society of America*, 24(3), pp. 560–570.

MEA (2005). Millenium Ecosystem Assessment: Ecosystems and human well-being: Synthesis. Washinton, DC. Island Press.

Merante, P., Dibari, C., Ferrise, R., Sánchez, B., Iglesias, A., Lesschen, J. P., Kuikman, P., Yeluripati, J., Smith, P. and Bindi, M. (2017) 'Adopting soil organic carbon management practices in soils of varying quality: Implications and perspectives in Europe', *Soil and Tillage Research*. Elsevier B.V., 165, pp. 95–106. doi: 10.1016/j.still.2016.08.001.

Met Office. (2003). 'Met Office rain radar data from the NIMROD system'. Retrieved from http://catalogue.ceda.ac.uk/uuid/82adec1f896a

Minasny, B., Malone, B. P., McBratney, A. B., Angers, D. A., Arrouays, D., Chambers, A., Chaplot, V., Chen, Z. S., Cheng, K., Das, B. S., Field, D. J., Gimona, A., Hedley, C. B., Hong, S. Y., Mandal, B., Marchant, B. P., Martin, M., McConkey, B. G., Mulder, V. L., *et al.* (2017) 'Soil carbon 4 per mille', *Geoderma*.

Elsevier B.V., 292, pp. 59–86. doi: 10.1016/j.geoderma.2017.01.002.

Miner, G. L., Stewart, C. E., Ippolito, J. A. and Steward, C. E. (2020) ‘Soil health management practices and crop productivity’, *Agricultural and Environmental Letters*, 5, p. e20023. doi: 10.1002/ael2.20023.

Mondelaers, K., Aertsens, J. and Huylenbroeck, G. Van (2009) ‘A meta-analysis of the differences in environmental impacts between organic and conventional farming’, *British Food Journal*, 111(10), pp. 1098–1119. doi: 10.1108/00070700910992925.

Muller, A., Diener, S., Schnyder, S., Stutz, K., Sedivy, C. and Dorn, S. (2006) ‘Quantitative pollen requirements of solitary bees : Implications for bee conservation and the evolution of bee – flower relationships’, *Biological Conservation*, 130, pp. 604–615. doi: 10.1016/j.biocon.2006.01.023.

Muller, A., Schader, C., El-Hage Scialabba, N., Brüggemann, J., Isensee, A., Erb, K. H., Smith, P., Klocke, P., Leiber, F., Stolze, M. and Niggli, U. (2017) ‘Strategies for feeding the world more sustainably with organic agriculture’, *Nature Communications*. Springer US, 8(1), pp. 1–13. doi: 10.1038/s41467-017-01410-w.

Mwebaze, P., Marris, G. C., Brown, M., MacLeod, A., Jones, G. and Budge, G. E. (2018) ‘Measuring public perception and preferences for ecosystem services: A case study of bee pollination in the UK’, *Land Use Policy*. Elsevier, 71, pp. 355–362. doi: 10.1016/j.landusepol.2017.11.045.

Nakagawa, S. and Schielzeth, H. (2013) ‘A general and simple method for obtaining R2 from generalized linear mixed-effects models’, *Methods in Ecology and Evolution*, 4, pp. 133–142. doi: 10.1111/j.2041-210x.2012.00261.x.

Natural Capital Coalition (2020) *What is a Natural Capital Approach?* Available at: https://naturalcapitalcoalition.org/wp-content/uploads/2019/06/NCC-WhatIs-NaturalCapitalApproach-FINAL.pdf (Accessed: 30 November 2021).

Natural Capital Coalition (2021) *Natural Capital Protocol*.

Natural Capital Committee (2017) *How to do it: a natural capital workbook. Version 1*. Available at: https://www.gov.uk/government/uploads/system/uploads/attachment_data/file/608852/ncc-natural-capital-workbook.pdf.

Natural Capital Committee (2019) *Advice on soil management*.

Nichols, R. N., Goulson, D. and Holland, J. M. (2019) ‘The best wildflowers for wild bees’, *Journal of Insect Conservation*. Springer International Publishing, 23(5), pp. 819–830. doi: 10.1007/s10841-019-00180-8.

Nix, (2018). ‘The John Nix Pocketbook for Farm Management, 49th Edition (2019)’. Melton Mowbray: Agro Business Consultants

North, A. C., Hodgson, D. J., Price, S. J. and Griffiths, A. G. F. (2015) ‘Anthropogenic and ecological drivers of amphibian disease (ranavirosis)’, *PLoS ONE*, 10(6), pp. 1–17. doi: 10.1371/journal.pone.0127037.

Norton, E. (2021) *Does natural capital provide a mechanism for making ‘good farming’ more investible?*, *Savills Blog*. Available at:

https://www.savills.co.uk/blog/article/292757/rural-property/does-natural-capital-provide-a-mechanism-for-making-good-farming-more-investible-.aspx (Accessed: 21 December 2021).

NSRI (2005). 'Soil Survey of England and Wales, 1983. National Soil Map NATMAP Vector 1:250,000'. National Soil Research Institute: http://www.landis.org.uk/data/

O'Sullivan, L., Creamer, R. E., Fealy, R., Lanigan, G., Simo, I., Fenton, O., Carfrae, J. and Schulte, R. P. O. (2015) 'Functional Land Management for managing soil functions: A case-study of the trade-off between primary productivity and carbon storage in response to the intervention of drainage systems in Ireland', *Land Use Policy*. Elsevier Ltd, 47, pp. 42–54. doi: 10.1016/j.landusepol.2015.03.007.

Obade, V. de P. and Lal, R. (2014) 'Soil quality evaluation under different land management practices', *Environmental Earth Sciences*, 72(11), pp. 4531–4549. doi: 10.1007/s12665-014-3353-z.

Obade, V. de P. and Lal, R. (2016) 'A standardized soil quality index for diverse field conditions', *Science of the Total Environment*. Elsevier B.V., 541, pp. 424–434. doi: 10.1016/j.scitotenv.2015.09.096.

Ohm, M., Paulsen, H. M., Moos, J. H. and Eichler-löbermann, B. (2017) 'Long-term negative phosphorus budgets in organic crop rotations deplete plant-available phosphorus from soil'. Agronomy for Sustainable Development. doi: 10.1007/s13593-017-0425-y.

Oldfield, E. E., Bradford, M. A. and Wood, S. A. (2019) 'Global meta-analysis of the relationship between soil organic matter and crop yields', *SOIL*, 5, pp. 15–32.

Ollerton, J., Erenler, H., Edwards, M. and Crockett, R. (2014) 'Extinctions of aculeate pollinators in Britain and the role of large-scale agricultural changes', *Science*, 346(6215), pp. 1360–1362. doi: 10.1017/cbo9780511563478.010.

Ollerton, J., Winfree, R. and Tarrant, S. (2011) 'How many flowering plants are pollinated by animals?', *Oikos*, 120(3), pp. 321–326. doi: 10.1111/j.1600-0706.2010.18644.x.

Olson, K. R. and Al-Kaisi, M. M. (2015) 'The importance of soil sampling depth for accurate account of soil organic carbon sequestration, storage, retention and loss', *Catena.* Elsevier B.V., 125, pp. 33–37. doi: 10.1016/j.catena.2014.10.004.

Ovando, P. (2020) *Application of the Natural Capital Protocol at Glensaugh Farm*. Available at: https://www.hutton.ac.uk/research/projects/application-natural-capital-protocol-glensaugh-farm.

Ovando, P. (2021) *Trial of the Natural Capital Protocol for land-based businesses: Glenlivet Estate Natural Capital Assessment Final Report.*

ONS (2020). Office for National Statistics; Index of Labour Costs per Hour, non-seasonally adjusted. Available at: https://www.ons.gov.uk/employmentandlabourmarket/peopleinwork/earningsandworkinghours/datasets/indexoflabourcostsperhourilchnonseasonallyadjusted (Accessed: 30 September 2020).

Ordnance Survey (2020). 'OS Vector Local Dataset, updated 8 October 2020'. Downloaded from: https://digimap.edina.ac.uk (Accessed: 12 February 2021).

ORC, (2017). 'Organic Farm Management Handbook, 11th Edition (January 2017)'.

ORC (2021). EU Targets 25% organic land in Europe by 2030. Available at: https://www.organicresearchcentre.com/news-events/news/eu-targets-25-organic-land-in-europe-by-2030/ (Accessed: 30 December 2021).

Owayss, A. A. *et al.* (2020) 'Phacelia tanacetifolia can enhance conservation of honey bees and wild bees in the drastic hot-arid subtropical Central Arabia', *Journal of Apicultural Research*. Taylor & Francis, 59(4), pp. 569–582. doi: 10.1080/00218839.2020.1735731.

Pagliai, M., Vignozzi, N. and Pellegrini, S. (2004) 'Soil structure and the effect of management practices', 79, pp. 131–143. doi: 10.1016/j.still.2004.07.002.

Patrignani, A. and Ochsner, T. E. (2015) 'Canopeo: A Powerful New Tool for Measuring Fractional Green Canopy Cover', *Agronomy Journal*, 107, pp. 2312–2320. doi: 10.2134/agronj15.0150.

Paustian, K., Collier, S., Baldock, J., Burgess, R., Creque, J., DeLonge, M., Dungait, J., Ellert, B., Frank, S., Goddard, T., Govaerts, B., Grundy, M., Henning, M., Izaurralde, R. C., Madaras, M., McConkey, B., Porzig, E., Rice, C., Searle, R., *et al.* (2019) 'Quantifying carbon for agricultural soil management: from the current status toward a global soil information system', *Carbon Management*. Taylor & Francis, 10(6), pp. 567–587. doi: 10.1080/17583004.2019.1633231.

Paustian, K., Larson, E., Kent, J., Marx, E. and Swan, A. (2019) 'Soil C Sequestration as a Biological Negative Emission Strategy', *Frontiers in Climate*, 1(8), pp. 1–11. doi: 10.3389/fclim.2019.00008.

Perl, T, Roy, S, Thompson, P, and Lees, J (2004). 'Groundwater Quality Review: Otter Sandstone. Analysis of the Otter Sandstone Groundwater Chemistry. Environment Agency Report GWQR019'.

Pernal, S. F. and Currie, R. W. (2000) 'Pollen quality of fresh and 1-year-old single pollen diets for worker honey bees (Apis mellifera L.)', *Apidologie*, 31(3), pp. 387–409. doi: 10.1051/apido:2000130.

Perrings, C., Jackson, L., Bawa, K., Brussaard, L., Brush, S., Gavin, T., Papa, R., Pascual, U. and De Ruiter, P. (2006) 'Biodiversity in agricultural landscapes: Saving natural capital without losing interest', *Conservation Biology*, 20(2), pp. 263–264. doi: 10.1111/j.1523-1739.2006.00390.x.

Petersen, B. and Snapp, S. (2015) 'What is sustainable intensification? Views from experts', *Land Use Policy*. Elsevier Ltd, 46, pp. 1–10. doi: 10.1016/j.landusepol.2015.02.002.

Peukert, S., Bol, R., Roberts, W., Macleod, C. J. A., Murray, P. J., Dixon, E. R. and Brazier, R. E. (2012) 'Understanding spatial variability of soil properties: A key step in establishing field- to farm-scale agro-ecosystem experiments', *Rapid Communications in Mass Spectrometry*, 26(20), pp. 2413–2421. doi: 10.1002/rcm.6336.

Peukert, S., Griffith, B. A., Murray, P. J., Macleod, C. J. A. and Brazier, R. E.

(2014) ‘Intensive Management in Grasslands Causes Diffuse Water Pollution at the Farm Scale’, *Journal of Environment Quality*, 43(6), p. 2009. doi: 10.2134/jeq2014.04.0193.

Pilgrim, E. S., Macleod, C. J. A., Blackwell, M. S. A., Bol, R., Hogan, D. V, Chadwick, D. R., Cardenas, L., Misselbrook, T. H., Haygarth, P. M., Brazier, R. E., Hobbs, P., Hodgson, C., Jarvis, S., Dungait, J., Murray, P. J. and Firbank, L. G. (2010) *Interactions Among Agricultural Production and Other Ecosystem Services Delivered from European Temperate Grassland Systems*, *Advances in Agronomy*. Burlington: Elsevier Inc. doi: 10.1016/S0065-2113(10)09004-8.

Poeplau, C., Vos, C. and Don, A. (2017) ‘Soil organic carbon stocks are systematically overestimated by misuse of the parameters bulk density and rock fragment content’, *SOIL*, 3, pp. 61–66. doi: 10.5194/soil-3-61-2017.

Ponisio, L. C., M’Gonigle, L. K., Mace, K. C., Palomino, J., de Valpine, P. and Kremen, C. (2014) ‘Diversification practices reduce organic to conventional yield gap’, *Proceedings of the Royal Society B: Biological Sciences*, 282(1799), pp. 20141396–20141396. doi: 10.1098/rspb.2014.1396.

De Ponti, T., Rijk, B. and Van Ittersum, M. K. (2012) ‘The crop yield gap between organic and conventional agriculture’, *Agricultural Systems*. Elsevier Ltd, 108, pp. 1–9. doi: 10.1016/j.agsy.2011.12.004.

Porter, J., Costanza, R., Sandhu, H. S., Sigsgaard, L. and Wratten, S. D. (2009) ‘The Value of Producing Food , Energy , and Ecosystem Services within an Agro-Ecosystem’, *AMBIO A Journal of the Human Environment*, 38(4), pp. 186–193. doi: 10.1579/0044-7447-38.4.186.

Porto, R. G., de Almeida, R. F., Cruz-Neto, O., Tabarelli, M., Viana, B. F., Peres, C. A. and Lopes, A. V. (2020) ‘Pollination ecosystem services: A comprehensive review of economic values, research funding and policy actions’, *Food Security*, 12, pp. 1425–1442. doi: 10.1007/s12571-020-01043-w.

Post, M. and Kwon, K. C. (2000) ‘Soil Carbon Sequestration and Land-Use Change : Processes and Potential’.

Potts, S. G., Biesmeijer, J. C., Kremen, C., Neumann, P., Schweiger, O. and Kunin, W. E. (2010) ‘Global pollinator declines: Trends, impacts and drivers’, *Trends in Ecology and Evolution*. Elsevier Ltd, 25(6), pp. 345–353. doi: 10.1016/j.tree.2010.01.007.

Poulton, P., Johnston, J., Macdonald, A., White, R. and Powlson, D. (2018) ‘Major limitations to achieving “4 per 1000” increases in soil organic carbon stock in temperate regions: Evidence from long-term experiments at Rothamsted Research, United Kingdom’, *Global Change Biology*, 24, pp. 2563–2584. doi: 10.1111/gcb.14066.

Power, A. G. (2010) ‘Ecosystem services and agriculture: tradeoffs and synergies’, *Philosophical Transactions of the Royal Society B: Biological Sciences*, 365(1554), pp. 2959–2971. doi: 10.1098/rstb.2010.0143.

Powlson, D. S., Bhogal, A., Chambers, B. J., Coleman, K., Macdonald, A. J., Goulding, K. W. T. and Whitmore, A. P. (2012) ‘The potential to increase soil carbon stocks through reduced tillage or organic material additions in England and Wales: A case study’, *Agriculture, Ecosystems and Environment*. Elsevier

B.V., 146(1), pp. 23–33. doi: 10.1016/j.agee.2011.10.004.

Prager, K. and McKee, A. (2014) *Use and awareness of soil data and information among local authorities, farmers and estate managers.* Available at: http://www.hutton.ac.uk/research/themes/realising-lands-potential/land-manager-attitudes-and-behaviours.

Prechsl, U. E., Wittwer, R., van der Heijden, M. G. A., Lüscher, G., Jeanneret, P. and Nemecek, T. (2017) 'Assessing the environmental impacts of cropping systems and cover crops: Life cycle assessment of FAST, a long-term arable farming field experiment', *Agricultural Systems*. Elsevier, 157, pp. 39–50. doi: 10.1016/j.agsy.2017.06.011.

Pretty, J., Brett, C., Gee, D., Hine, R., Mason, C., Morison, J., Raven, H. and van der Bijl, G. (2000) 'An Assessment of the Total External Costs of UK Agriculture', *Agricultural Systems*, 65(2), pp. 133–136. Available at: http://www.julespretty.com/wp-content/uploads/2013/09/1.-AgSyst-pdf.pdf.

Pretty, J. N., Mason, C. F., Nedwell, D. B., Hine, R. E., Leaf, S. and Dils, R. (2003) 'Environmental costs of freshwater eutrophication in England and Wales', *Environmental Science and Technology*, 37(2), pp. 201–208. doi: 10.1021/es020793k.

Prout, J. M., Shepherd, K. D., McGrath, S. P., Kirk, G. J. D. and Haefele, S. M. (2020) 'What is a good level of soil organic matter? An index based on organic carbon to clay ratio', *European Journal of Soil Science*, pp. 1–11. doi: 10.1111/ejss.13012.

Puttock, A., Graham, H. A., Ashe, J., Luscombe, D. J. and Brazier, R. E. (2021) 'Beaver dams attenuate flow: A multi-site study', *Hydrological Processes*, 35(2), pp. 1–18. doi: 10.1002/hyp.14017.

Pywell, R. F., Heard, M. S., Woodcock, B. A., Hinsley, S., Ridding, L., Nowakowski, M. and Bullock, J. M. (2015) 'Wildlife-friendly farming increases crop yield: evidence for ecological intensification', *Proceedings of the Royal Society B: Biological Sciences*, 282(1816), p. 20151740. doi: 10.1098/rspb.2015.1740.

Pywell, R. F., Warman, E. A., Carvell, C., Sparks, T. H., Dicks, L. V., Bennett, D., Wright, A., Critchley, C. N. R. and Sherwood, A. (2005) 'Providing foraging resources for bumblebees in intensively farmed landscapes', *Biological Conservation*, 121(4), pp. 479–494. doi: 10.1016/j.biocon.2004.05.020.

Pywell, R. F., Warman, E. A., Hulmes, L., Hulmes, S., Nuttall, P., Sparks, T. H., Critchley, C. N. R. and Sherwood, A. (2006) 'Effectiveness of new agri-environment schemes in providing foraging resources for bumblebees in intensively farmed landscapes', *Biological Conservation*, 129(2), pp. 192–206. doi: 10.1016/j.biocon.2005.10.034.

Rabot, E., Wiesmeier, M., Schlüter, S. and Vogel, H.-J. (2018) 'Soil structure as an indicator of soil functions: A review', *Geoderma*. Elsevier, 314, pp. 122–137. doi: 10.1016/j.geoderma.2017.11.009.

Rakotovololona, L., Beaudoin, N., Ronceux, A., Venet, E., Mary, B., Agroimpact, I. U. R. and Bugny, B. (2019) 'Driving factors of nitrate leaching in arable organic cropping systems in Northern France', *Agriculture, Ecosystems and*

Environment. Elsevier, 272, pp. 38–51. doi: 10.1016/j.agee.2018.10.005.

Ramshorst, J. van, Markwitz, C., Hill, T., Clement, R., Knol, A. and Siebicke, L. (2020) *First application of low-cost eddy covariance for CO2 fluxes over agroforestry*, *EGU2020-10329*. Available at: https://presentations.copernicus.org/EGU2020/EGU2020-10329_presentation.pdf.

Ray, R. L., Griffin, R. W., Fares, A., Elhassan, A., Awal, R., Woldesenbet, S. and Risch, E. (2020) 'Soil CO 2 emission in response to organic amendments , temperature , and rainfall', *Scientific Reports*. Springer US, pp. 1–14. doi: 10.1038/s41598-020-62267-6.

Rickson, R., Deeks, L., Corstanje, R., Newell-Price, P., Kibblewhite, M., Chambers, B., Bellamy, P., Holman, I., James, I., Jones, R., Kechavarsi, C., Mouazen, A., Ritz, K. and Waine, T. (2012) 'Indicators of soil quality - physical properties (SP1611). Final Report to DEFRA', pp. 1–45. Available at: http://randd.defra.gov.uk/Default.aspx?Module=More&Location=None&ProjectID=17595.

Ritz, K., Black, H. I. J., Campbell, C. D., Harris, J. A. and Wood, C. (2009) 'Selecting biological indicators for monitoring soils: A framework for balancing scientific and technical opinion to assist policy development', *Ecological Indicators*, 9(6), pp. 1212–1221. doi: 10.1016/j.ecolind.2009.02.009.

Robinson, D. A., Hockley, N., Cooper, D. M., Emmett, B. A., Keith, A. M., Lebron, I., Reynolds, B., Tipping, E., Tye, A. M., Watts, C. W., Whalley, W. R., Black, H. I. J., Warren, G. P. and Robinson, J. S. (2013) 'Natural capital and ecosystem services, developing an appropriate soils framework as a basis for valuation', *Soil Biology and Biochemistry*. Elsevier Ltd, 57, pp. 1023–1033. doi: 10.1016/j.soilbio.2012.09.008.

Robinson, D. A., Hockley, N., Dominati, E., Lebron, I., Scow, K. M., Reynolds, B., Emmett, B. A., Keith, A. M., de Jonge, L. W., Schjønning, P., Moldrup, P., Jones, S. B. and Tuller, M. (2012) 'Natural Capital, Ecosystem Services, and Soil Change: Why Soil Science Must Embrace an Ecosystems Approach', *Vadose Zone Journal*, 11(1). doi: 10.2136/vzj2011.0051.

Robinson, D. A., Lebron, I. and Vereecken, H. (2009) 'On the Definition of the Natural Capital of Soils: A Framework for Description, Evaluation, and Monitoring', *Soil Science Society of America Journal*, 73(6), p. 1904. doi: 10.2136/sssaj2008.0332.

Robinson, D. A., Panagos, P., Borrelli, P., Jones, A., Montanarella, L., Tye, A. and Obst, C. G. (2017) 'Soil natural capital in europe; a framework for state and change assessment', *Scientific Reports*. Springer US, 7(1), p. 6706. doi: 10.1038/s41598-017-06819-3.

Robinson, R. A. and Sutherland, W. J. (2002) 'Post-war changes in arable farming and biodiversity in Great Britain', *Journal of Applied Ecology*, 39, pp. 157–176.

Rodríguez, J. P., Beard, T. D., Bennett, E. M., Cumming, G. S., Cork, S. J., Agard, J., Dobson, A. P. and Peterson, G. D. (2006) 'Trade-offs across space, time, and ecosystem services', *Ecology and Society*, 11(1). doi: 10.5751/ES-01667-110128.

Rogger, M., Agnoletti, M., Alaoui, A., Bathurst, J. C., Bodner, G., Borga, M., Chaplot, V., Gallart, F., Glatzel, G., Hall, J., Holden, J., Holko, L., Horn, R., Kiss, A., Quinton, J. N., Leitinger, G., Lennartz, B., Parajka, J., Peth, S., *et al.* (2017) 'Land use change impacts on floods at the catchment scale: Challenges and opportunities for future research', *Water Resouces Research*, 53, pp. 5209–5219. doi: 10.1002/2017WR020723.

Rollin, O., Benelli, G., Benvenuti, S., Decourtye, A., Wratten, S. D., Canale, A. and Desneux, N. (2016) 'Weed-insect pollinator networks as bio-indicators of ecological sustainability in agriculture. A review', *Agronomy for Sustainable Development*, 36(8), pp. 1–22. doi: 10.1007/s13593-015-0342-x.

Röös, E., Mie, A., Wivstad, M., Salomon, E., Johansson, B., Gunnarsson, S., Wallenbeck, A., Hoffmann, R., Nilsson, U., Sundberg, C. and Watson, C. A. (2018) 'Risks and opportunities of increasing yields in organic farming. A review', *Agronomy for Sustainable Development*. Agronomy for Sustainable Development, 38(14), pp. 1–21.

Roulston, T. H., Cane, J. H. and Buchmann, S. L. (2000) 'What governs protein content of pollen: Pollinator preferences, pollen-pistil interactions, or phylogeny?', *Ecological Monographs*, 70(4), pp. 617–643. doi: 10.1890/0012-9615(2000)070[0617:WGPCOP]2.0.CO;2.

Roulston, T. H. and Goodell, K. (2011) 'The role of resources and risks in regulating wild bee populations', *Annual Review of Entomology*, 56, pp. 293–312. doi: 10.1146/annurev-ento-120709-144802.

Rundlöf, M., Nilsson, H. and Smith, H. G. (2008) 'Interacting effects of farming practice and landscape context on bumble bees', *Biological Conservation*, 141(2), pp. 417–426. doi: 10.1016/j.biocon.2007.10.011.

Sandhu, H. S., Wratten, S. D., Cullen, R. and Case, B. (2008) 'The future of farming: The value of ecosystem services in conventional and organic arable land. An experimental approach', *Ecological Economics*, 64(4), pp. 835–848. doi: 10.1016/j.ecolecon.2007.05.007.

Sandhu, H., Wratten, S., Costanza, R. and Pretty, J. (2015) 'Significance and value of non-traded ecosystem services on farmland', *PeerJ*, 3(e762), pp. 1–22. doi: 10.7717/peerj.762.

Sandhu, H., Wratten, S., Costanza, R., Pretty, J., Porter, J. R. and Reganold, J. (2015) 'Significance and value of non-traded ecosystem services on farmland', *PeerJ*, 3, p. e762. doi: 10.7717/peerj.762.

Scarlat, N., Martinov, M. and Dallemand, J. (2010) 'Assessment of the availability of agricultural crop residues in the European Union : Potential and limitations for bioenergy use', *Waste Management*. Elsevier Ltd, 30(10), pp. 1889–1897. doi: 10.1016/j.wasman.2010.04.016.

Scheper, J., Holzschuh, A., Kuussaari, M., Potts, S. G., Rundlöf, M., Smith, H. G. and Kleijn, D. (2013) 'Environmental factors driving the effectiveness of European agri-environmental measures in mitigating pollinator loss - a meta-analysis', *Ecology Letters*, 16(7), pp. 912–920. doi: 10.1111/ele.12128.

Schipanski, M. E., Barbercheck, M., Douglas, M. R., Finney, D. M., Haider, K., Kaye, J. P., Kemanian, A. R., Mortensen, D. A., Ryan, M. R., Tooker, J. and

White, C. (2014) ‘A framework for evaluating ecosystem services provided by cover crops in agroecosystems’, *Agricultural Systems*. Elsevier Ltd, 125, pp. 12–22. doi: 10.1016/j.agsy.2013.11.004.

Scott, C. (2020) *Farm Business Survey 2018/2019: Organic Farming in England*.

Seufert, V., Ramankutty, N. and Foley, J. A. (2012) ‘Comparing the yields of organic and conventional agriculture’, *Nature*, 485, pp. 2–7. doi: 10.1038/nature11069.

Shah, A., Askegaard, M., Rasmussen, I. A., Jimenez, E. M. C. and Olesen, J. E. (2017) ‘Productivity of organic and conventional arable cropping systems in long-term experiments in Denmark’, *European Journal of Agronomy*. Elsevier, 90, pp. 12–22. doi: 10.1016/j.eja.2017.07.001.

Sharma, P., Laor, Y., Raviv, M., Medina, S., Saadi, I., Krasnovsky, A., Vager, M., Levy, G. J., Bar-Tal, A. and Borisover, M. (2017) ‘Green manure as part of organic management cycle: Effects on changes in organic matter characteristics across the soil profile’, *Geoderma*, 305, pp. 197–207. doi: 10.1016/j.geoderma.2017.06.003.

Sharpley, A. (2016) ‘Managing agricultural phosphorus to minimize water quality impacts’, 2014(February), pp. 1–8.

Silcock, P., Rowcroft, P., Keiboom, E., Dunscombe, R. and Russ, C. (2018) *Trial of the Natural Capital Protocol for land-based businesses. Overview Report for Crown Estate Scotland.*

Silcock, P. and Russ, C. (2018) *Trial of the Natural Capital Protocol for land-based businesses Glenlivet Estate Natural Capital Assessment Final Report*.

Silgram, M. and Chambers, B. J. (2002) ‘Effects of long-term straw management and fertilizer nitrogen additions on soil nitrogen supply and crop yields at two sites in eastern England’, *Journal of Agricultural Science*, 139(2), pp. 115–127. doi: 10.1017/S0021859602002435.

Smith, A. (2013). ‘Digital Geological Map of Great Britain, information notes, 2013’. *British Geological Survey Open Report*, OR/13/007. 54pp.

Smith, A. C., Harrison, P. A., Pérez Soba, M., Archaux, F., Blicharska, M., Egoh, B. N., Erős, T., Fabrega Domenech, N., György, I., Haines-Young, R., Li, S., Lommelen, E., Meiresonne, L., Miguel Ayala, L., Mononen, L., Simpson, G., Stange, E., Turkelboom, F., Uiterwijk, M., *et al.* (2017) ‘How natural capital delivers ecosystem services: A typology derived from a systematic review’, *Ecosystem Services*, 26, pp. 111–126. doi: 10.1016/j.ecoser.2017.06.006.

Smith, K. A., Jackson, D. R. and Withers, P. J. A. (2001) ‘Nutrient losses by surface run-off following the application of organic manures to arable land. 2. Phosphorus’, *Environmental Pollution*, 112(1), pp. 53–60. doi: 10.1016/S0269-7491(00)00098-1.

Smith, O. M., Cohen, A. L., Rieser, C. J., Davis, A. G., Taylor, J. M., Adesanya, A. W., Jones, M. S., Meier, A. R., Reganold, J. P., Orpet, R. J., Northfield, T. D. and Crowder, D. W. (2019) ‘Organic Farming Provides Reliable Environmental Benefits but Increases Variability in Crop Yields: A Global Meta-Analysis’, *Frontiers in Sustainable Food Systems*, 3(62), pp. 1–10. doi:

10.3389/fsufs.2019.00082.

Smith, P., Ashmore, M., Black, H., Burgess, P., Evans, C., Hails, R., Potts, S., Quine, T., Thomson, A., Biesmeijer, K., Breeze, T., Broadmeadow, M., Ferrier, R., Freer, J., Hansom, J., Haygarth, P., Hesketh, H., Hicks, K., Johnson, A., *et al.* (2011) *Regulating services - chapter 14*, *Reports from the UK National Ecosystem Assessment*. Available at: http://uknea.unep-wcmc.org/Resources/tabid/82/Default.aspx.

Smith, P., House, J. I., Bustamante, M., Sobocká, J., Harper, R., Pan, G., West, P. C., Clark, J. M., Adhya, T., Rumpel, C., Paustian, K., Kuikman, P., Cotrufo, M. F., Elliott, J. A., Mcdowell, R., Griffiths, R. I., Asakawa, S., Bondeau, A., Jain, A. K., *et al.* (2016) 'Global change pressures on soils from land use and management', *Global Change Biology*, 22(3), pp. 1008–1028. doi: 10.1111/gcb.13068.

Smukler, S. M., Sánchez-Moreno, S., Fonte, S. J., Ferris, H., Klonsky, K., O'Geen, A. T., Scow, K. M., Steenwerth, K. L. and Jackson, L. E. (2010) 'Biodiversity and multiple ecosystem functions in an organic farmscape', *Agriculture, Ecosystems and Environment*. Elsevier B.V., 139, pp. 80–97. doi: 10.1016/j.agee.2010.07.004.

Snapp, S. S., Gentry, L. E. and Harwood, R. (2010) 'Management intensity - not biodiversity - the driver of ecosystem services in a long-term row crop experiment', *Agriculture, Ecosystems and Environment*. Elsevier B.V., 138, pp. 242–248. doi: 10.1016/j.agee.2010.05.005.

Soil Association (2021a) *EU Equivalent Organic Standards: Farming and growing*.

Soil Association (2021b) *Will the Sustainable Farming Incentive (SFI) deliver for organic?* Available at: https://www.soilassociation.org/blogs/2021/december/16/will-the-sustainable-farming-incentive-sfi-deliver-for-organic/ (Accessed: 2 January 2022).

Southern Water (2018). TA 11.WN02 Nitrate Business Case, September 2018, Version 1.0. Southern Water. Available at: https://www.southernwater.co.uk/media/1945/ta11wn02-business-case-nitrate_redacted.pdf (Accessed: 30 December 2021).

Squire, G. R., Hawes, C., Valentine, T. A. and Young, M. W. (2015) 'Degradation rate of soil function varies with trajectory of agricultural intensification', *Agriculture, Ecosystems and Environment*. Elsevier B.V., 202, pp. 160–167. doi: 10.1016/j.agee.2014.12.004.

Shrubsole, G (2019) Who Owns England? How We Lost Our Land and How to Take It Back. HarperCollins Publishers

Stagnari, F., Maggio, A., Galieni, A. and Pisante, M. (2017) 'Multiple benefits of legumes for agriculture sustainability: an overview', *Chemical and Biological Technologies in Agriculture*. Springer International Publishing, 4(1), pp. 1–13. doi: 10.1186/s40538-016-0085-1.

Stampa, E., Schipmann-Schwarze, C. and Hamm, U. (2020) 'Consumer perceptions, preferences, and behavior regarding pasture-raised livestock products: A review', *Food Quality and Preference*. Elsevier, 82, p. 103872. doi:

10.1016/j.foodqual.2020.103872.

Stein-Bachinger, K., Gottwald, F., Haub, A. and Schmidt, E. (2021) ‘To what extent does organic farming promote species richness and abundance in temperate climates? A review’, *Organic Agriculture*, 11, pp. 1–12. doi: 10.1007/s13165-020-00279-2.

Steinbeiss, S., Bebler, H., Engels, C., Temperton, V. M., Buchmann, N., Roscher, C., Kreutziger, Y., Baade, J., Habekost, M. and Gleixner, G. (2008) ‘Plant diversity positively affects short-term soil carbon storage in experimental grasslands’, *Global Change Biology*, 14, pp. 2937–2949. doi: 10.1111/j.1365-2486.2008.01697.x.

Steinhilber, P., Shipley, P. and Vvedenskaya, A. (2016) *University of Maryland Extension: Estimating corn silage and grain*.

Stoate, C., Báldi, A., Beja, P., Boatman, N. D., Herzon, I., van Doorn, A., de Snoo, G. R., Rakosy, L. and Ramwell, C. (2009) ‘Ecological impacts of early 21st century agricultural change in Europe - A review’, *Journal of Environmental Management*, 91(1), pp. 22–46. doi: 10.1016/j.jenvman.2009.07.005.

Stoate, C., Boatman, N. D., Borralho, R. J., Carvalho, C. R., De Snoo, G. R. and Eden, P. (2001) ‘Ecological impacts of arable intensification in Europe’, *Journal of Environmental Management*, 63(4), pp. 337–365. doi: 10.1006/jema.2001.0473.

Stochlic, R. and Sierra, L. (2007) ‘Conventional, mixed and “deregistered”. Organic Farmers: Entry Barriers and Reasons for Exiting Organic Production in California’, *California Institute for Rural Studies*.

Stopes, C., Lord, E. I., Philipps L.* and Woodward, L. (2002) ‘Nitrate leaching from organic farms and conventional farms following best practice’, *Soil Use and Management*, 18(3), pp. 256–263. doi: 10.1079/SUM2002128.

Stuart, M. E. and Lapworth, D. J. (2016) ‘Macronutrient status of UK groundwater: Nitrogen, phosphorus and organic carbon’, *Science of the Total Environment*, 572, pp. 1543–1560. doi: 10.1016/j.scitotenv.2016.02.181.

Sutter, L., Jeanneret, P., Bartual, A. M., Bocci, G. and Albrecht, M. (2017) ‘Enhancing plant diversity in agricultural landscapes promotes both rare bees and dominant crop-pollinating bees through complementary increase in key floral resources’, *Journal of Applied Ecology*, 54(6), pp. 1856–1864. doi: 10.1111/1365-2664.12907.

Sutton, M. A., Billen, G., Bleeker, A., Erisman, J. W., Grennfelt, P., Grinsven, H. Van, Grizzetti, B., Howard, C. M. and Leip, A. (2011) ‘European Nitrogen Assessment - Technical Summary’, (January).

Tilman, D., Cassman, K. G., Matson, P. A., Naylor, R. and Polasky, S. (2002) ‘Agricultural sustainability and intensive production practices’, *Nature*, 418(6898), pp. 671–677. doi: 10.1038/nature01014.

Tisdell, C. (1996) ‘Economic indicators to assess the sustainability of conservation farming projects: An evaluation’, *Agriculture, Ecosystems and Environment*, 57(2–3), pp. 117–131. doi: 10.1016/0167-8809(96)01017-1.

Tol, R. S. J. (2019) ‘A social cost of carbon for (almost) every country’, *Energy*

Economics. Elsevier B.V., 83, pp. 555–566. doi: 10.1016/j.eneco.2019.07.006.

Tóth, Z., Hornung, E. and Báldi, A. (2018) 'Effects of set-aside management on certain elements of soil biota and early stage organic matter decomposition in a High Nature Value Area , Hungary', *Nature Conservation*, 29, pp. 1–26. doi: 10.3897/natureconservation.29.24856.

Trenbirth, H. and Dutton, A. (2020) *Woodland natural capital accounts : ecosystem services for England , Scotland , Wales and Northern Ireland*. Available at: https://www.ons.gov.uk/economy/environmentalaccounts/bulletins/woodlandnaturalcapitalaccountsuk/ecosystemservicesforenglandscotlandwalesandnorthernireland2020.

Tschumi, M., Albrecht, M., Bärtschi, C., Collatz, J., Entling, M. H. and Jacot, K. (2016) 'Perennial, species-rich wildflower strips enhance pest control and crop yield', *Agriculture, Ecosystems and Environment*. Elsevier B.V., 220, pp. 97–103. doi: 10.1016/j.agee.2016.01.001.

Tuck, S. L., Winqvist, C., Mota, F., Ahnström, J., Turnbull, L. A. and Bengtsson, J. (2014) 'Land-use intensity and the effects of organic farming on biodiversity: A hierarchical meta-analysis', *Journal of Applied Ecology*, 51(3), pp. 746–755. doi: 10.1111/1365-2664.12219.

Tully, K. L. and Mcaskill, C. (2020) 'Promoting soil health in organically managed systems : a review', *Organic Agriculture*. Organic Agriculture, 10, pp. 339–358.

Tuomisto, H. L., Hodge, I. D., Riordan, P. and Macdonald, D. W. (2012) 'Does organic farming reduce environmental impacts ? e A meta-analysis of European research', *Journal of Environmental Management*. Elsevier Ltd, 112(834), pp. 309–320. doi: 10.1016/j.jenvman.2012.08.018.

Twiston-Davies, G., Becher, M. A. and Osborne, J. L. (2021) 'BEE-STEWARD: A research and decision-support software for effective land management to promote bumblebee populations', *Methods in Ecology and Evolution*, 12(10), pp. 1809–1815. doi: 10.1111/2041-210X.13673.

UK Water Projects (2015). 'Lilleshall Nitrate Improvement Scheme: Providing a water treatment plant at Redhill Reservoir'. Water Projects Online. Available at: https://waterprojectsonline.com/wp-content/uploads/case_studies/2015/Lilleshall-Nitrate-Improvement-Scheme-2015.pdf (Accessed: 30 December 2021).

Van-camp, L., Bujarrabal, B., Gentile, A. R., Jones, R. J. A., Montanarella, L., Olazabal, C. and Selvaradjou, S. (2004) 'Reports of the Technical Working Groups Established under the Thematic Strategy for Soil Protection. EUR 21319 EN/3.', *Office for Official Publications of the European Communities, Luxembourg*, p. 872.

Vandenbygaart, A. J., Bremer, E., McConkey, B. G., Ellert, B. H., Janzen, H. H., Angers, D. A., Carter, M. R., Drury, C. F., Lafond, G. P. and McKenzie, R. H. (2011) 'Impact of Sampling Depth on Differences in Soil Carbon Stocks in Long-Term Agroecosystem Experiments', *Soil Science Society of America Journal,* 75(1), pp. 226–234. doi: 10.2136/sssaj2010.0099.

Verschuuren, J. (2017) 'Towards a regulatory design for reducing emissions from agriculture: Lessons from Australia's carbon farming initiative', *Climate Law*, 7(1), pp. 1–51. doi: 10.1163/18786561-00701001.

Verzeaux, J., Alahmad, A., Habbib, H., Nivelle, E., Roger, D., Lacoux, J., Decocq, G., Hirel, B., Catterou, M., Spicher, F., Dubois, F., Duclercq, J. and Tetu, T. (2016) 'Cover crops prevent the deleterious effect of nitrogen fertilisation on bacterial diversity by maintaining the carbon content of ploughed soil', *Geoderma*. Elsevier B.V., 281, pp. 49–57. doi: 10.1016/j.geoderma.2016.06.035.

Van Vooren, L., Reubens, B., Broekx, S., De Frenne, P., Nelissen, V., Pardon, P. and Verheyen, K. (2017) 'Ecosystem service delivery of agri-environment measures: A synthesis for hedgerows and grass strips on arable land', *Agriculture, Ecosystems and Environment*. Elsevier, 244(November 2016), pp. 32–51. doi: 10.1016/j.agee.2017.04.015.

Walmsley, A. and Sklenička, P. (2017) 'Various effects of land tenure on soil biochemical parameters under organic and conventional farming − Implications for soil quality restoration', *Ecological Engineering*, 107, pp. 137–143. doi: 10.1016/j.ecoleng.2017.07.006.

Water World (2015). 'Ion exchange plant reduces nitrates to 5 mg per litre for Yorkshire Water'. Available at: https://www.waterworld.com/international/article/16202598/ion-exchange-plant-reduces-nitrates-to-5-mg-per-litre-for-yorkshire-water (Accessed: 30 December 2021).

Wang, L., Stuart, M. E., Bloomfield, J. P., Butcher, A. S., Gooddy, D. C., Mckenzie, A. A., Lewis, M. A. and Williams, A. T. (2011) 'Prediction of the arrival of peak nitrate concentrations at the water table at the regional scale in Great Britain', *Hyrrological Processes*, 26, pp. 226–239. doi: 10.1002/hyp.8164.

Wang, L., Stuart, M. E., Lewis, M. A., Ward, R. S., Skirvin, D., Naden, P. S., Collins, A. L. and Ascott, M. J. (2016) 'The changing trend in nitrate concentrations in major aquifers due to historical nitrate loading from agricultural land across England and Wales from 1925 to 2150', *Science of the Total Environment*. The Authors, 542, pp. 694–705. doi: 10.1016/j.scitotenv.2015.10.127.

Wang, L., Stuart, M. E., S, B. A., Goody, D. C. and Bloomfield, J. P. (2013) 'The nitrate time bomb : a numerical way to investigate nitrate storage and lag time in the unsaturated zone', *Environmental Geochemical Health*, 35, pp. 667–681. doi: 10.1007/s10653-013-9550-y.

Webb, J., Harrison, R. and Ellis, S. (2000) 'Nitrogen fluxes in three arable soils in the UK', *European Journal of Agronomy*, 13, pp. 207–223.

Webb, J., Loveland, P. J., Chambers, B. J., Mitchell, R. and Garwood, T. (2001) 'The impact of modern farming practices on soil fertility and quality in England and Wales', *The Journal of Agricultural Science*, 137, pp. 127–138. doi: 10.1017/S0021859601001290.

Welsch, J., Songling, C., Buckley, H. L., Lehto, N. J., Jones, E. E. and Case, B. S. (2019) 'How many samples? Soil variability affects confidence in the use of common agroecosystem soil indicators', *Ecological Indicators*, 102, pp. 401–409. doi: 10.1016/j.ecolind.2019.02.065.

Wheater, H. and Evans, E. (2009) ‘Land use, water management and future flood risk’, *Land Use Policy*, 26S, pp. S251–S264. doi: 10.1016/j.landusepol.2009.08.019.

Whitehead, D. (2020) ‘Management of Grazed Landscapes to Increase Soil Carbon Stocks in Temperate, Dryland Grasslands’, *Frontiers in Sustainable Food Systems*, 4, pp. 1–7. doi: 10.3389/fsufs.2020.585913.

Wilbois, K. and Schmidt, J. E. (2019) ‘Reframing the Debate Surrounding the Yield Gap between Organic and Conventional Farming’, *Agronomy*, 9(82), pp. 1–16. doi: 10.3390/agronomy9020082.

Williams, A. and Hedlund, K. (2013) ‘Indicators of soil ecosystem services in conventional and organic arable fields along a gradient of landscape heterogeneity in southern Sweden’, *Applied Soil Ecology*. Elsevier B.V., 65, pp. 1–7. doi: 10.1016/j.apsoil.2012.12.019.

Williams, D. M., Blanco-canqui, H., Francis, C. A. and Galusha, T. D. (2017) ‘Organic Farming and Soil Physical Properties: An Assessment after 40 Years’, *Agronomy Journal*, 109(2), pp. 600–609. doi: 10.2134/agronj2016.06.0372.

Withers, P. and Foy, B. (2006) ‘Phosphorus cycling in UK agriculture and implications for phosphorus loss from soil’, *Soil Use and Management*, (January). doi: 10.1079/SUM200181.

Withers, P. J. A., Neal, C., Jarvie, H. P. and Doody, D. G. (2014) ‘Agriculture and eutrophication: Where do we go from here?’, *Sustainability*, 6, pp. 5853–5875. doi: 10.3390/su6095853.

Wolton, R. (2018) ‘The Natural Capital of Hedges : Briefing note’, *Hedgelink UK*, (May 2018). Available at: https://devonhedges.org/wp-content/uploads/2018/11/Natural-Capital-of-Hedges.-briefing-8-May-2018-Rob-Wolton.pdf.

Wood, T. J., Holland, J. M. and Goulson, D. (2015) ‘A comparison of techniques for assessing farmland bumblebee populations’, *Oecologia*, 177(4), pp. 1093–1102. doi: 10.1007/s00442-015-3255-0.

Wood, Thomas J., Holland, J. M., Hughes, W. O. H. and Goulson, D. (2015) ‘Targeted agri-environment schemes significantly improve the population size of common farmland bumblebee species’, *Molecular Ecology*, 24(8), pp. 1668–1680. doi: 10.1111/mec.13144.

Yeluripati, J., Rivington, M., Lilly, A., Baggaley, N., Mckenzie, B., Aitkenhead, M. and Matthews, K. (2018) *Payment for carbon sequestration in soils : A scoping study Key points (Climate X Change)*. doi: 10.13140/RG.2.2.26883.55847.

Zhou, M. and Butterbach-bahl, K. (2014) ‘Assessment of nitrate leaching loss on a yield-scaled basis from maize and wheat cropping systems’, *Plant Soil*, 374, pp. 977–991. doi: 10.1007/s11104-013-1876-9.

Appendix A:

Appendix A.1: Data used to calculate specific conversion figure to calculate percentage SOC from SOM

Data used to collect local SOM to SOC conversion factor. The percentage of soil organic matter (SOM) as soil organic carbon (SOC) was first calculated based on the sub-set data using TC from elemental analysis. Calculated as:

$$\%\ of\ SOM\ as\ SOC = (TC\ \div\ SOM) * 100$$

The mean of percentage SOC was then calculated to determine the conversion factor. This was rounded to 0.52 for simplicity. A similar method was conducted by linear regression equation and this yielded much the same result.

Site	SOM% (LOI)	TC% (elemental analysis)	% of SOM as SOC
1	2.48	1.31	52.78
2	2.26	1.17	51.95
3	3.22	1.85	57.42
4	3.19	1.67	52.19
5	2.94	1.59	54.08
6	3.35	1.85	55.13
7	2.72	1.51	55.55
8	2.01	0.99	49.45
9	1.84	0.86	46.47
10	2.74	1.38	50.51
11	2.84	1.70	59.93
12	2.98	1.46	48.89
13	4.02	1.86	46.19
14	3.46	1.67	48.15
15	2.41	1.09	45.10
16	2.88	1.30	45.21
17	2.75	1.45	52.55
18	2.53	1.16	45.73
19	2.66	1.43	53.65
20	2.66	1.37	51.65
21	2.41	1.13	47.01
22	2.93	1.69	57.68
23	2.70	1.42	52.70
24	2.70	1.61	59.67
25	3.51	1.96	55.78
26	3.05	1.51	49.48
27	3.78	2.02	53.49
28	2.62	1.31	50.11
29	3.29	1.72	52.28

30	2.83	1.22	42.97
31	3.13	1.49	47.64
32	3.23	1.66	51.42
33	2.40	1.40	58.50
34	3.90	1.97	50.49
35	2.73	1.44	52.71
36	3.29	1.93	58.66
37	2.51	1.18	46.89
38	2.40	1.30	54.00
39	3.50	1.57	44.89
40	2.42	1.21	49.83
41	2.20	1.25	56.86
42	2.65	1.10	41.32
43	1.79	0.95	53.24
44	2.44	1.49	61.23
45	2.17	1.21	55.62
46	2.66	1.41	52.82
47	2.60	1.34	51.58
48	2.48	1.12	45.20
49	2.80	1.32	47.29
50	2.59	1.32	50.89
51	3.01	1.61	53.42
52	2.03	1.19	58.47
53	1.94	1.04	53.35
54	2.29	1.20	52.23
Mean % SOM as SOC			51.71
Standard deviation			4.60
Figure was rounded to 52% (i.e. a conversion factor of 0.52)			

Appendix A.2: Linear model outputs for predicting first cut silage yields on three fields (21 sites) on the organic Home Farm (Farm 3)

Second silage crop yields (secondDMyield) were used to predict first silage crop yields across three study fields.

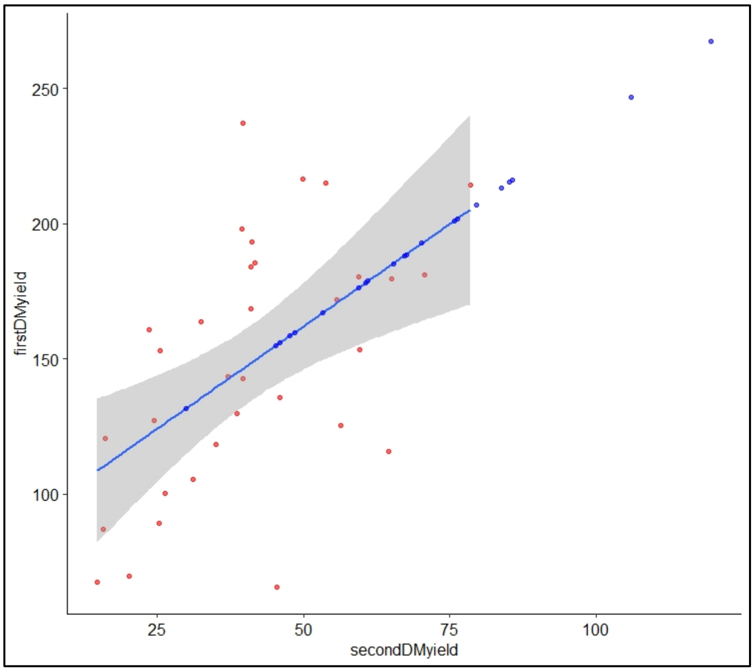
Model output:

```
Lm(firstDMyield ~ secondDMyield, data = data)
Residuals:
    Min      1Q  Median      3Q     Max
-89.558 -23.389   1.018  28.230  90.745

Coefficients:
              Estimate Std. Error t value Pr(>|t|)
(Intercept)    86.2750    18.6321   4.630 6.18e-05 ***
secondDMyield   1.5139     0.4218   3.589  0.00113 **
---
Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

Residual standard error: 39.44 on 31 degrees of freedom
  (21 observations deleted due to missingness)
Multiple R-squared:  0.2935,	Adjusted R-squared:  0.2708
F-statistic: 12.88 on 1 and 31 DF,  p-value: 0.001128
```

Plot of linear model with real data (red), predicted values (blue) and the standard error of the model:



Appendix B:

Appendix B.1: Farm management terms used to rank farm intensity

Selecting management terms to rank farm intensity and/or model impact on soil properties:

Indicators distinguishing different farm management practices were adapted from Buchi et al., (2019), using and/or tweaking those that were considered applicable in this context and where farm management data allowed. Buchi et al., (2019) investigate a suite of agricultural indicators that could be used to unveil the hidden side of cropping classification (e.g. the similarities and differences between conventional, no-till and organic systems) and in drawing conclusions about yield, environmental benefits or soil properties.

A simplified selection of the indicators were used, with *a-priori* justification, to rank farm intensity and in model selection to explore what management practices might be driving soil natural capital condition. A composite index “Farm Intensity” was created by ranking the intensity for each farm for each of the management practices.

Where possible the same/similar farm management data was collated for the farms in this study over a 6 year period from 2013 – 2018. The past six years was selected on the grounds of the length of the organic rotation for the Home Farm (Farm 3) and based on the reliability of some of the farm management data.

Mgmt. description	Definition	Scale	Farm data required	Ref. material required	Assumption required	Further details
Ann_orgN	Annual available nitrogen (N) applied as slurry, FYM or grazing animals from 2013 – 2018 Units: kg ha^{-1}	Field	Farm records Farm 1: 2016 – 2019 Farm 3: 2018 – 2019 Farm 2: Crop Std. Farm 4: Std. annual Farm 5: 2018 – 2019			For Farm 4 and Farm 5 Farm N inputs from grazing animal excreta were estimated using DM input data for dairy cows from (Jacobs *et al.*, 2020) and proportions of available N in FYM using data from AHDB Nutrient Mgmt guide (RB209). Slurry (pig or cattle) and FYM data was provided for the period specified for each farm. Assumptions were then made for the inputs for preceding years based on the cropping records. Where information on kg available N from slurry or FYM were provided (i.e. for Farm 1 and Farm 3 Farms) then these values were used. Otherwise RB209 (AHDB, 2020) values were used to calc. available N input from quantity of slurry or FYM input. Annual records for Farm 2 were based on inputs specific to maize crops following discussion with farmer.
Ann_inorgN	Annual available N applied as inorganic fertilizer from 2013 – 2018 Units: kg ha^{-1}	Field	Farm records Farm 1: 2016 – 2019 Farm 3: NA organic Farm 2: 2013 - 2018 Farm 4: Std. annual			Assumptions were made for the years preceding the fertilizer records provided. These were based on existing applications for each crop and applied to each year using longer term cropping records.

			Farm 5: NA organic			
Ann_totN	Total of the two above Units: kg ha^{-1}	Field	Farm records Farm 1: 2016 – 2019 Home: 2018 – 2019 Farm 2: 2013 - 2018 Farm 4: Std. annual Farm 5: 2018 – 2019			Combination of the two input above.
Ann_orgP	Annual available phosphate (P) applied as slurry, FYM or grazing animals from 2013 – 2018 Units: kg ha^{-1}		Farm records Farm 1: 2016 – 2019 Farm 3: 2018 – 2019 Farm 2: Crop Std. Farm 4: Std. annual Farm 5: 2018 – 2019			For Farm 4 and Farm 5 Farm P inputs from grazing animal excreta were estimated using DM input data for dairy cows from Jacobs et al., (2020) and proportions of available P in FYM using data from AHDB Nutrient Mgmt guide (RB209). Slurry (pig or cattle) and FYM data was provided for the period specified for each farm. Assumptions were then made for the inputs for preceding years based on the cropping records. RB209 (AHDB, 2020) values were used to calc. available P input from quantity of slurry or FYM input. Annual records for Farm 2 were based on inputs specific to maize crops following discussion with farmer.
Ann_inorgP	Annual available P applied as inorganic	Field	Farm records Farm 1: 2016 – 2019			Assumptions were made for the years preceding the fertilizer records provided. These were based on existing applications for each crop and applied to each year using the longer term cropping records.

	fertilizer from 2013 – 2018 Units: kg ha^{-1}		Farm 3: 2018 – 2019 Farm 2: 2013 - 2018 Farm 4: Std. annual Farm 5: 2018 – 2019			
Ann_totP	Total of the two above Units: kg ha^{-1}	Field	Farm records Farm 1: 2016 – 2019 Farm 3: 2018 – 2019 Farm 2: 2013 - 2018 Farm 4: Std. annual Farm 5: 2018 – 2019			Combination of the two P inputs above.
Ann_OM_ kgha	Annual organic matter (dry portion) applied as slurry, FYM or by grazing animals from 2013 – 2018 Units: kg ha^{-1}	Field	Farm records Farm 1: 2016 – 2019 Farm 3: 2018 – 2019 Farm 2: Crop std. Farm 4: Std. annual			Slurry and FYM input data was provided for the years specified. Assumptions were made for the years preceding organic matter input records. These were based on existing applications for each crop and applied to each year using the longer term cropping records. Annual records for Farm 2 were based on inputs specific to maize crops following discussion with farmer.

			Farm 5: 2018 – 2019			
Ann. stable OM Input (IC)	Annual proportion of stable organic matter (slurry and FYM) likely to be incorporated into the soil as soil organic matter (based on isohumic coefficients) Units: kg ha^{-1}	Field	Using organic matter input data (as above)			The isohumic coefficient is defined as the fraction of applied organic matter which is 'transformed' into soil organic matter (Maillard and Angers, 2014). It was calculated using dry matter data and coefficients for each input type from (Büchi *et al.*, 2019). See Appendix B.2.
No. spray apps 2019	The number of separate fungicide and/or herbicide applications conducted ahead of the harvest of the main crop in 2019	Field	Farm records: Farm 1: 2019 Farm 2: 2019 Farm 4: Std. annual NA on organic fields			Note: This does not account for the total number of different products. Multiple products were often in included in one spray operation.
No. crops	Number of cultivated crops between 2013	Field	Cropping data for all farms was provided			No assumptions required

	and 2018 (incl. cover crops)		from 2013 – 2018			
No. different crops	Number of different cultivated crops between 2013 – 2018 (incl. cover crops)	Field	Cropping data for all farms was provided from 2013 – 2018			No assumptions required
Crop diversity	Number of different crops divided by the total number of crops between 2013 – 2018	Field	Cropping data for all farms was provided from 2013 – 2018			No assumptions required
Yrs. grass	Number of whole years where field was in grass (excl. over wintered grass leys)	Field	Cropping data for all farms was provided from 2013 – 2018			No assumptions required
No. yrs cover crop	Number of times field has been in a winter cover crop (excl. grazed stubble turnips)	Field	Cropping data for all farms was provided from 2013 – 2018			No assumptions required Some gaps in ow cropping for Farm 3 Farm – filled in with assumptions based on rotation stage.
No. grazing years	Number of years field cattle grazed from 2013 – 2018 (winter forage grazing	Field	Cropping data for all farms was provided from 2013 – 2018			Winter forage grazing was assumed to account for 0.25 of a year, with period running November, December, January.

	was considered 0.25 years)					
Crop sampled	The crop at the time of soil sampling in 2018/19	Field	Field observation 2018/19			
No. primary tillage ops	Number of primary tillage operations conducted between 2013 – 2018 (includes mouldboard and chisel ploughing, HEVA, top down and de-stoning cultivations)	Field	Farm records Farm 1: 2016 – 2019 Farm 3: 2018 – 2019 Farm 2: Crop std. Farm 4: Std. annual Farm 5: 2018 – 2019			For the years preceding detailed farm records on cultivations then primary tillage had to be estimated on the basis of field cropping history. The type of tillage was assumed based on current practice and discussions with farmer. Secondary tillage operations were omitted as records were not kept for all farms.
No. times mb plough	Number of times mouldboard ploughing has been used as the primary tillage method between 2013-2018	Field	Farm 1: 2016 – 2019 Farm 3: 2018 – 2019 Farm 2: Crop std. Farm 4: Std. annual Farm 5: 2018 – 2019			For the years preceding detailed farm records on cultivations then the use of the mouldboard plough had to be assumed on the basis of field cropping history. The assumption was made on the cropping history, current cultivation practices, discussion with the farmer and/or standard crop cultivation practice using Nix (2018).

Years since tillage	Time between soil sample being collected and the time the field was last tilled	Field	Detailed farm records with date of last tillage were available for Farm 1, Farm 3 and Farm 2 Farm.			Assumptions had to be made about the exact date of when pasture fields at Farm 4 and Farm 5 Farms were last tilled as these pre-dated 2013 in some instances. Note: Originally calculated in days since tillage and re-scaled to years to incorporate into model runs.
Est_passes_2018	The estimated number of field passes conducted in 2018 ahead of the soil sample being collected	Field	Detailed farm records for Farm 3, Farm 2, Farm 5 and Farm 1 Farm.			Farm 4 passes were assumed on the basis of standard annual pasture management provided by farmer. Passes included were applications or organic and inorganic fertilizer, harvest operations, drilling passes, spray applications and additional miscellaneous passes.
Yield 2019	Biomass yield data for all arable or silage scenarios on Farm 1, Farm 3 and Farm 2 Farm (see Methods) in t ha^{-1}. Number of times field grazed for pasture.	Variable	Field data Farm grazing records: Farm 5 Farm: 2019 Farm 4 Farm: 2019			Crop samples over a known area were collected in 2019 ahead of harvest to calculate DM t ha^{-1} crop biomass. Number of times the whole field was grazed in 2019 was determined from grazing records provided by the farmer for Farm 5 and Farm 4 Field.

CoP 2019	Cost of Production of the 2019 crop Units: £ ha^{-1}	Field	Farm operation records for 2019 crop			The same contractor costs were used for all farms based on local contractor pricing for standard operations. This included all cultivations, seed bed prep, fertiliser and spray application and harvest costs. Where local contractor details did not cover a particular operation then costs were taken from the Nix Farm Pocketbook (Nix, 2018). Spray and inorganic fertiliser costs were taken from Nix (2019) based on the active ingredients shown in farm records. The Farm 3 Farm provided most organic seed costs. All other seed costs were taken from Nix (2018). Grass seed costs were annualised and included in calculations for leys and pasture. Dairy herd management operations were not included in grazing pasture CoP.

Appendix B.2: Isohumic coefficients used to calculate soil organic matter input and data used to estimate grazing animal inputs

FYM, Slurry and Dirty Water input data used for calculating nutrient and organic matter inputs

Material	Nitrogen (N) Kg N t-1	Source of N data	Phosphate (P) kg P2O5 t-1	Source P data	Dry Matter	Kg OM/t	Dry Matter Source	(IC) isohumic coefficient	DM content used for IC calc. (kg t or kg m3)	IC source
Cattle Slurry	1.3	CDE gatekeeper	0.3	RB209 2020 (low band)	2%	20	RB209 2020 (low band)	0.1	50	(Büchi *et al.*, 2019)
Cattle FYM	0.6	CDE gatekeeper	1.9	RB209 2020 (standard)	25%	250	RB209 2020 (standard)	0.5	150	(Büchi *et al.*, 2019)
Horse FYM	0.5	RB209 2020	3	RB209 2020	25%	250	RB209 2020 (standard)	0.4	270	(Büchi *et al.*, 2019)
Pig Slurry	Variable (1.31 - 1.9)	Farm 1 Gatekeeper	Variable (0.45 - 0.98)	Farm 1 Gatekeeper	2%	20	RB209 2020 (low band)	0.1	33	(Büchi *et al.*, 2019)
Pig FYM	1.1	Farm 1 Gatekeeper	3.6	Farm 1 Gatekeeper	25%	25	RB209 2020 (standard)	0.35	40	(Büchi *et al.*, 2019)
Dirty Water	0.15	RB209 2020	0.05	RB209 2020	0.50%	5	RB209 2020 (standard)	0.1	5	(Büchi *et al.*, 2019)

Note: Low Band estimates from the AHDB RB209 Fertilizer manual (AHDB 2020) were used for both cattle and slurry data for estimating phosphate input and dry matter input of organic material. The low band estimates was applied rather than the standard on the basis of discussion with both of the farmers that applied these inputs and on any past slurry analysis. Both farmers suggested slurry was particularly ‘wet’, with a low dry matter content.

Calculations of for organic matter and nutrient inputs from grazing animals

Livestock animal	DM inputs from excreta				Nutrient inputs from excreta			
	Mg DM animal $^{-1}$ yr^{-1}	kg DM excreta animal^{-1} hr^{-1}	Carbon content of excreta DM	Source excreta input and C content	Nitrogen (N) Kg N t^{-1}	Source of N data	Phosphate (P) kg P_2O_5 t^{-1}	Source P data
Dairy cow	1.425	0.16	0.38	Jacobs *et al.*, (2020)	0.9	RB209 (for fresh FYM) 2020	1.9	RB209 (for fresh FYM) 2020

Appendix B.3: Field management terms used in models to determine drivers behind soil natural capital condition

The same process was followed for selecting the model terms for each model selection run:

1. Check distribution of the data. If non-normality detected, log transform data and identify if there is an improvement
2. Plot all management terms alongside predicted variable in a scatterplot matrix (using Pairs Panels in R) to identify potential significant relationships and multi-collinearity issues
3. Create a global linear mixed effects model of all terms that were considered could have an impact on the predicted variable based on literature and field experience. All terms were additive. Random terms were used to account for variance within field and variance within field replicates, terms 'Field' and 'Field Group'
4. Run global model and identify any significant terms – re-scale variables if necessary
5. Check for multicollinearity issues using the Performance package in R and singularity issues (check for risk of overfitting)
6. Remove terms causing high multicollinearity (VIF > 10) in accordance with what terms were more significant or were considered to have a greater impact on predictor variable (based on literature and field experience)
7. Finalise the linear mixed effects model terms
8. Input selected terms and random effects into line code for the GLmulti model selection package in R. All terms were additive not interactive.
9. Run model selection process

Model terms used to explain soil carbon storage and n-potential:

Note singularity issue flagged in the refined model, as such Num_grazing_yrs and Num_mb_plough had to be removed – this seems to allow the model to run without issue. I.e. Variance for the random effects are shown (and are not 0 or basically 0).

However, it is important to note that for the Glmulti run it throws up lots of singular fit warnings. They are not preventing the model run but it remains an issue unless Num_tillage is removed. It is odd that this is an issue in the multi model run however and not in the actual simple model run….

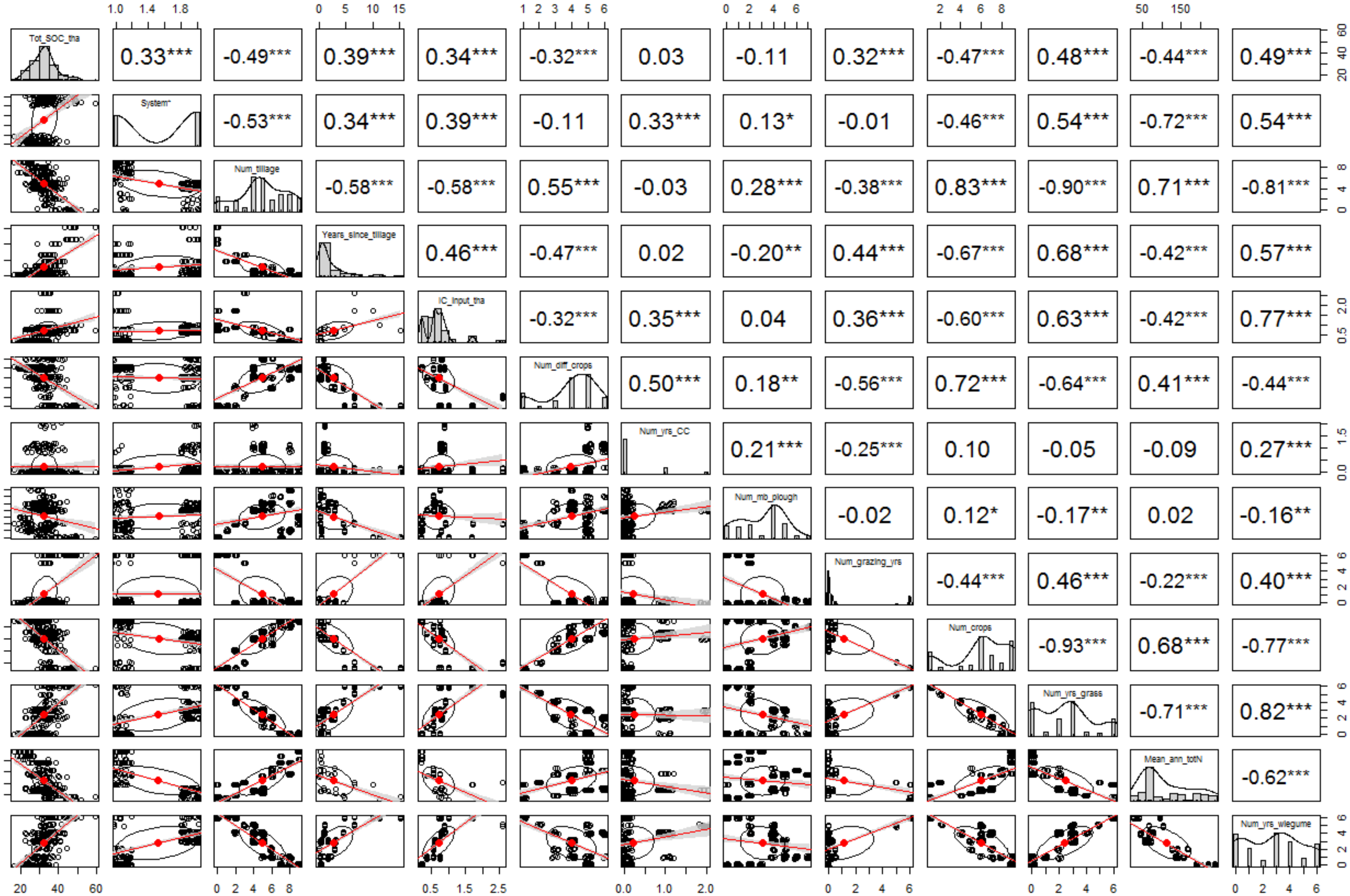
As the model runs properly for the complete terms and for the final 'best' model then Num_tillage has been retained in the model terminology.

Management code used in model	Defined as:	A priori justification	Citations	Potential multicollinearity issues
System	Organic or conventional	It is recognised that within different systems there might be more similar mgmt. and as such similar outcomes. To control for this effect, system is included as fixed effect. It is not considered viable to include random effects with less than 5 groups (Harrison, 2015).	(Loaiza-Puerta *et al.*, 2018)	NA
Num_tillage	Number of primary tillage operations conducted between 2013 – 2018 (includes mouldboard and chisel ploughing, HEVA, top down and de-stoning cultivations).	Frequency and intensity of soil disturbance has been associated with soil carbon losses.	(Busari *et al.*, 2015; R. Lal, 2015; Büchi *et al.*, 2017)	Num_yrs_grass (high -) Mean_ann_tot_N (high +) Num_diff_crops (mod) Num_crops (high +)

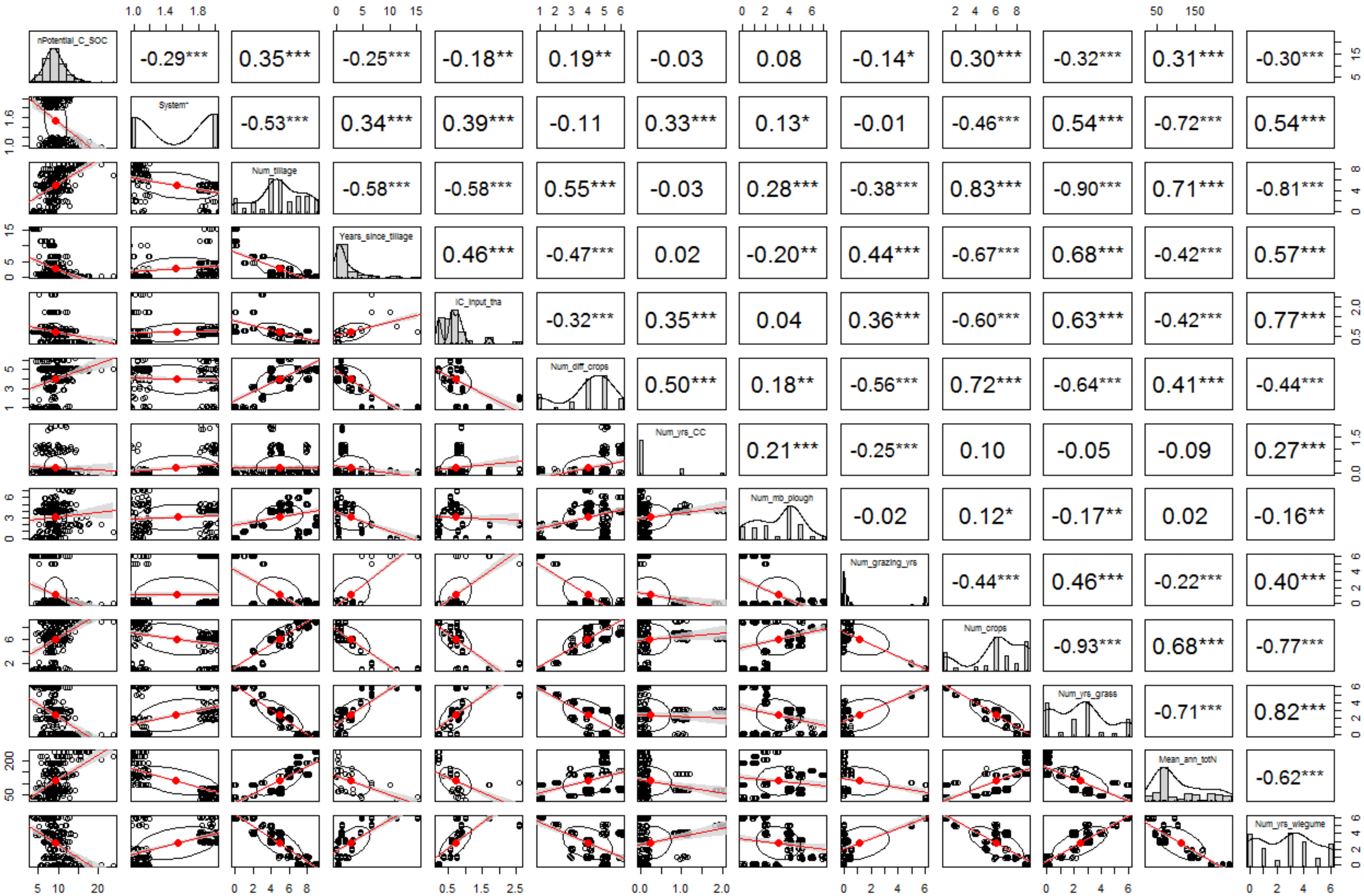
Years_since_tillage	Time between soil sample being collected and the time the field was last tilled (originally calculated in days and re-scaled to years for model run)	It is widely recognised that tillage can lead to the losses of soil carbon. The influence of time since last tillage was therefore considered to have an impact on soil carbon storage.	(Post and Kwon, 2000; Carolan and Fornara, 2016)	Num_tillage Num_grazing_yrs Num_yrs_grass Num_diff_crops Num_crops
IC_input_tha	Average annual proportion of stable organic matter (slurry and FYM) likely to be incorporated into the soil as soil organic matter (based on isohumic coefficients) between 2013 - 2018 Units: t ha^{-1}	The level of soil organic matter input would be expected to impact on the amount of soil carbon storage. Higher expected stable organic matter inputs would be expected to result in higher carbon storage levels.	(Powlson *et al.*, 2012)	Num_yrs_CC (high) Num_diff_crops (low)
Num_diff_crops	Number of different cultivated crops between 2013 – 2018 (incl. cover crops)	Considered an indicator of 'biodiversity' in other studies. Could alter C dynamics and soil biology.	(McDaniel et al., 2014)	Num_yrs_grass Num_grazing_yrs Num_tillage
Num_yrs_CC	Number of times field has been in a winter cover crop (excl. grazed stubble turnips).	CC recognised for retention of top soil and potential to build SOC.	(Verzeaux *et al.*, 2016)	Mean_ann_IC_input Num_diff_crops (mod)
Omitted parameters based on high multi collinearity issues:				
Num_mb_plough	Number of times mouldboard ploughing has been used as the primary tillage method between 2013-2018	Inversion tillage has been associated with higher losses of soil carbon than lower disturbance tillage methods. Removed on account of singularity issues (assuming overfitting) – when removed from the carbon storage model it seemed to rectify the issue	(Busari *et al.*, 2015; R. Lal, 2015; Büchi *et al.*, 2017)	None
Num_grazing_yrs	Number of years field cattle grazed from 2013 – 2018 (winter forage grazing was considered 0.25 years)	Cattle grazing can contributes soil organic matter into the soil through excreta and through trampling in senescent material.	(Leach *et al.*, 2014; Machmuller *et al.*, 2015;	Num_tillage Num_yrs_grass Num_diff_crops Num_crops

		Removed on account of singularity issues (assuming overfitting) – when removed from the carbon storage model it seemed to rectify the issue	Jacobs *et al.*, 2020)	
Num_crops	Number of cultivated crops between 2013 and 2018 (incl. cover crops)	The number of crops during a period provides information on the turnover of cropping during that time. More frequent cropping increases the potential for frequent soil disturbance and the removal of biomass from the field.	(McDaniel et al., 2014)	Num_tillage
Num_yrs_grass	Number of whole years where field was in grass (excl. over wintered grass leys)	Including grass leys and putting fields to grass has been considered a way of building SOC. Separating from grazing is hard though!	(Loaiza-Puerta *et al.*, 2018)	Num_tillage (high) Num_diff_crops (low)
Mean_ann_totN	Average annual total nitrogen input (the combination of average organic and inorganic N inputs) Units: kg ha^{-1}	Nitrogen plays an important role in the below ground carbon cycle and is important in the development of soil organic matter.	(Van Groenigen *et al.*, 2017)	Num_tillage
Num_yrs_wlegume	Number of years or part years (if cover crop) that field had a legume as part of the crop (clover in the ley, pasture or included in cover crop)	Legumes have been shown to increase the storage of soil carbon. High multicollinearity are unsurprising particularly with the number of years in grass as this accounts for most of the legume crop application on the estate.	(Stagnari *et al.*, 2017)	Num_tillage Num_crops IC_input Num_yrs_grass Mean_ann_totN High VIF > 10

Overview scatter plot matrix for mgmt terms for carbon storage with Kendall Rank correlations



Overview scatter plot matrix for mgmt terms for n-potential with Kendall Rank correlations



Model terms used to explain bulk density:

Final ‘good’ model (incl. removed terms) still showed Num_yrs_grass with a VIF of 10.09. I.e. unacceptable

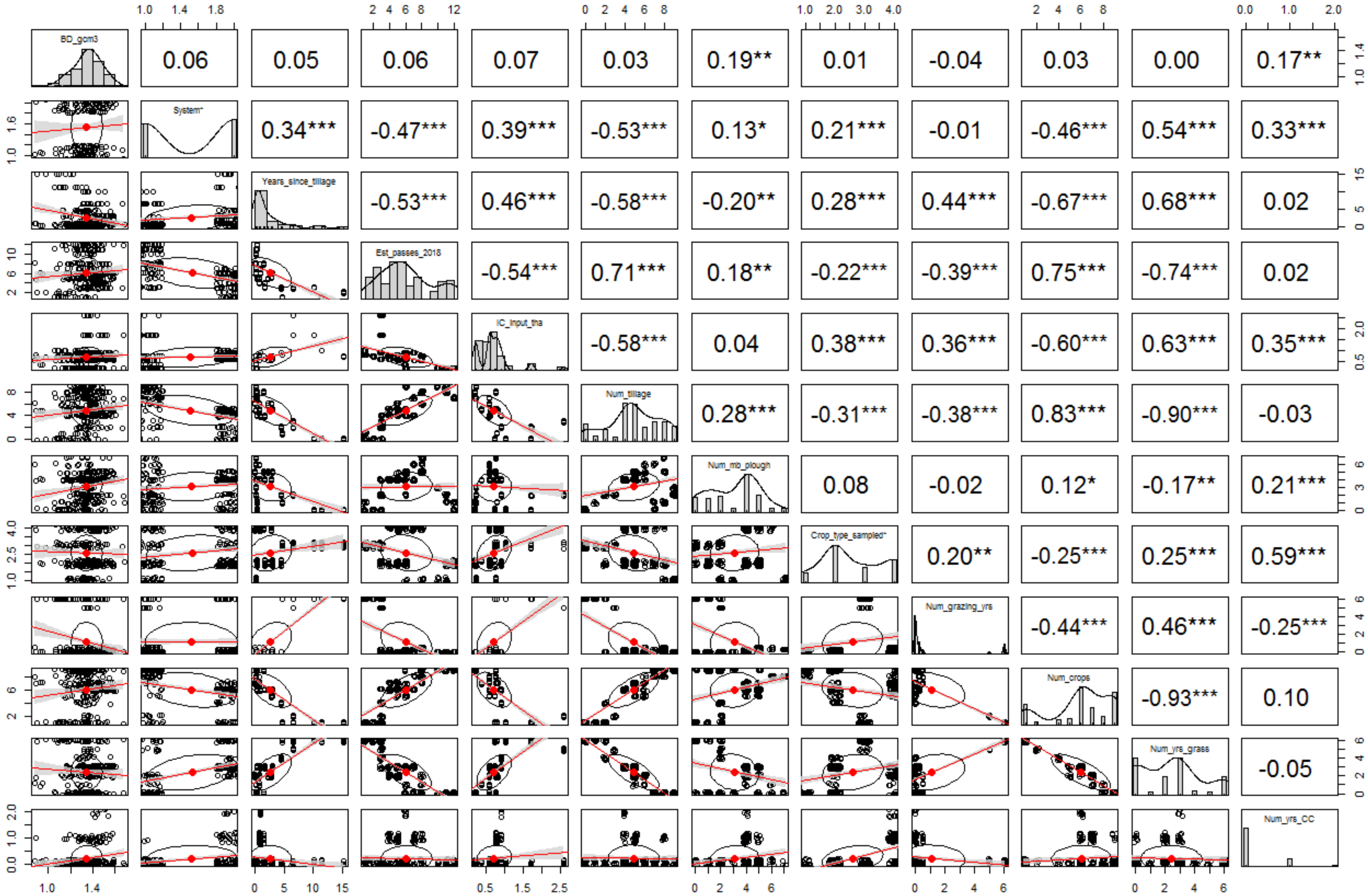
With Num_yrs_grass removed this reduced the VIF and allowed for an acceptable model – albeit with no significant terms

Management code used in model	Defined as:	A priori justification	Citations	Potential multicollinearity issues
System	Organic or conventional	It is recognised that within different systems there might be more similar mgmt. and as such similar outcomes. To control for this effect, system is included as fixed effect. It is not considered viable to include random effects with less than 5 groups (Harrison, 2015).		
Years_since_tillage	Time between soil sample being collected and the time the field was last tilled (originally calculated in days and re-scaled to years for model run)	Particularly on the sandy soil of the estate the soil can take time to ‘settle’ in the period after tillage. For example topsoil density would expected to be lower in the days after tillage, prior to processes of soil settling and consolidation. In the longer term (years), however, time since cultivation shows a decrease bulk density (e.g. under long term fallow or pasture).	(Daigh and Dejong-hughes, 2017)	Num_tillage Num_grazing_yrs Num_yrs_grass Num_diff_crops Num_crops
Est_passes_2018	The estimated number of field passes conducted in 2018 ahead of the soil sample being collected	Frequent field passes and trafficking with heavy machinery for cultivations, fertilizer application and harvest would be expected to increase soil compaction and bulk density.	(Gregory *et al.*, 2015)	

IC_input_tha	Average annual proportion of stable organic matter (slurry and FYM) likely to be incorporated into the soil as soil organic matter (based on isohumic coefficients) between 2013 - 2018 Units: t ha^{-1}	Increased soil organic matter has been shown to improve soil structure and reduce compaction.	(Pagliai et al., 2004)	Num_yrs_CC (high) Num_diff_crops (low)
Num_tillage	Number of primary tillage operations conducted between 2013 – 2018 (includes mouldboard and chisel ploughing, HEVA, top down and de-stoning cultivations).	Frequency of tillage and how heavily the soil is worked can have a significant impact on soil structure.	(Pagliai et al., 2004; Büchi *et al.*, 2017)	Num_yrs_grass Years_since_tillage Num_grazing_yrs Num_crops Num_diff_crops
Num_mb_plough	Number of times mouldboard ploughing has been used as the primary tillage method between 2013-2018	Frequency of tillage and how heavily the soil is worked can have a significant impact on soil structure. Mouldboard ploughing in particular has been shown to increase compaction, reducing soil porosity.	(Pagliai et al., 2004; Büchi *et al.*, 2017)	None
Crop_type_sampled	The type of crop that was in the field during the collection of soil samples. Crops were grouped into four; grass (ley), pasture, winter forage (wforage; incl. stubble Mean_ann_totNann_totNturnips and grazed cover crops direct drilled to previous crop) and arable (winter cereal crops).	It was identified that the other mgmt. terms might not capture different management associated with specific crop types. Crop type might have a short term impact on bulk density. For example crops experiencing differing levels of traffic or grazing disturbance. To control for potential influence of crop type it was included as a fixed effect. It is not considered viable to include random effects with less than 5 groups (Harrison, 2015).		None

Omitted parameters based on high multi collinearity issues:				
Num_yrs_grass	Number of whole years where field was in grass (excl. over wintered grass leys)	The length of time in grass has an impact on the frequency of soil disturbance in a field. In addition longer term grass leys have been shown to have lower BD than cropped fields (source). Num_yrs_grass was removed due to high VIF issues (particularly with number tillage).	(Loaiza-Puerta *et al.*, 2018)	Num_tillage (high) Num_diff_crops (low) Num_crops
Num_crops	Number of cultivated crops between 2013 and 2018 (incl. cover crops)	The number of crops during a period provides information on the turnover of cropping during that time. More frequent cropping increases the potential for frequent soil disturbance. Note: Term significant in model but due to multicollinearity issues tillage retained.	(Ball *et al.*, 2005)	Num_tillage Num_yrs_grass
Num_grazing_yrs	Number of years field cattle grazed from 2013 – 2018 (winter forage grazing was considered 0.25 years)	It is recognised that grazing can cause compaction issues and alter soil structure.	(Bilotta et al., 2007)	Num_tillage Years_since_tillage Num_yrs_grass Num_diff_crops Num_crops
Num_yrs_CC	Number of times field has been in a winter cover crop (excl. grazed stubble turnips).	Winter crops can have an impact on soil structure, protecting against soil erosion and potentially alleviating compaction. Inclusion in the model resulted in the term having a VIF > 10 and as such was removed.	(Chen et al., 2014)	Mean_ann_IC_input Num_diff_crops (mod)

Overview scatter plot matrix for mgmt terms for bulk density with Kendall Rank correlations



Model terms used to explain Olsen-P:

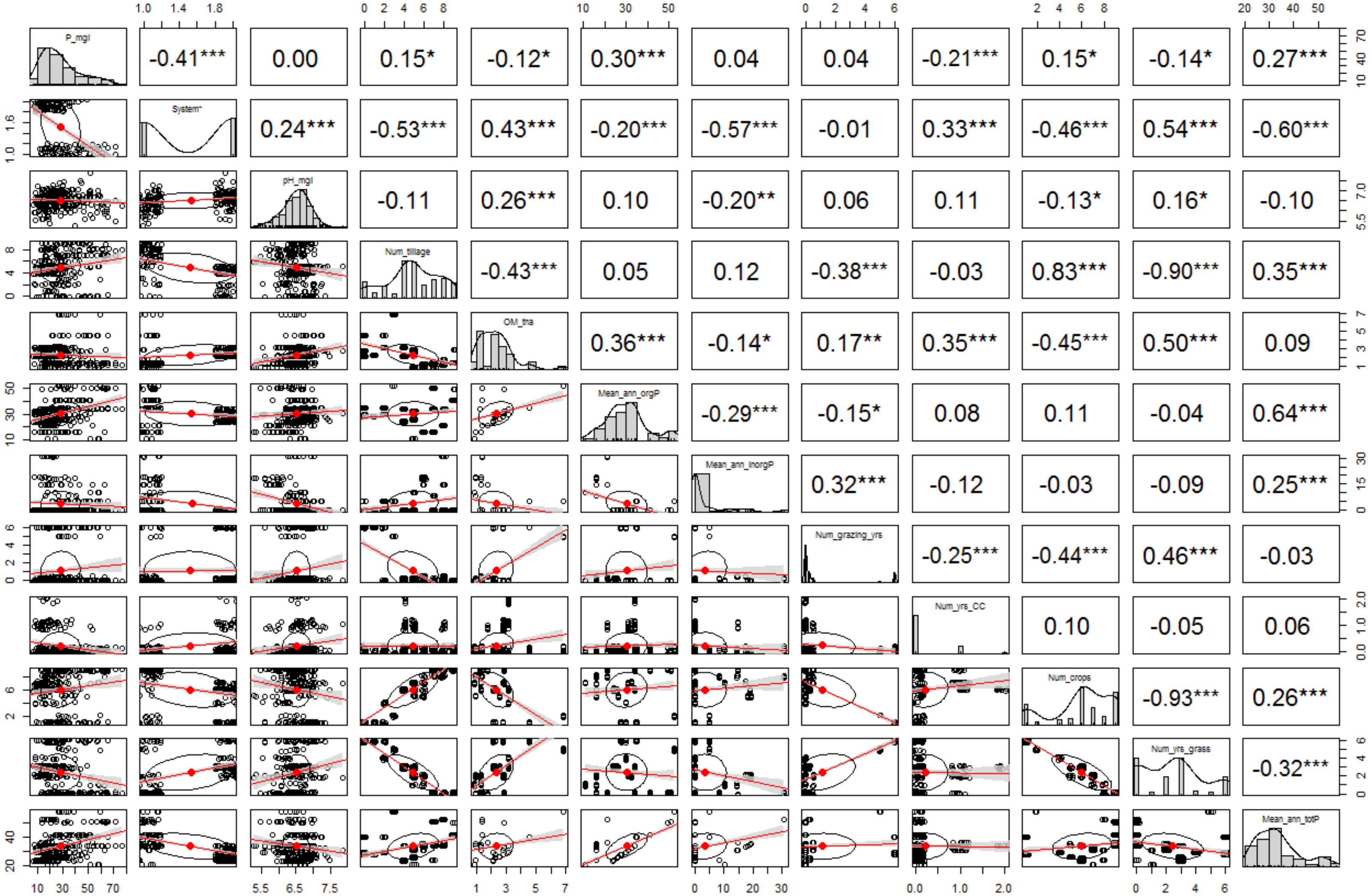
If Num_yrs_grass included then this has a VIF of >10 and as such was excluded.

Management code used in model	Defined as:	A-priori justification	Citations	Potential multicollinearity issues
System	Organic or conventional	It is recognised that within different systems there might be more similar mgmt. and as such similar outcomes. To control for this effect, system is included as fixed effect. It is not considered viable to include random effects with less than 5 groups (Harrison, 2015).	(Ohm *et al.*, 2017)	NA
pH	Soil pH (from soil sample data) was used as a proxy for amendments (liming) to address reduced pH.	Farm mgmt. records were not available for liming operations and as such pH was used. It is considered that those fields with a higher pH have been targeted in the past. It is widely recognised that pH can reduce the bioavailability of P and other nutrients in solution.	(Dungait *et al.*, 2012)	None
Num_tillage	Number of primary tillage operations conducted between 2013 – 2018 (includes mouldboard and chisel ploughing, HEVA, top down and de-stoning cultivations).	Frequency of soil disturbance has the potential to increase vulnerability of the soil to run-off and erosion, which is recognised as the main pathway for soil P losses from agriculture (Dungait *et al.*, 2012).	(Sharpley, 2016)	Num_yrs_grass Years_since_tillage Num_grazing_yrs Num_crops Num_diff_crops
OM_tha	Annual organic matter (dry portion) applied as slurry, FYM or by grazing animals from 2013 – 2018	Organic matter inputs from slurry, FYM and excreta provide a valuable supply of bioavailable P as it is broken down and	(Smith et al., 2001; AHDB, 2020)	Num_yrs_CC (high) Num_diff_crops (low)

	Units: t ha^{-1}	incorporated into the soil. Poorly timed applications can lead to significant P losses.		
Mean_ann_orgP	Annual available phosphate (P) applied as slurry, FYM or grazing animals from 2013 – 2018 Units: kg ha^{-1}	The scale of P both from fertilizer or organic matter inputs would be expected to have an impact on soil P.		None
Mean_ann_inorgP	Annual available P applied as inorganic fertilizer from 2013 – 2018 Units: kg ha^{-1}	The scale of P both from fertilizer or organic matter inputs would be expected to have an impact on soil P.		None
Num_grazing_yrs	Number of years field cattle grazed from 2013 – 2018 (winter forage grazing was considered 0.25 years)	It is recognised that grazing animal excreta can contain significant amounts of bioavailable P. One farmer explained that they consider themselves fairly self-sufficient on P based on cattle dunging during grazing.	(Withers and Foy, 2006)	Num_tillage Years_since_tillage Num_yrs_grass Num_diff_crops Num_crops
Num_yrs_CC	Number of times field has been in a winter cover crop (excl. grazed stubble turnips).	It is recognised that main losses of soil P occur during soil surface run-off and erosion. Cover cropping is considered a way to reduce soil and P losses.	(Sharpley, 2016)	None
Omitted parameters based on high multi collinearity issues:				
Num_yrs_grass	Number of whole years where field was in grass (excl. over wintered grass leys)	Losses of soil P usually occur through run-off as opposed to leaching through the soil profile. It is recognised that grass crops can slow flow Num_yrs_grass was removed due to high VIF issues (particularly with number tillage).	(Haygarth et al., 2002)	Num_tillage (high) Num_diff_crops (low) Num_crops
Num_crops	Number of cultivated crops between 2013 and 2018 (incl. cover crops)	The number of crops during a period provides information on the turnover of cropping during		Num_tillage Num_yrs_grass

		that time. Frequent crop turnover influences the risk of soil and P run-off. Note: Term causing high VIF and therefore removed.		
Mean_ann_totP	Total of organic and inorganic inputs	Removed due to high collinearity issues with organic P inputs		Mean_ann_orgP

Overview scatter plot matrix for mgmt terms for bioavailable P with Kendall Rank correlations

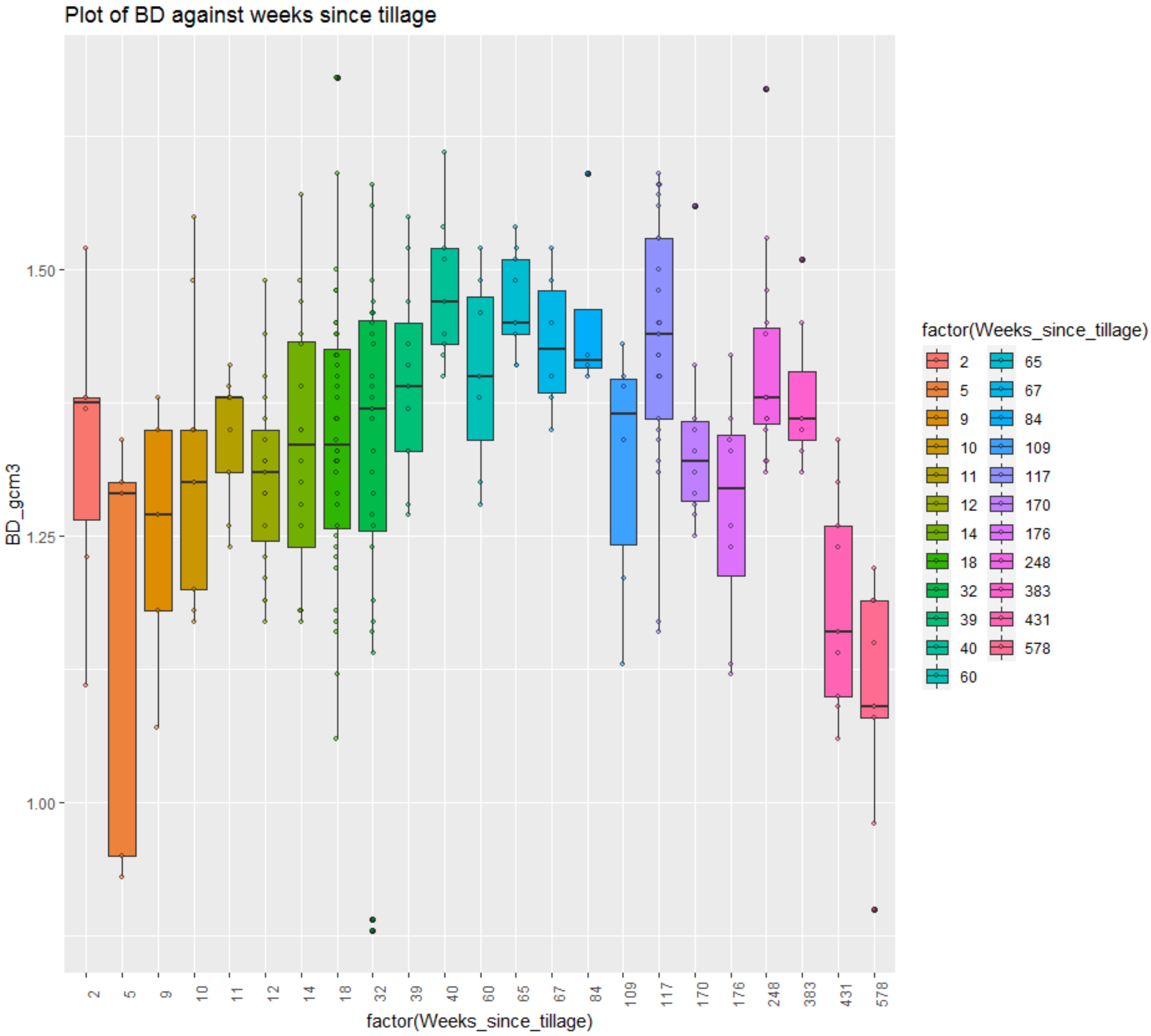


Appendix B.4: Table showing pairwise comparison of soil properties across farm intensities – outputs from linear mixed effects models with post-hoc analysis

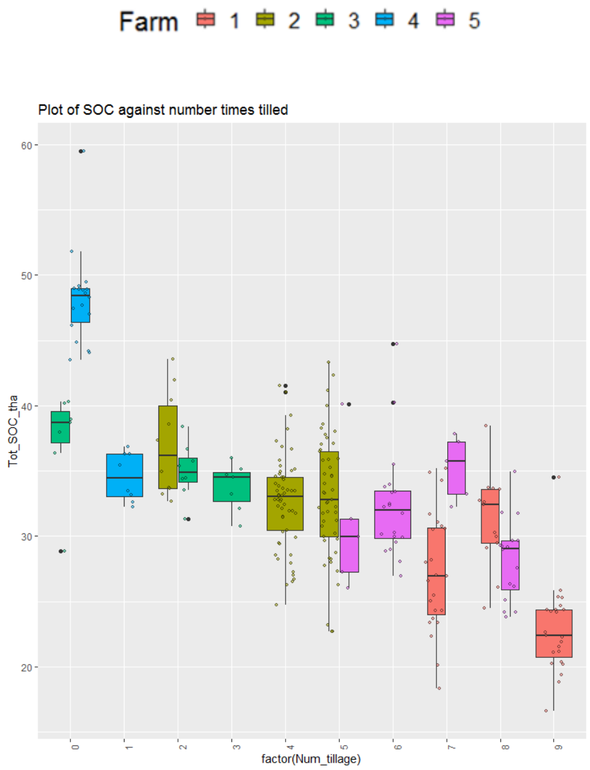
*Table: Showing the p-values from the pairwise comparison of soil properties against intensity from post-hoc analysis (tukey method) of linear mixed and generalised linear mixed effects models. Significant differences are shown with * p = <0.05, **p = <0.01 or *** p = <0.001. Trends that show potential significance are shown in bold.*

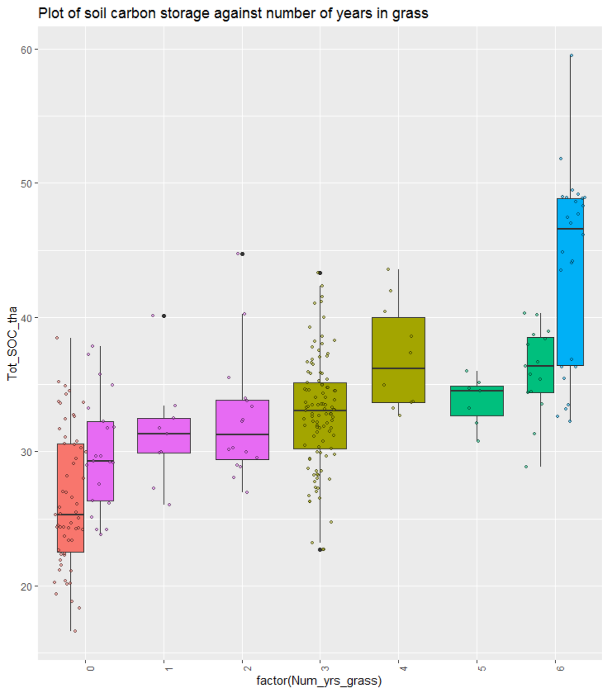
Intensity Pair	BD	Olsen-P	Carbon stocks	nPotential	Biomass yield
1 - 2	0.973	0.618	0.146	0.293	0.19
1 - 3	0.814	**0.04***	**0.016***	**0.083**	**0.01****
1 - 4	0.908	1.000	**0.046***	0.334	-
1 - 5	0.307	0.854	**0.001*****	**0.050***	-
2 - 3	0.373	0.414	0.543	0.921	0.23
2 - 4	0.673	0.798	0.511	0.990	-
2 - 5	0.461	1.000	**0.008****	0.337	-
3 - 4	1.000	0.199	0.949	1.000	-
3 - 5	**0.079**	0.773	**0.018***	0.519	-
4 - 5	0.179	0.911	0.133	0.697	-

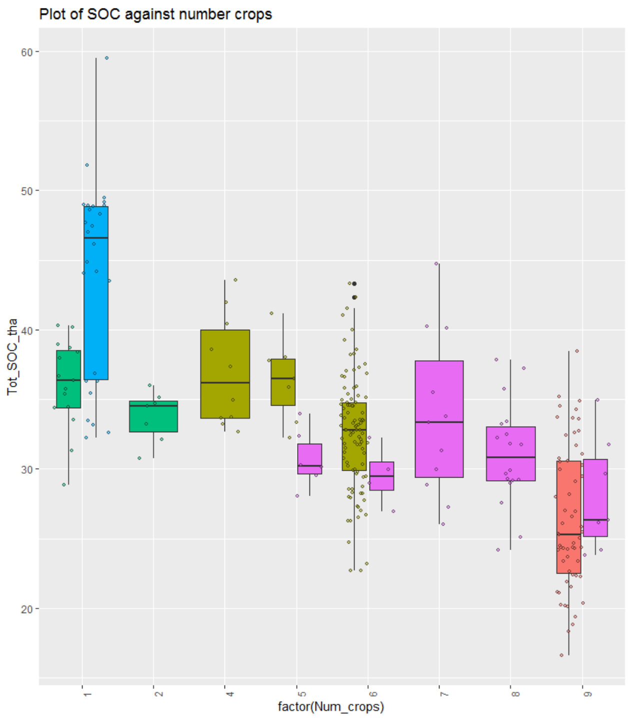
Appendix B.5: Showing the relationship between time since tillage and bulk density

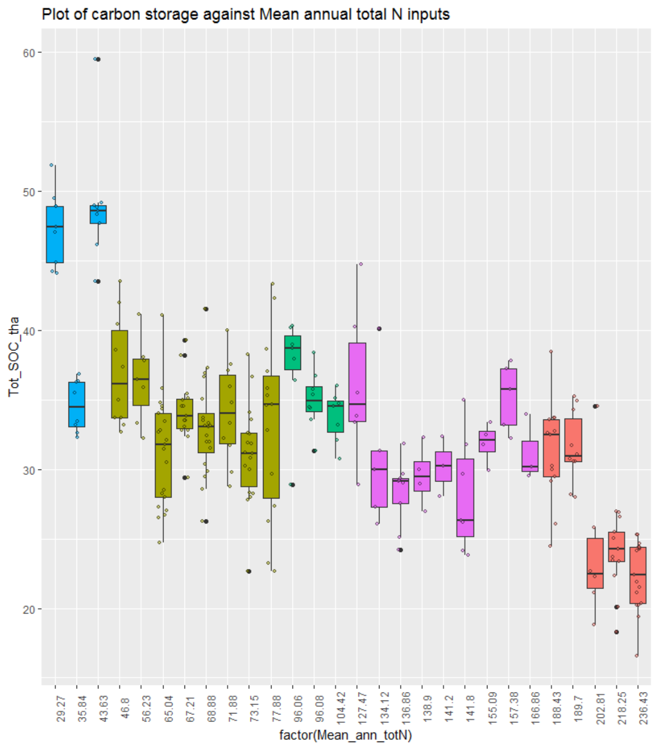


Appendix B.6: Showing the relationship between soil carbon storage and the different mgmt. variables exhibiting high multicollinearity






Plot of SOC against number crops
Tot_SOC_tha
60
50
40
30
20
1
2
4
5
6
7
8
9
factor(Num_crops)


Plot of carbon storage against Mean annual total N inputs
Tot_SOC_tha
60
50
40
30
20
29.27
35.84
43.63
46.8
56.23
65.04
67.21
68.88
71.88
73.15
77.88
96.06
96.08
104.42
127.47
134.12
136.86
138.9
141.2
141.8
155.09
157.38
166.86
188.43
189.7
202.81
218.25
236.43
factor(Mean_ann_totN)

Appendix B.7: Model outputs of detailed linear regression of soil conditions against yield

The model outputs analysing the relationship between soil properties and biomass yield for each sub set of fields are shown the table below. It is evident that only decreasing pH had a significant impact on the biomass yield of maize. This is counter to convention and further data exploration shows this could be skewed by an outlier. One scenario where there looks to be a positive trend is between soil carbon stocks and conventional grass yield ($p = 0.07$), which is highlighted and underlined in the table. However, this was not seen across the other crops and no other soil properties had a significant effect on biomass yield.

Table: Showing linear and where [G] generalized linear model outputs (estimates and below standard error) of the relationship between soil properties and main crop yield for organic triticale (Org trit), conventional winter wheat (Con wwheat), organic silage, conventional haylage and conventional maize. All fields used in the model were under the same or very similar management.

	Org trit	Con wwheat	Org silage	Con haylage[G]	Con maize[G]
(Intercept)	-3.44	12.49	46.61	-32.92	50.62 *
	[-27.77, 20.88]	[-78.97, 103.95]	[-19.56, 112.78]	[-81.98, 16.14]	[13.86, 87.37]
BD_gcm3	2.84	5.71	8.48	-5.93	-3.46
	[-6.70, 12.37]	[-30.35, 41.77]	[-10.19, 27.14]	[-17.27, 5.40]	[-12.65, 5.73]
Tot_SOC_tha	0.05	0.51	0.28	**0.46**	0.17
	[-0.12, 0.22]	[-0.62, 1.63]	[-0.19, 0.75]	[0.05, 0.88]	[-0.22, 0.56]
nPotential_C_SOC	0.22	0.35	-0.25	-0.36	0.43
	[-0.23, 0.68]	[-0.81, 1.52]	[-1.64, 1.14]	[-1.18, 0.45]	[-0.34, 1.21]
P_mgl	-0.01	0.00	-0.11	0.04	0.02
	[-0.14, 0.12]	[-0.29, 0.30]	[-0.39, 0.18]	[-0.06, 0.14]	[-0.08, 0.13]
pH_mgl	0.61	-1.88	-7.11	6.78	**-6.11 ***
	[-2.46, 3.69]	[-9.62, 5.86]	[-17.51, 3.29]	[-0.42, 13.99]	[-11.19, -1.04]
FieldSelwoods_Bank	**3.83 ***				
	[0.89, 6.78]				
FieldSelwoods_Barn	1.32				
	[-1.11, 3.76]				
FieldPT_Behind_House		3.07			
		[-5.39, 11.54]			
FieldPT_Lane		-1.75			
		[-8.00, 4.49]			
FieldKnapps_Left_2nd			-5.99		
			[-12.99, 1.00]		
FieldKnapps_Left_3rd			-3.56		
			[-7.37, 0.24]		
FieldBackside				-5.90	
				[-11.44, -0.36]	
FieldGordons 2				0.72	
				[-3.05, 4.50]	
FieldGordons_1				5.61	
				[-0.25, 11.46]	
FieldResevoir_Field					-2.06
					[-5.43, 1.31]
N	24	23	21	15	23
R2	0.38	0.19	0.38		
AIC	103.25	143.79	112.44	64.96	112.98
BIC	113.86	154.01	121.84	72.04	122.06
Pseudo R2				0.74	0.32

*** p < 0.001; ** p < 0.01; * p < 0.05.

Appendix C:

Appendix C.1: Measuring soil natural capital and soil-based ecosystem services

It was highlighted in the introduction the difficulties associated with linking measurements of soil condition and soil functions to soil-based ES. This study set out to investigate a suite of ecosystem goods and services and the selection of metrics to measure these is detailed below. The objective was to select measurements of properties or where more appropriate soil functions, which could be linked to soil-based ES, attempting to trace these through the framework presented in Figure 1. A number of these metrics, such as soil carbon, play an important role in the delivery of multiple soil-based ES, influencing soil structure, fertility and biological activity.

Soil structure and stability (Water cycle regulation and the provision of food)

The physical structure of the soil influences a range of soil functions and poor soil structure can reduce crop yields, increase soil erosion (causing pollution issues) and exacerbate flooding. This can have significant economic consequences (Graves et al. 2015); soil compaction in England and Wales estimated at costing £204 million, associated with losses in crop productivity and a further £168 million in flood damages (Graves *et al.*, 2015). A frequent measure of soil structure, and one frequently used in soil-based ES studies is soil bulk density (Greiner *et al.*, 2017). Bulk density (BD), the mass of a unit volume of dry soil is an important indicator in the level of pore space (i.e. the space available for air and water) within a soil, providing information on the level compaction (Cardoso *et al.*, 2013). BD is heavily impacted by soil management, with for example, long-term tillage considered to increase BD through depletion of soil organic matter (SOM), compaction and weakening soil structure (Brady and Weil, 2008).

The stability of soil and its resilience to management or natural events (e.g. heavy rainfall) is also critical in the sustained delivery of soil-based ES. Soil stability is thought to be linked to the content of soil organic carbon (SOC) and fine soil particles (clay and silt), which become associated in the development of soil aggregates (Merante et al. 2017). Micro and macro soil aggregates and their arrangement are critical in the support of soil functions dictating water retention and movement, soil aeration and biological activity (Merante *et al.*, 2017). A high n-potential (>10) suggests a low SOC relative to clay content, suggesting the presence of non-complexed clays which are more easily dispersed in water and more vulnerable to soil degradation (e.g. compaction or erosion). A low n-potential (<10) suggests a high SOC relative to clay content, showing that most of the clay is likely complexed with SOC which increases soil stability. The indicator has the additional benefit of providing insight into how much additional capacity a soil has to store carbon (i.e. high n-potentials with non-complexed clay have the capacity to bind to and store more carbon).

In this study BD and n-potential are adopted as indicators of the capacity for soil to filter and store water (reducing run-off and flood risk), support crop growth (increasing the provision of food) and be resilient to management and soil erosion (impacting damages to other NC forms).

Soil fertility and a medium for plant growth (The provision of food)

The provision of food and fibre is a critically important ecosystem good and soil is the most important medium for plant growth, supporting the growth of crops and livestock forage. With global population set to exceed 9 billion by 2050 it is important that we understand the capacity for different agricultural systems to provide crops that support human nutrition (Muller *et al.*, 2017).

Crop growth and yield potential is strongly influenced by the macro and micro nutrients available to the plant and the pH of soil (which limits the uptake of nutrients). These nutrients are the building blocks of plant and animal life with, N, P, and K all of particular importance for plant growth (Brady and Weil, 2008; Dungait *et al.*, 2012), along with other nutrients such as sulphur (S), calcium (Ca) and Mg. Soil organic matter is critical in the release and storage of these nutrients (Brady and Weil, 2008). These can be considered important properties of soil NC, contributing to the soil function of plant growth and ultimately the provision of food. The study combines measurements of these properties (as indicators of soil fertility) alongside field collected data on crop biomass yield (i.e. the soil function) to consider the capacity for each system to deliver the provision of food.

Carbon storage (Climate regulation)

Interest in the capacity for soils to sequester carbon and mitigate climate change has been growing in interest in recent years (Minasny *et al.*, 2017), with soil management recognised as being critical in determining whether soils act as a net sink or source of greenhouse gasses (FAO, 2017; Paustian *et al.*, 2019a). Minasny et al., (2017) present the ‘4 per mille Soils for Food Security and Climate” launched by the COP21, which has the goal of increasing soil organic matter stocks by 0.4% per year. The paper highlights the significant role soils can play in storing carbon, suggesting that meeting the 4 per mille targets in the top 1m of global agricultural soils could lead to the sequestration of between 2 – 3 Gt C year^{-1}, effectively offsetting 20 – 35% of global anthropogenic greenhouse gas emissions (Minasny *et al.*, 2017). Whilst there is disagreement over whether the ‘4 per mille’ target is feasible (Van Groenigen *et al.*, 2017; Poulton *et al.*, 2018) it is widely accepted that maintaining soil carbon storage and increasing sequestration will help regulate the climate (Paustian *et al.*, 2019a), whilst contributing the delivery of other soil-based ES.

It was beyond the scope of this study to measure soil carbon over time and as such it was not possible to measure the rate of carbon sequestration. Instead carbon stocks were calculated, a frequently used indicator in soil-based ES studies (Greiner *et al.*, 2017), to compare current carbon storage between the two systems. The assumption was made that organic soil conditions were very similar to neighbouring conventional fields prior to conversion in 2007, enabling an assessment of whether conversion to organic agriculture has increased carbon storage over time. Carbon stocks are considered alongside n-potential to suggest whether soils have the capacity to store more carbon.

Organic matter decomposition (Supporting nutrient cycling and carbon sequestration)

The decomposition of organic material (e.g. plant litter or farm manures) by soil biota is an important soil process playing a critical role in the cycling of nutrients and carbon (Keuskamp *et al.*, 2013; Ghaley *et al.*, 2014a). The importance of each has been highlighted previously, relating to the contribution to the delivery of soil-based ES, crop growth and the provision of food and the regulation of the climate. The breakdown of organic material is critical for the development of plants, ensuring that the nutrients become available through decomposition and mineralisation (Ghaley *et al.*, 2014a). The processing of organic material and the role of soil biota is also integral to the carbon cycle influencing whether soil becomes a carbon sink or source (Paustian *et al.*, 2019a; Ray *et al.*, 2020). There is a growing interest in the importance of soil biology in the delivery of soil-based ES (Ritz *et al.*, 2009; Griffiths *et al.*, 2016) and decomposition rate was measured in this study to gain an important insight into the biological activity of a soil.

Nutrient storage and filtration (The provision of drinking water)

Soil plays a critical role in filtering and storing nutrients, with soil condition and soil management determining the risk of nutrient losses to surface and ground waters (Beaudoin *et al.*, 2005; Harris *et al.*, 2006; Knudsen *et al.*, 2006). A particular issue is the loss of nitrate, with nitrate contamination having both environmental and economic consequences, resulting a loss of valuable farm nutrients for land managers and increasing treatment drinking water treatment costs and prices paid by consumers (Stuart and Lapworth, 2016). Groundwater is a particularly important resource, supplying 30% of the UK's drinking water (EA, 2018), but the quality of groundwater aquifers has declined significantly as a consequence of increased nitrate leaching from agricultural land (Wang *et al.*, 2016). Aquifers now regularly exceeding drinking water standards laid out in the EU Drinking Water Directive (Stuart and Lapworth, 2016). The current loss of nitrate from agricultural soils can therefore be considered an ecosystem disservice, with management strategies to reduce nitrate leaching (below the drinking water limit of 50 mg NO_3 l^{-1}) offering the potential to improve the quality and provision of drinking water. Nitrate leaching was measured to determine the scale of losses from each system and the implications this could have on drinking water supply.

Appendix C.2:

Porous pot site comparisons

The typical organic rotation at the Home Farm (Farm 3):

Year	Crop	Following:	Entry into:	Field pair (seasons porous pots sampled)
Summer 1	Spring wheat	Grass	CC or stubble turnips	
Winter 1	CC or Stubble turnips (grazed)	Spring wheat	Spring triticale	Pair 3 (2019 – 2020)
Summer 2	Spring triticale	CC or stubble turnips	CC or stubble turnips	
Winter 2	CC or stubble turnips (grazed)	Spring triticale	Spring triticale	Pair 2 (2018 – 2019)
Summer 3	Spring triticale	CC or stubble turnips	Grass clover ley	
Winter 3	Grass clover ley	Grass clover ley		Pair 2 (2019 – 2020)
Summer 4	1st season Grass	Grass clover ley		
Winter 4	Grass clover ley	Grass clover ley		Pair 1 (2018 – 2019)
Summer 5	2nd season Grass	Grass clover ley		
Winter 5	Grass clover ley	Grass clover ley		Pair 1 (2019 – 2020)
Summer 6	3rd season Grass	Grass clover ley		
Winter 6	Grass clover ley	Grass clover ley	Spring wheat	Pair 3 (2018 – 2019)

Note: This is just the 'general' organic rotation and it is subject to adjustment based on field performance and cropping regime. For example the organic field selected in Pair 1 was not kept in grass for the full three seasons as planned. Instead it was ploughed in spring 2020 due to poor grass growth the previous year. All cultivations took place after the completion of porous pot sampling however and therefore this field is considered as a reference site for grass staying in grass.

Pair 1: Aiming to capture nitrate leaching data from the grass – grass phase of the rotation:

2018/19 winter – Grass – grass phase of the rotation

2019/20 winter – Grass – grass phase of the rotation

Variable	Organic field (Behind Sawmills)	Conventional field (Gordons 1)
Years of organic rotation covered	Winters 4 and 5	
Seed	Rye-grass white clover	Rye-grass white clover
Ley age	Seeded summer 2017	Seeded winter 2016
Cultivation depth/type	6 to 10 inches. Mouldboard plough.	8 – 10 inches. Mouldboard plough.
Inputs	2018: 52 kg N ha (as dairy slurry) 2019: 110.5 kg N ha (as dairy slurry)	2018: 114.7 kg N ha as artificial fertiliser 2019: 124.7 kg N ha as artificial fertiliser
Outputs	3 cuts silage per year	2 haylage cuts per year
Soil	Bromsgrove 541b	Bromsgrove 541b
Slope	Field mean: 3.49 degrees But field variable, site selected would be fairly flat	Field mean: 8.16 deg But field down sloping with fairly flat base which could be selected for pots. Apparent this was old ADAS site for NVZ research.
Aspect	Field mean: 154 deg (south east) But site selected would be mostly south facing.	Field mean: 299 deg (north west) But site selected is mostly west facing.
Field size	3.52 ha	1.63ha
Groundwater protection zone	1	3

Pair 2: Aiming to capture nitrate leaching data from the arable – arable and arable – grass (or as close as possible) phases of the rotation:

Variable	Organic field (Selwoods Barn)	Conventional field (Staddons)
Years of rotation covered	Winters 2 and 3	
2018/19 winter crop/cover	Cover crop (grazed)	Stubble turnips (grazed)
2019 harvest crop	Spring wheat	Maize
2019/20 winter crop/cover	Cover crop or into grass	Winter barley (selected as being close as possible to recently established grass crop)
2020 harvest crop	Grass	Winter barley
Cultivation depth/type	6 to 10 inches for cover or turnips plough out in Jan/Feb. Stubble turnips/cover crop sown with single pass topdown cultivator.	Plough first, 8 – 10inches unless left as maize stubble, in this case a Heva deep legged cultivator used to around 1ft depth. Followed by single pass cultivator, rotivator with a drill on it.
N Inputs	2018: 42.2 kg N ha (as FYM) 2019: 96.5 kg N ha (as FYM and dairy slurry)	2018: 230.9 kg N ha (as FYM and artificial fertiliser) 2019: 153.8 kg N ha (as FYM and artificial fertiliser)
Outputs	Wholecrop wheat (2019) and silage grass (2020).	Forage maize, combinable winter barley and grazed winter forage crops.
Soil	Bromsgrove 541b	Bromsgrove 541b
Slope	Field mean: 4.84 deg Fairly sloping field but flatter areas to top and to the west of the field that could be selected.	
Aspect	Field mean: 156 deg (south east) Mostly south east facing	
Field size	8.91ha	
Groundwater protection zone	3	3

Pair 3: Aiming to capture nitrate leaching data from the grass – arable and arable – arable phase of the rotation (with winter grazing):

Variable	Organic field (Chantry)	Conventional field (Bicton 3 – High Banks Lower)
Years of organic rotation covered	Winters 6 and 1	
2018/19 winter crop/cover	Rye-grass clover ley (seeded 2016). Ploughed out Jan/Feb.	Rye-grass clover ley (seeded in 2015). Potential ploughed out in April/May
2019 harvest crop	Spring wheat (in 1st week Feb)	Original plan: Forage rape. Grazed off by June/July. Actual cropping: Fodder beet kept in until spring 2020
2019/20 winter crop/cover	Cover crop or stubble turnips (grazed)	Original plan: Fast grass – rapid 1yr temporary ley (late autumn/winter grazed) Actual cropping: Fodder beet in until spring 2020
2020 harvest crop	Spring triticale	Maize
Cultivation depth/type	6 to 10 inches for grass plough out. Topdown one pass cultivator used for cover and stubble turnips.	Mouldboard ploughed out of grass ley. Light cultivation. Triple k pig tines. Into forage. Triple K spring tine cultivator into maize.
N Inputs	2018: 87.7 kg N ha (as dairy slurry) 2019: 41.5 kg N ha (as FYM)	2018: 106.4 kg N ha (as artificial fertiliser) 2019: 110.5 kg N ha (as FYM and artificial fertiliser)
Outputs	Combinable wheat crop (2019), wholecrop triticale (2020) and grazed winter forage crops over winter.	Three silage cuts off long term grass leys. 1 spring cut and later autumn/winter grazing off 1 yr leys. Silage maize for clamp and grazed fodder beet.
Soil	Bromsgrove 541b	Bromsgrove 541b

Slope	Field mean: 4.3 degrees But highly variable. Site selected was fairly flat.	Field mean: 3.7 deg. Site selected flat, lower part of people sloping.
Aspect	Field mean: 162 deg (south) Site selected would be south/south west facing	Field mean: 210 deg (south west)
Field size	3.6ha	3.3ha
Groundwater protection zone	1	3

Appendix C.3: Results from mixed effects models comparing differences between organic and conventional field sites (As discussed in results Section 5.3)

	Soil carbon stock	Biomass yield	Soil nitrogen stock	N-Potential	TBI_K (glmer)	BD (glmer)
(Intercept)	25.77 ***	13.74 ***	2.44 ***	10.67 ***	46.70 ***	1.30 ***
	[23.78, 27.76]	[11.80, 15.68]	[2.18, 2.69]	[9.40, 11.93]	[40.42, 52.98]	[1.18, 1.42]
treatmentorg	3.12 **	-5.00 ***	0.38 *	-0.28	4.68	0.04
	[1.13, 5.11]	[-6.94, -3.06]	[0.12, 0.64]	[-1.54, 0.99]	[-1.60, 10.96]	[-0.08, 0.16]
sub_treatmentgrass_arable	2.74 *	2.00	0.24	-0.75	-4.34	0.02
	[0.30, 5.17]	[-0.38, 4.37]	[-0.08, 0.56]	[-2.30, 0.79]	[-12.05, 3.37]	[-0.13, 0.17]
sub_treatmentgrass_grass	1.41	-1.43	0.16	-0.29	0.51	0.07
	[-1.02, 3.85]	[-3.80, 0.95]	[-0.16, 0.47]	[-1.83, 1.26]	[-7.17, 8.18]	[-0.07, 0.22]
N	54	54	53	54	54	54
N (field)	18	18	18	18	18	18
AIC	273.37	240.85	84.81	238.73	-473.39	-46.59
BIC	285.30	252.79	96.63	250.66	-461.46	-34.66
R2 (fixed)	0.28	0.57	0.18	0.03		
R2 (total)	0.45	0.80	0.22	0.15		

*** $p < 0.001$; ** $p < 0.01$; * $p < 0.05$.

Appendix D:

Appendix D.1: Flower list including all recorded species, those used in the Bee-Steward (BS) model runs for determining bumblebee population (i.e. plants utilised by bumblebees) and those used to calculate nectar (N) and pollen (P) across the landscape.

Latin	Common	Count	Flower Count Method	Incl. in BS model	Used in N and P calcs.	Pollen source	Nectar source	Pollen protein source
Achillea millefolium	Yarrow	16	single capitulum	y	y	Hicks	Hicks	
Aethusa cynapium	Fools_parsley	1223	individual flower	n	y	Hicks (Apiacaceae pollen data)	Baude	
Anagallis arvensis	Scarlet_pimpernel	5557	individual flower	y	y	Gibbs and Talavera (2001)	Baude	
Anchusa officinalis	Common_bugloss	6616	individual flower	y	y	Beehave	Beehave	Beehave
Anthriscus sylvestris	Cow_parsley	20	floral unit	n	n	na	na	
Arabidopsis thaliana	Thale_cress	5	individual flower	n	n	na	na	
Capsella bursa-pastoris	Shepherds_purse	2501	individual flower	n	y	Hicks	Hicks	
Carduus nutans	Musk_thistle	4341	Single capitulum	y	y	BeeHave (spear thistle pollen data)	BeeHave (spear thistle nectar data)	BeeHave (spear thistle pollen data)
Cerastium fontanum	Common_Mousear	45	individual flower	n	y	Hicks	Hicks	
Cichorium intybus	Chicory	5563	Single capitulum	y	y	Jablonski and Koltowski (2000)	Jablonski and Koltowski (2000)	Jablonski and Koltowski (2000)
Cirsium arvense	Creeping_thistle	260	Single capitulum	y	y	Hicks	Hicks	Hicks
Crepis capillaris	Smooth_hawksbeard	9249	Single capitulum	y	y	Hicks	Baude	
Daucus carota	Wild_carrot	12068	floral unit	y	y	Hicks	Hicks	
Epilobium ciliatum	American_willowherb	17	individual flower	y	y	Hicks	Hicks	
Erodium cicutarium	Common_storks_bill	392	individual flower	y	y	Hicks (Doves foot cranesbill pollen data)	Hicks (Doves foot cranesbill pollen data)	
Ervilia hirsuta/Vicia hirsuta	Hairy_tare	1172	individual flower	n	y	Hicks	Hicks	
Fagopyrum esculentum	Buckwheat	2809	individual flower	y	y	Hicks - (knotgrass pollen data)	Baude (knotgrass nectar data)	Hicks - knotgrass (same family)

Fumaria officinalis	Common_fumitory	28318	individual flower	y	y	Hicks	Hicks	
Galium aparine	Cleavers	153	individual flower	n	n	na	Baude	
Geranium dissectum	Cut_leaved_cranesbill	184	individual flower	y	y	Hicks (Doves foot cranesbill pollen data)	Hicks (Doves foot cranesbill nectar data)	
Geranium molle	Dovesfoot_cranesbill	270	individual flower	y	y	Hicks	Hicks	
Glebionis segetum	Corn_marigold	10	individual flower	n	y	Hicks	Hicks	Hicks
Heracleum sphondylium	Common_hogweed	78	floral unit	n	n	na	na	
Hypochaeris radicata	Common_cats_ear	80	Single capitulum	y	y	Hicks	Hicks	Hicks
Lamium amplexicaule	Henbit_dead_nettle	41	individual flower	y	y	Beehave (Red dead nettle pollen data)	Beehave (red dead nettle nectar data)	Beehave (Red dead nettle pollen data)
Lamium hybridum	Cut_leaved_dead_nettle	77	individual flower	y	y	Beehave (Red dead nettle pollen data)	Beehave (red dead nettle nectar data)	Beehave (Red dead nettle pollen data)
Lapsana communis	Nipplewort	7	Single capitulum	n	y	Hicks	Hicks	
Leucanthemum vulgare	Ox_eye_daisy	346	Single capitulum	y	y	Hicks	Hicks	Hicks
Malva sylvestris	Common_mallow	22	floral unit	y	y	Hicks (musk mallow pollen data)	Hicks (musk mallow nectar data)	
Matricaria discoidea	Pineapple_weed	200	Single capitulum	n	y	Hicks	Hicks	
Matricaria recutita	Scented_mayweed	128	Single capitulum	n	y	Hicks (scentless mayweed pollen data)	Hicks (scentless mayweed nectar data)	Hicks (scentless mayweed pollen data)
Medicago lupulina	Black_medick	4680	individual flower	y	y	no pollen data	Baude	
Medicago sativa	Lucerne	30	individual flower	y	y	Calculated	Baude	Forcone et al., (2011)
Mentha arvensis	Corn_mint	145	individual flower	y	y	Hicks - (mean pollen data for Lamiaceae)	Baude (Thymus polytrichus nectar data)	Roulston, Cane and Bucmann 2000 - mean For Lamiaceae
Misopates orontium	Weasels_snout	354	individual flower	y	y	no pollen data	Baude (Common Toadflax nectar data)	
Myosotis arvensis	Field_forgetmenot	112	individual flower	y	y	Hicks	Hicks	
Orobanche minor	Common_broomrape	637	individual flower	n	n	na	Baude (Euphrasia agg. Nectar data)	
Papaver rhoeas	Common_poppy	1372	individual flower	y	y	Hicks	Hicks	Hicks

Persicaria maculosa	Redshank	72	individual flower	y	y	Hicks	Hicks	
Phacelia tanacetifolia	Lacy_phacelia	29417	individual flower	y	y	Owyass et al., (2020)	Owyass et al., (2020) - coarse estimate of sugar content	Pernal et al., (2015)
Polygonum aviculare agg.	Knotgrass	142375	individual flower	n	y	Hicks	Baude	
Raphanus raphanistrum	Wild_radish	80030	individual flower	y	y	Hicks (charlock pollen data)	Baude	Hicks (charlock pollen data)
Senecio vulgaris	Groundsel	10607	Single capitulum	n	y	Hicks	Baude	
Sherardia arvensis	Field_madder	688	individual flower	n	y	Calculated	Calculated	
Silene dioica	Red_campion	37	individual flower	y	y	Hicks	Hicks	
Silene latifolia	White_campion	39	individual flower	y	y	Hicks (red campion pollen data)		
Sinapis alba L.	White_mustard	256	individual flower	y	y	Hicks - charlock	Hicks - charlock	Hicks - charlock
Sisymbrium officinale	Hedge_mustard	1111	individual flower	n	y	Hicks	Hicks	Hicks
Solanum nigrum	Black_nightshade	974	individual flower	y	y	Various - See Pollen spreadsheet	Baude	Roulston, Cane and Buchmann, (2000)
Sonchus asper	Prickly_sow_thistle	6913	Floral unit (inflorecence)	y	y	Hicks	Hicks	
Spergula arvensis/Spergularia rupicola	Corn_spurrey	6884	individual flower	n	y	no pollen data	Baude	
Stachys arvensis	Field_woundwort	2770	individual flower	y	y	Beehave (hedge woundwort pollen data)	Beehave - hedgewoundwort	Beehave - hedgewoundwort
Stellaria media	Common_chickweed	271	individual flower	n	y	Hicks	Hicks	
Thlaspi arvense	Field_penny_cress	60	individual flower	n	n	na	na	
Trifolium incarnatum	Crimson_clover	17630	individual flower	y	y	Beehave (red clover pollen data)	Beehave - red clover	Beehave - red clover
Tripleurospermum inodorum	Scentless_mayweed	20324	Single capitulum	n	y	Hicks	Hicks	Hicks
Veronica arvensis	Wall_speedwell	2	individual flower	n	y	Hicks	Hicks	
Veronica persica	Field_speedwell	4950	individual flower	y	y	Hicks	Hicks	
Veronica serpyllifolia	Thyme_leaved_speedwell	94	individual flower	y	y	Hicks (field speedwell pollen data)		
Viola arvensis	Field_pansy	7181	individual flower	y	y	Calculated using other viola species	Baude	
Cultivar - unknown latin name	Mariana_vetch	14	individual flower	y	y	Beehave (common vetch pollen data)	Beehave (common vetch nectar data)	Beehave (common vetch pollen data)

Helianthus annuus	Sunflower	220	Single capitulum	y	y	Beecher 2016	Beecher 2016	Pernal and Currie 2000
Trifolium hybridum	Alsike_clover	1045	individual flower	y	y	Beehave	Beehave	Beehave
Brassica napus	Oilseed_rape	82	individual flower	y	y	Beehave	Beehave	Beehave
Centaurea nigra	Common_knapweed	4339	Single capitulum	y	y	Beehave	Beehave	Beehave
Cirsium vulgare	Spear_thistle	922	Single capitulum	y	y	Beehave	Beehave	Beehave
Lamium purpureum	Red_dead_nettle	1650	individual flower	y	y	Beehave	Beehave	Beehave
Lotus corniculatus	Birdsfoot_trefoil	7638	individual flower	y	y	Beehave	Beehave	Beehave
Ranunculus	Buttercup	976	individual flower	y	y	Beehave	Beehave	Beehave
Jacobaea vulgaris	Ragwort	74	individual flower	y	y	Beehave	Beehave	Beehave
Taraxacum agg.	Dandelion	283	Single capitulum	y	y	Beehave	Beehave	Beehave
Trifolium pratense	Red_clover	126905	individual flower	y	y	Beehave	Beehave	Beehave
Trifolium repens	White_clover	726717	individual flower	y	y	Beehave	Beehave	Beehave
Vicia sativa	Common_vetch	58	individual flower	y	y	Beehave	Beehave	Beehave

Appendix D.2: Mowing protocol

For organic and conventional grass leys the annual average flower density was calculated from the five survey visits. Abundance immediately after grass cutting was determined by conducting a pre and post-cutting test across six organic rye-grass clover transects. Flower counts were conducted on each transect immediately prior and immediately after silage cutting in June to determine the relative abundance after grass cutting. For organic cereals 2 and 3 the impact of cereal harvest and cultivations for the following crop were captured through the five survey visits. The average annual flower density for each flower species was calculated using the May, June and September visits, which represent the flowers in the field prior to and after the disturbance period. Late July and August transect data were then used to calculate the mean relative abundance of flowering plants following harvest. An overview of the mowing protocol for the model can be found in Appendix 2.

Flower name	Management date	End date	Date source for flower density
Organic cereals 2 and 3:			
"Weed_species_1"	Based on phenology	200 (19th July/harvest date)	Transect survey means from May and June
"Weed_species_cut"	200	244	Calc. in model using mean relative abundance of flowers from late July and August transect surveys.
"Weed_species_2"	244	Based on phenology	Transect survey data from late September
"Flower_S1"	Based on phenology	179 (end June.1st cut)	Transect survey means May and June
"Flower_cut_1"	179	200	3.5% density of existing plants in May/June surveys*
"Flower_S2"	200	231 (mid August 2nd cut)	Transect survey means for July and August
"Flower_cut_2"	231	252	3.5% density of plants from July/August surveys
"Flower_S3"	252	Based on phenology	

*The 3.5% is based on field examination of the amount of clover and other plants remaining immediately before and immediately after silage cutting. The data suggests a 96.5% reduction in flowers.

Appendix D.3: Flower species list with pollen and nectar data used for BEE-STEWARD simulations

Flowerspecies	pollen_g/flower	nectar_ml/flower	proteinPollenProp	concentration_mol/l	start Day	stopDay	corollaDepth_mm	intFlowerTime_s
"Heather"	0.00114	0.000098	0.139	0.9436	182	273	3	0.6
"Bell_heather"	0.000733	0.000249	0.139	0.9436	121	334	5.5	0.6
"Alsike_clover"	0.000502	0.000616	0.208688	0.985977	181	242	10	0.6
"Bugle"	0.00065	0.00081	0.072104	0.824738	120	211	10	0.6
"Burdock"	0.00043	0.002289	0.11179	0.886487	181	272	3.9	0.6
"Oilseed_rape"	0.001507	0.021032	0.256083	1.413034	120	242	5	0.6
"Giant_bindweed"	0.00091	0.009954	0.264567	0.664622	181	272	0	0.6
"Common_knapweed"	0.0024	0.002104	0.158971	1.340767	151	272	3	0.6
"Greater_knapweed"	0.002023	0	0.29775	0	181	272	13.6	0.6
"Rosebay_willowherb"	0.01145	0	0.205722	0	181	272	0	0.6
"Marsh_thistle"	0.005053	0.000639	0.145379	0.892359	181	272	3	0.6
"Spear_thistle"	0.003067	0.001825	0.190333	1.290291	151	303	6.2	0.6
"Hawthorn"	0.000113	0.001875	0.153979	1.023306	120	180	0	0.6
"Foxglove"	0.021637	0.001633	0.227517	0.824326	151	272	7	0.6
"Wild_teasel"	0.014552	0.009761	0.198505	1.085792	181	242	10	0.6
"Vipers_bugloss"	0.001737	0.000922	0.180278	0.668138	151	272	6.7	0.6
"Ground_ivy"	0.000897	0.002618	0.190697	0.872367	59	150	7	0.6
"Bluebell"	0.001877	0	0.363	0	90	180	0	0.6
"St_Johns_wort"	0.0005	0	0.139074	0	151	272	0	0.6
"Field_scabious"	0.00888	0	0.1195	0	181	272	0	0.6
"White_dead_nettle"	0.00122	0.002168	0.228	0.756159	120	364	7.7	0.6
"Red_dead_nettle"	0.000673	0.005453	0.228	1.012757	59	303	7	0.6
"Birdsfoot_trefoil"	0.000983	0.000843	0.358	0.697634	120	303	9	0.6
"Selfheal"	0.000341	0.000582	0.258046	0.662448	151	272	8	0.6
"Blackthorn"	3.33E-05	9.33E-05	0.272	0.779652	59	150	0	0.6
"Buttercup"	0.000766	0.000197	0.1206	0.715746	120	303	0	0.6
"Dog_rose"	0.000669	0	0.090724	0	151	211	0	0.6
"Bramble"	0.000479	0.006824	0.126	0.500418	120	272	0	0.6
"Average_Willow"	0.010303	0.002607	0.257375	1.134482	59	119	0	0.6
"Ragwort"	0.00017	0	0.155	0	151	303	0	0.6
"Hedge_woundwort"	0.0008	0.001465	0.145433	1.044243	181	242	9	0.6
"Comfry"	0.000953	0.004318	0.097107	0.989791	120	211	17	0.6
"Dandelion"	0.000433	0.00047	0.091663	1.294673	1	364	1.2	0.6
"Red_clover"	0.000502	0.000616	0.208688	0.985977	120	277	10	0.6
"White_clover"	0.000413	0.000667	0.2307	0.980297	136	276	2	0.6
"Tufted_vetch"	0.00085	0.001587	0.129583	0	151	242	6.8	0.6
"Common_vetch"	0.00038	0.00086	0.428	0.81534	120	277	7	0.6
"Crop_Field_beans"	0.00065	0.00086	0.2	1.28	153	182	19	0.6
"Crop_Oilseed_rape"	0.000239	0.00055	0.2	1.5	114	136	5	0.6
"Crop_Maize"	0.0353	0	0.2	0	197	210	0	0.6
"Crop_Cereals"	0	0	0	0	0	0	0	0.6

"American_willowherb "	2.78E-05	0.000447	0.205722	0.9436	152	277	3.5	0.6
"Black_medick"	0	0.000005	0.225	0.9436	91	243	2.5	0.6
"Buckwheat"	0.000024	0.000009	0.2104	0.9436	182	276	0.53	0.6
"Black_nightshade"	0.000002	0	0.514	0	182	277	0	0.6
"Common_bugloss"	0.001737	0.000922	0.180278	0.668138	151	276	6	0.6
"Common_cats_ear"	0.002972	0.005689	0.17	0.9436	152	274	6.2	0.6
"Chicory "	0.029001	0.00261	0.1534	0.9436	172	276	6.33	0.6
"Common_fumitory"	0.000039	0.000068	0.191	0.9436	121	303	7.5	0.6
"Common_mallow"	0.018516	0.001669	0.21657	0.9436	152	274	0	0.6
"Common_poppy "	0.048511	0.000002	0.191	0.9436	152	304	0	0.6
"Common_storks_bill"	0.00027	0.000059	0.197396	0.9436	152	276	6.8	0.6
"Corn_mint"	0.000136	0.000076	0.228	0.9436	121	303	4	0.6
"Creeping_thistle"	0.000844	0.008052	0.219	0.9436	182	277	1.2	0.6
"Crimson_clover"	0.000502	0.000616	0.208688	0.985977	121	175	8.2	0.6
"Cut_leaved_cranesbill "	0.00027	0.000059	0.197396	0.9436	121	243	6.8	0.6
"Dovesfoot_cranesbill "	0.00027	0.000059	0.197396	0.9436	91	273	6.8	0.6
"Field_forgetmenot"	0.000001	0.000067	0.180278	0.9436	91	276	0	0.6
"Field_pansy "	0.000968	0.000165	0.197396	0.9436	91	304	14	0.6
"Field_speedwell"	0.000251	0.000014	0.197396	0.9436	1	365	0.7	0.6
"Field_woundwort"	0.0008	0.001465	0.145433	1.044243	91	277	6.5	0.6
"Henbit_dead_nettle "	0.000673	0.005453	0.228	1.012757	60	304	20	0.6
"Lacy_phacelia "	0.000166	0.001355	0.301	0.9436	152	277	4.4	0.6
"Lucerne "	0.000255	0.000451	0.225	0.9436	152	212	3.95	0.6
"Musk_thistle "	0.003067	0.001825	0.190333	1.290291	152	276	12	0.6
"Ox_eye_daisy"	0.008768	0.00159	0.28	0.9436	152	243	1.65	0.6
"Prickly_sow_thistle "	0.000978	0.001833	0.243788	0.9436	152	277	5	0.6
"Rapeseed"	0.001507	0.021032	0.256083	1.413034	120	242	5	0.6
"Red_campion"	0.000936	0.000447	0.197396	0.9436	91	273	16.7	0.6
"Redshank"	0.000019	0.000091	0.2104	0.9436	121	304	0.75	0.6
"Scarlet_pimpernel "	0.000191	0	0.197396	0	152	276	0	0.6
"Smooth_hawksbeard "	0.001183	0.000028	0.243788	0.9436	152	277	3.8	0.6
"Sunflower"	0.0287	0.00081	0.1486	1.25	237	264	3.29	0.6
"Thyme-leaved_speedwell"	0.000251	0.000014	0.197396	0.9436	60	304	0.7	0.6
"Weasels_snout "	0	0.001679	0	0.9436	152	277	12.5	0.6
"White_campion "	0.000936	0.000447	0.197396	0.9436	91	273	16.7	0.6
"Wild_carrot "	0.000018	0.000084	0.29	0.9436	152	277	0	0.6
"Wild_radish"	0.000124	0.000355	0.338	0.9436	121	277	11.96	0.6
"White_mustard"	0.000124	0.000018	0.338	0.9436	121	277	12	0.6
"Yarrow "	0.002012	0.000096	0.243788	0.9436	152	276	2.2	0.6
"Mariana_vetch"	0.00038	0.00086	0.428	0.81534	120	272	7	0.6
"Cut_leaved_dead_nettle"	0.000673	0.005453	0.228	1.012757	60	304	14	0.6

Appendix D.4 Flower species list with pollen and nectar data used to determine pollen and nectar provision at the landscape scale

Flowerspecies	pollen_g/flower	nectar_ml/flower	proteinPollenProp	concentration_mol/l	startDay	stopDay	corollaDepth_mm	intFlowerTime_s
"Heather"	0.00114	0.000098	0.139	0.9436	182	273	3	0.6
"Bell_heather"	0.000733	0.000249	0.139	0.9436	121	334	5.5	0.6
"Alsike_clover"	0.000502	0.000616	0.208688	0.985977	181	242	10	0.6
"Bugle"	0.00065	0.00081	0.072104	0.824738	120	211	10	0.6
"Burdock"	0.00043	0.002289	0.11179	0.886487	181	272	3.9	0.6
"Oilseed_rape"	0.001507	0.021032	0.256083	1.413034	120	242	5	0.6
"Giant_bindweed"	0.00091	0.009954	0.264567	0.664622	181	272	0	0.6
"Common_knapweed"	0.0024	0.002104	0.158971	1.340767	151	272	3	0.6
"Greater_knapweed"	0.002023	0	0.29775	0	181	272	13.6	0.6
"Rosebay_willowherb"	0.01145	0	0.205722	0	181	272	0	0.6
"Marsh_thistle"	0.005053	0.000639	0.145379	0.892359	181	272	3	0.6
"Spear_thistle"	0.003067	0.001825	0.190333	1.290291	151	303	6.2	0.6
"Hawthorn"	0.000113	0.001875	0.153979	1.023306	120	180	0	0.6
"Foxglove"	0.021637	0.001633	0.227517	0.824326	151	272	7	0.6
"Wild_teasel"	0.014552	0.009761	0.198505	1.085792	181	242	10	0.6
"Vipers_bugloss"	0.001737	0.000922	0.180278	0.668138	151	272	6.7	0.6
"Ground_ivy"	0.000897	0.002618	0.190697	0.872367	59	150	7	0.6
"Bluebell"	0.001877	0	0.363	0	90	180	0	0.6
"St_Johns_wort"	0.0005	0	0.139074	0	151	272	0	0.6
"Field_scabious"	0.00888	0	0.1195	0	181	272	0	0.6
"White_dead_nettle"	0.00122	0.002168	0.228	0.756159	120	364	7.7	0.6
"Red_dead_nettle"	0.000673	0.005453	0.228	1.012757	59	303	7	0.6
"Birdsfoot_trefoil"	0.000983	0.000843	0.358	0.697634	120	303	9	0.6
"Selfheal"	0.000341	0.000582	0.258046	0.662448	151	272	8	0.6
"Blackthorn"	3.33E-05	9.33E-05	0.272	0.779652	59	150	0	0.6
"Buttercup"	0.000766	0.000197	0.1206	0.715746	120	303	0	0.6
"Dog_rose"	0.000669	0	0.090724	0	151	211	0	0.6
"Bramble"	0.000479	0.006824	0.126	0.500418	120	272	0	0.6
"Average_Willow"	0.010303	0.002607	0.257375	1.134482	59	119	0	0.6
"Ragwort"	0.00017	0	0.155	0	151	303	0	0.6
"Hedge_woundwort"	0.0008	0.001465	0.145433	1.044243	181	242	9	0.6
"Comfry"	0.000953	0.004318	0.097107	0.989791	120	211	17	0.6
"Dandelion"	0.000433	0.00047	0.091663	1.294673	1	364	1.2	0.6
"Red_clover"	0.000502	0.000616	0.208688	0.985977	120	277	10	0.6
"White_clover"	0.000413	0.000667	0.2307	0.980297	136	276	2	0.6
"Tufted_vetch"	0.00085	0.001587	0.129583	0	151	242	6.8	0.6
"Common_vetch"	0.00038	0.00086	0.428	0.81534	120	277	7	0.6
"Crop_Field_beans"	0.00065	0.00086	0.2	1.28	153	182	19	0.6
"Crop_Oilseed_rape"	0.000239	0.00055	0.2	1.5	114	136	5	0.6
"Crop_Maize"	0	0	0	0	0	0	0	0.6

"Crop_Cereals"	0	0	0	0	0	0	0	0.6
"American_willowherb"	2.78E-05	0.000447	0.205722	0.9436	152	277	3.5	0.6
"Black_medick"	0	0.000005	0.225	0.9436	91	243	2.5	0.6
"Buckwheat"	0.000024	0.000009	0.2104	0.9436	182	276	0.53	0.6
"Black_nightshade"	0.000002	0	0.514	0	182	277	0	0.6
"Common_bugloss"	0.001737	0.000922	0.180278	0.668138	151	276	6	0.6
"Common_cats_ear "	0.002972	0.005689	0.17	0.9436	152	274	6.2	0.6
"Chicory "	0.029001	0.00261	0.1534	0.9436	172	276	6.33	0.6
"Common_fumitory "	0.000039	0.000068	0.191	0.9436	121	303	7.5	0.6
"Common_mallow"	0.018516	0.001669	0.21657	0.9436	152	274	0	0.6
"Common_poppy "	0.048511	0.000002	0.191	0.9436	152	304	0	0.6
"Common_storks_bill"	0.00027	0.000059	0.197396	0.9436	152	276	6.8	0.6
"Corn_mint"	0.000136	0.000076	0.228	0.9436	121	303	4	0.6
"Corn_spurrey"	0.000005	0.000037	0.197396	0.9436	152	273	0	0.6
"Creeping_thistle"	0.000844	0.008052	0.219	0.9436	182	277	1.2	0.6
"Crimson_clover"	0.000502	0.000616	0.208688	0.985977	121	175	8.2	0.6
"Cut_leaved_cranesbill"	0.00027	0.000059	0.197396	0.9436	121	243	6.8	0.6
"Dovesfoot_cranesbill"	0.00027	0.000059	0.197396	0.9436	91	273	6.8	0.6
"Field_forgetmenot"	0.000001	0.000067	0.180278	0.9436	91	276	0	0.6
"Field_madder"	0.000006	0.000029	0.197396	0.9436	91	304	4.5	0.6
"Field_pansy "	0.000968	0.000165	0.197396	0.9436	91	304	14	0.6
"Field_speedwell"	0.000251	0.000014	0.197396	0.9436	1	365	0.7	0.6
"Field_woundwort"	0.0008	0.001465	0.145433	1.044243	91	277	6.5	0.6
"Fools_parsley"	0.000018	0.000021	0.29	0.9436	182	277	0	0.6
"Groundsel"	0.000227	0.000001	0.243788	0.9436	1	365	4	0.6
"Hairy_tare"	0.000052	0.000083	0.261	0.9436	121	277	4	0.6
"Hedge_mustard "	0.00002	0.000011	0.22	0.9436	152	277	0	0.6
"Henbit_dead_nettle "	0.000673	0.005453	0.228	1.012757	60	304	20	0.6
"Knotgrass "	0.000024	0.000009	0.2104	0.9436	182	304	0	0.6
"Lacy_phacelia "	0.000166	0.001355	0.301	0.9436	152	277	4.4	0.6
"Lucerne "	0.000255	0.000451	0.225	0.9436	152	212	3.95	0.6
"Common_Mousear "	0.000287	0.000036	0.197396	0.9436	91	277	4	0.6
"Musk_thistle "	0.003067	0.001825	0.190333	1.290291	152	276	12	0.6
"Ox_eye_daisy"	0.008768	0.00159	0.28	0.9436	152	243	1.65	0.6
"Pineapple_weed"	0.001089	0	0.243788	0.9436	152	212	0.87	0.6
"Prickly_sow_thistle "	0.000978	0.001833	0.243788	0.9436	152	277	5	0.6
"Rapeseed"	0.001507	0.021032	0.256083	1.413034	120	242	5	0.6
"Red_campion"	0.000936	0.000447	0.197396	0.9436	91	273	16.7	0.6
"Redshank"	0.000019	0.000091	0.2104	0.9436	121	304	0.75	0.6
"Scarlet_pimpernel "	0.000191	0	0.197396	0	152	276	0	0.6
"Scented_mayweed "	0.00283	0.004369	0.15	0.9436	152	243	0.87	0.6
"Scentless_mayweed "	0.00283	0.004369	0.15	0.9436	152	277	0.87	0.6
"Shepherds_purse "	0.000008	0.000028	0.271	0.9436	1	365	0	0.6

"Smooth_hawksbeard "	0.001183	0.000028	0.243788	0.9436	152	277	3.8	0.6
"Sunflower"	0.0287	0.00081	0.1486	1.25	237	264	3.29	0.6
"Thyme-leaved_speedwell"	0.000251	0.000014	0.197396	0.9436	60	304	0.7	0.6
"Weasels_snout "	0	0.001679	0	0.9436	152	277	12.5	0.6
"White_campion "	0.000936	0.000447	0.197396	0.9436	91	273	16.7	0.6
"Wild_carrot "	0.000018	0.000084	0.29	0.9436	152	277	0	0.6
"Wild_radish"	0.000124	0.000355	0.338	0.9436	121	277	11.96	0.6
"White_mustard"	0.000124	0.000018	0.338	0.9436	121	277	12	0.6
"Yarrow "	0.002012	0.000096	0.243788	0.9436	152	276	2.2	0.6
"Mariana_vetch"	0.00038	0.00086	0.428	0.81534	120	272	7	0.6
"Common_chickweed"	0.000005	0.000037	0.197396	0.9436	1	365	4	0.6
"Corn_marigold"	0.004962	0.002874	0.182	0.9436	152	194	3.5	0.6
"Cut_leaved_dead_nettle"	0.000673	0.005453	0.228	1.012757	60	304	14	0.6

Appendix D.5: Converting data from Baude and Hicks et al.,

Data from Hicks et al., (2016) and Baude et al., (2016) are presented in different units (nectar sugar; µg day^{-1} and pollen; µL day^{-1}) to the format used in Bee-Steward (nectar; ml day^{-1} and pollen; g day^{-1}). Data from Hicks et al., (2016) and Baude et al., (2016) were converted using the following calculations kindly provided by Grace Twiston-Davies:

Nectar data:

$$Nectar\ as\ ml\ per\ day = (((1 - mol\ sugar\ per\ l) + 1\left(\frac{nectar\ as\ ug\ per\ day}{sucrose}\right) * 1/1000$$

Here mol/l (concentration of nectar sugar per litre) was calculated as 0.9436 mol/l. The amount of the sugar which was considered to be was calculated as sucrose = 342.3 g/mol.

The following rational was used for the calculations:

Convert nectar concentration [%] to mol/l:
32.3% (w/v) concentration is 323g sucrose/l
sucrose molar mass: 342.3 g/mol
32.3% ***concentration*** *as mol/l: 323[g/l] / 342.3 [g/mol] =* ***0.9436 mol/l***

Nectar volume (e.g.):
342g sucrose of 1M = 1l
145g of 1M = 0.4236l
145ug of 1M = 0.4236ul
145ug of 0.9436M = 0.4489ul
or ***0.0004489ml per day***

Pollen data:

$$pollen\ as\ g\ per\ day = volume\ of\ pollen\ (as\ ul) * mean\ weightof\ 1\mu l\ pollen$$

The mean weight of 1µl pollen was taken as 0.008142 g based on existing pollen data from a variety of plants used in Buchmann and Orouke 1991[37].

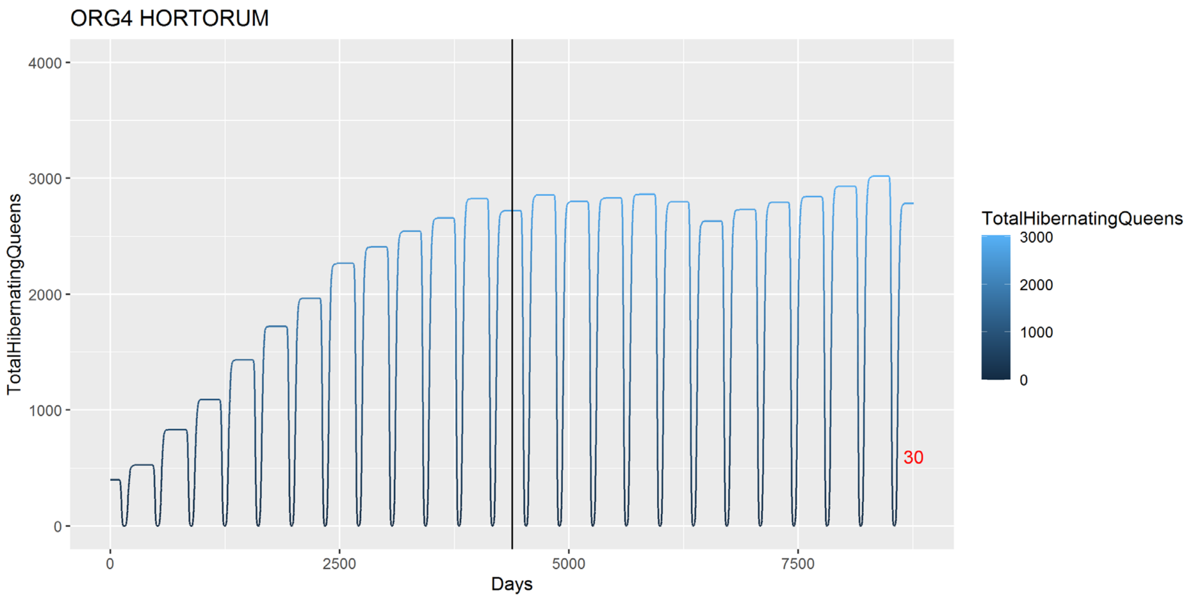
[37] Buchmann, S. L., & O'Rourke, M. K. (1991). Importance of pollen grain volumes for calculating bee diets. GRANA, 30(3-4), 591-595. https://doi.org/10.1080/00173139109427817

Appendix D.6: BEE-STEWARD simulation population stabilisation tests

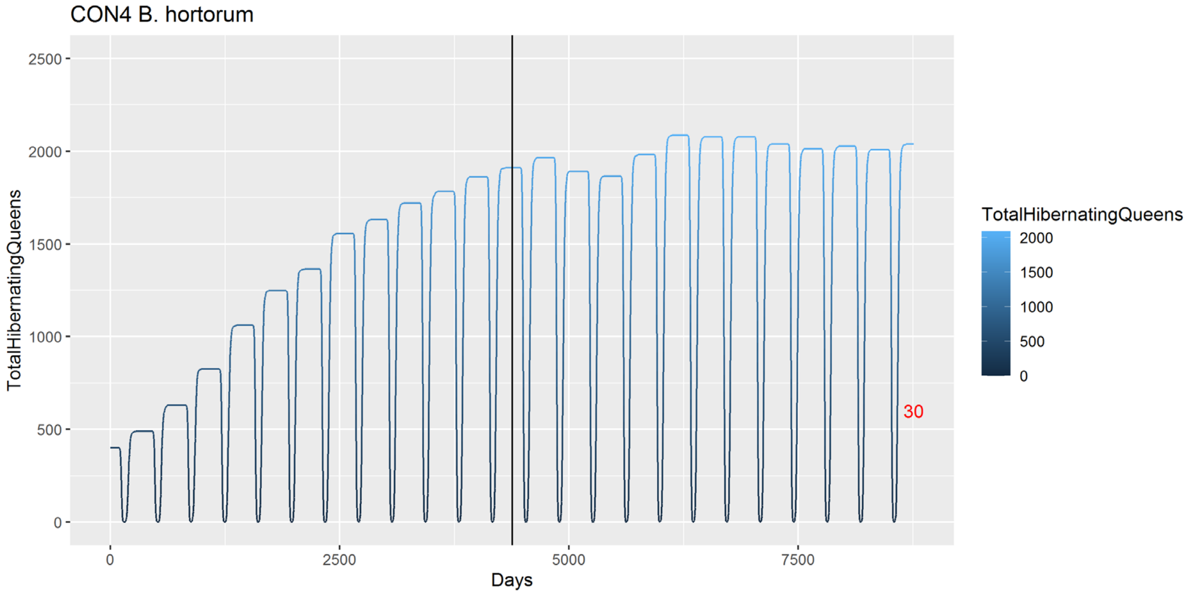
Population stabilisation was determined based on the number of hibernating queens as in (Knapp *et al.*, 2019). The point at which the population stabilised in the model was conducted visually using the mean number of hibernating queens from 30 model runs. This is shown in the plots below for both Bombus hortorum and Bombus terrestris. A 12 year stabilisation point was selected (shown by the black line in the plots) and only data after this point was analysed. The red number shows the number of surviving runs out of 30 model runs. Initial starting queens was set at 400 for both species based on advice from Grace Twiston-Davies and Matthias Becher (i.e. the BEE-STEWARD model development team).

Bombus hortorum stabilisation:

Organic scenario:

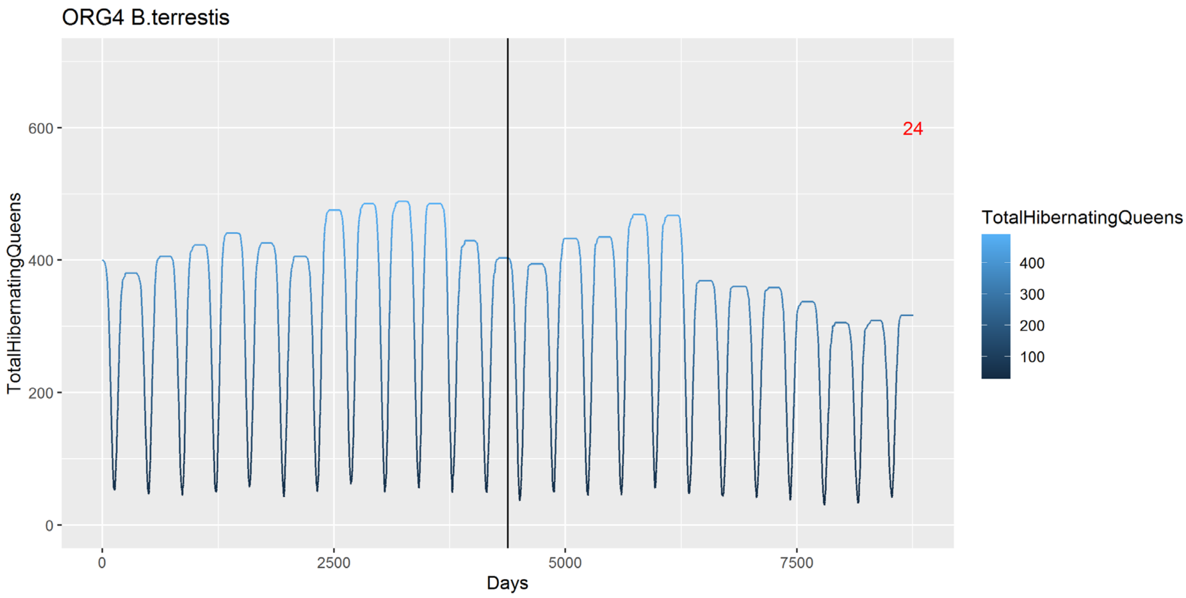


Conventional scenario:



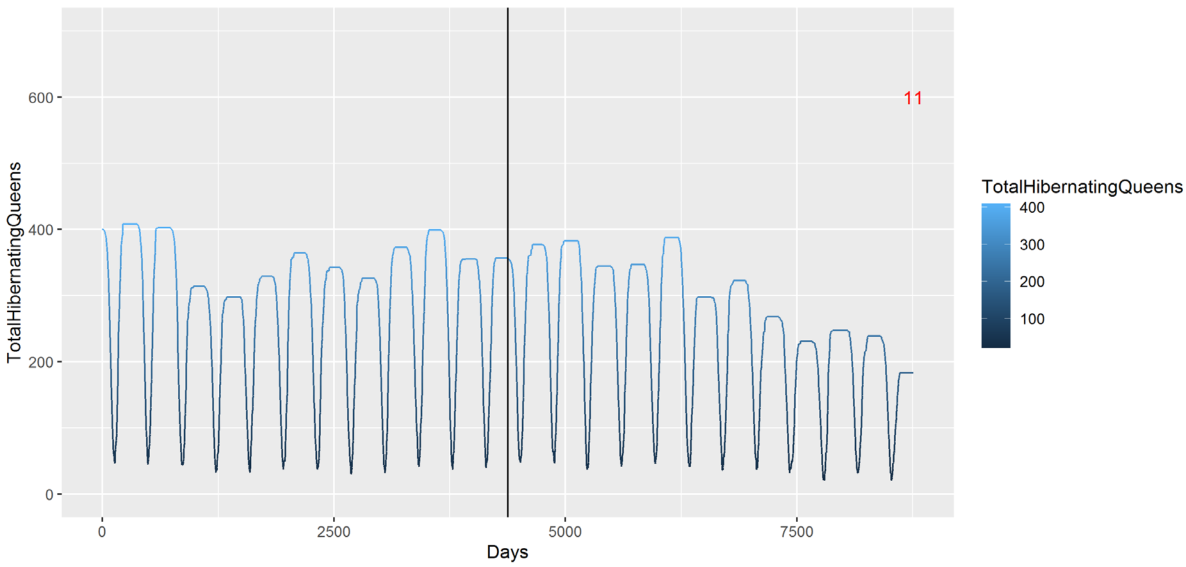
Bombus terrestris stabilisation:

Organic scenario:



Conventional scenario:

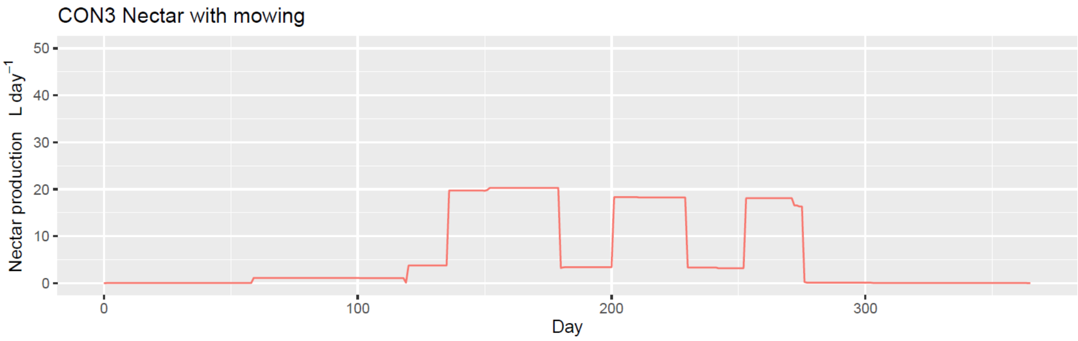
CON4 B.terrestris

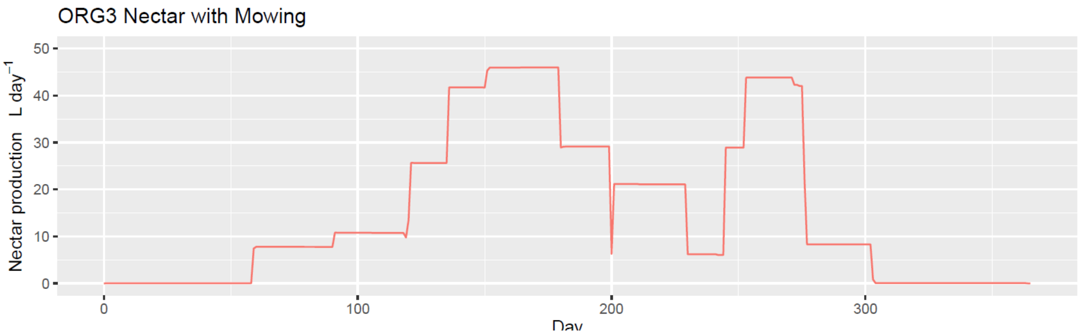


Appendix D.7: Nectar and pollen availability throughout a typical year

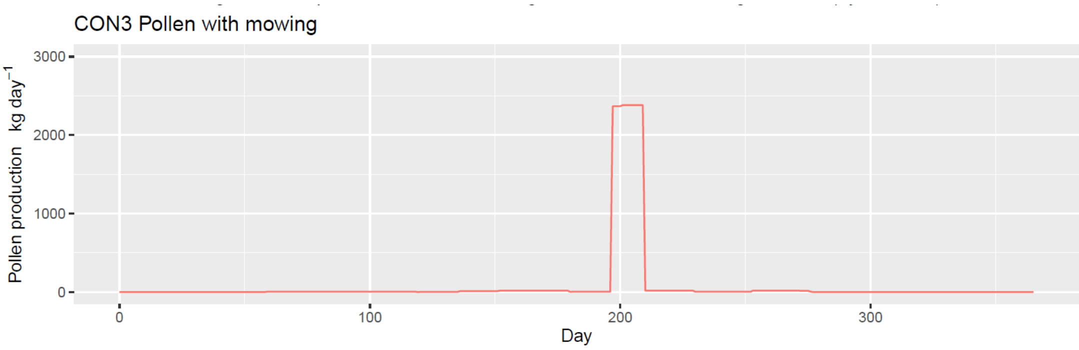
Nectar throughout the year across the study landscape:

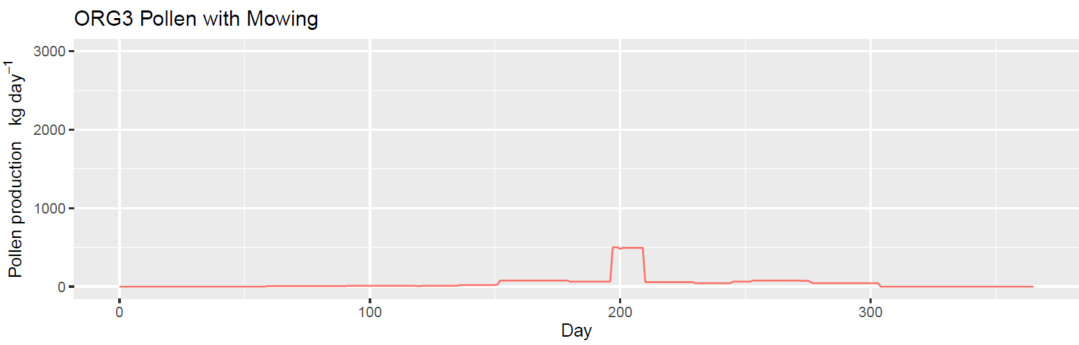
Step changes show where silage mowing and/or organic wholecrop harvest have impacted nectar availability.





Pollen throughout the year across the study landscape:





Appendix E:

Appendix E.1: The cost of nitrate removal at WTW Water Treatment works

The water treatment process:

Figure one provides an overview of the water treatment process at WTW Water Treatment works. Further details can be found in a document titled: Ion Exchange Plant Operation. It is important to note that the ion exchange plant is always running water through one cell. More cells are bought into use and more water is processed through the plant incrementally when final blended water concentrations exceed 38 mg l^{-1}. WTW try to have a buffer of around 25% between actual nitrate concentrations and the drinking water limit of 50 mg l^{-1}).

The costs associated with nitrate removal:

Capital costs:

The ion exchange plant at WTW Water Treatment Works was installed in 2006. WTW were unable to provide any information on how much the plant cost to install. The installation of ion exchange plants are recognised as being expensive (Environment Agency, 2019) with installation costs ranging from £250,000 for a small plant, up to £8million for larger plants (Table 1).

Due to the expense of installation it is considered important to incorporate an estimate of the annualised capital costs into annual treatment cost saving scenarios. Data was therefore collected on the costs of other nitrate treatment plants installed in the UK. Where information is available, data can be combined on the scale of the plant (how much it treats each day) and the cost of installation in order to estimate indicative costs for the WTW Ion Exchange plant. These figures are presented in the table below. It is important to note that costs for other plants can be hard to come by, often cannot be separated from other capital costs (e.g. the building or infrastructure around the plant) or are not accompanied by data on the plant capacity.

Table 1: Cost data from the installation of other ion exchange nitrate removal plants in the UK.

Year	Cost (£'millions)	Amount of water *m3 / day)	Cost per daily abstraction (£/m3)	Water provider
2015	7.5	22,000	340.9090909	Yorkshire Water
2015	5.6	13000	430.7692308	Severn Trent Water
2016	0.25	904	276.5486726	Cholderton Water Company
Mean cost per m3 of capacity:			£349.41	
Other plants installed with incomplete data:				
2012	1.7 - 2.5	?	?	Cambridge Water Company
2018	4.1 - 8	?	?	Southern Water
2018	7	?	?	Anglian Water
?	4	10500 (?)		Wessex Water
?	3.73	?	?	NA cited in Green Alliance Doc
2020	8	?		Wessex Water

FIGURE REMOVED – SHOULD NOT BE SHARED WITHOUT SWW APPROVAL

Figure 1: Diagram of the nitrate treatment program at WTW Water Treatment Works showing incoming boreholes and treatment protocol. Contanstant pumping through plant is conducted at 37 mg $l^{-1}$with additional treatment when nitrate exceeds 38 mg l^{-1}.

Based on a nitrate treatment capacity of 30m^3 per hour for each cell (5 in total) at WTW Ion Exchange plant it is reasonable to estimate that the total daily treatment capacity for the plant is 3,600m^3. Combined with a mean installation cost per m^3 of daily capacity, this would equate to an estimated installation cost of £1,257,876 or around £1.25 million.

An estimate of £1.25 million was therefore selected as the capital investment to install the plant. The capital expenditure was annualised over 20 years with a discount rate of 3.5%.

Juntakut et al., (2020) use the same time period of 20 years in there accounting on nitrate treatment solutions in the US, albeit with a slightly higher discount rate of 5%. **https://www.mdpi.com/2073-4441/12/2/428**

Operational costs:

Operational costs for nitrate removal can be significant with typical ranges for larger plants of around £250,000 to £500,000 per year. The only data found for a smaller plant, operating at a similar scale (903 m3 per day) was for the Cholderton Water Company, which has an annual operation cost of £23,000 based on labour and power.

Information to calculate operational costs were provided by WTW staff who works at the WTW.

Salt:

Salt is required to recharge each cell and enables nitrate to be removed from drinking water. WTW staff initially said that around 60 tonnes of salt is bought in each year at a price of roughly £130 per tonne. WTW staff has subsequently given me access to more accurate information on the actual salt purchases since 2012 (see Table 1). Whilst there is some overlap in stock between years this was fairly small and as such it was assumed that amount purchased, reflected the amount used in that year. A mean of 99.68 tonnes of salt per year was used in annual operation cost calculations.

Table 2: Showing the sum of annual salt orders for WTW Anion Exchange Plant from 2012 to 2019

TABLE REMOVED – NOT TO BE SHARED

Labour:

WTW staff suggested that a rough estimate for their time spent working on the plant was between 1 – 2 days per month. Low (1 day, 8 hours), middle (1.5 days, 12 hours) and high (2 days, 16 hours) have therefore been applied in labour cost scenarios.

No data on staff costs were available from WTW and therefore industry standards were taken from Office for National Statistics Labour Costs per Hour (ILCH) data: https://www.ons.gov.uk/employmentandlabourmarket/peopleinwork/earningsandworkinghours/datasets/indexoflabourcostsperhourilchseasonallyadjusted

A mean labour cost per hour of £25.40 was calculated for 2019 for the Electricity, Gas and Water Supply Industry (including Wages and Salaries, National Insurance Contributions, Employer Pension Contributions, Sickness, Maternity and Paternity Payments and Benefits in Kind). The figure was used in annual operation cost calculations.

Electricity:

Total electricity data for the plant was gathered from site in December 2020. Readings for the number of operation hours (129515 hours, 14.78 years) and the total energy consumption (737262 kwh) were recorded. The two figures combined allow a calculation of the average annual energy usage, 49866 kwh.

No data on electricity rates was explicitly provided by WTW.

Energy rates per kwh were instead taken from the National Statistics dataset on the Non-domestic Energy Prices by Quarter from the Department for Business, Energy and Industrial Strategy: https://www.gov.uk/government/statistical-data-sets/gas-and-electricity-prices-in-the-non-domestic-sector

Water companies are identified as some of the largest electricity consumers (Majid et al., 2020) and therefore prices were selected from the 'Extra Large' business band (>150,000 MWH). The prices shown in the dataset are fully delivered prices, including all elements except VAT. The mean from all quarters for 2019 was £0.114 per kwh and was used in the calculation of annual operation costs.

Spreadsheet on estimates for operational and capital costs:

Table 3 provides a lower, middle and higher estimate for the annual operational costs at WTW Anion Exchange Plant. The range of estimates is on the basis of differences in labour estimates. The middle estimate of £23,438 has been used in the calculations of the benefits under different land management scenarios.

Table 3: Table showing the estimated annual operational costs for WTW anion Exchange plant including electricity, labour and salt

	Low estimate (£)	Central (£)	High estimate (£)	VAT status
Electricity	6821.69	6821.69	6821.69	Incl.
Labour	2438.4	3657.6	4876.8	?
Salt	12958.4	12958.4	12958.4	Not provided
Total	**22,218.49**	**23,437.69**	**24,656.89**	

It is important to note that operational costs do not include the disposal of the waste nitrate rich brine which depending on the disposal pathway can be expensive to deal with. E.g. https://marketplace.wessexwater.co.uk/challenges/ion-exchange-brine-challenge/

The combined capital and operations costs are shown in Table 4. The data shows an annual total cost of £111,389.03 per year.

Table 4: Showing total annual costs including both operational and annualised capital cost estimates for WTW Ion Exchange Plant

	Cost	Comment
Capital Cost (£)	£1,250,000	Estimate
Annualised Capital Cost (£)	£87,951.35	r = 3.5%, T = 20
Operational Cost (£)	£23,437.69	Mid-range
Total Cost per year (£)	£111,389.03	Combined

Appendix E.2: Connecting nitrate leaching to aquifer nitrate concentrations – Otter sandstone and Budleigh Salterton Pebble Bed Aquifer

Overview of nitrate leaching and potential benefits generated by the conversion to organic agriculture by Clinton Devon Estate

Aims:

- To link standard nitrogen inputs from farmland with current concentrations of nitrate in the aquifer
- To establish the potential impact that conversion to organic farming in the Lower Otter Valley could have on future nitrate concentrations in the aquifer
- To quantify the value of the benefits that might arise from past conversion to organic agriculture

Key principles/caveats:

- The study requires assumptions to be made about the extent of the aquifer, the annual recharge, nitrate leaching inputs and the length of time it takes for nitrate to reach the aquifer. These assumptions are backed up by relevant literature.
- The study acknowledges that the groundwater aquifer is a complex system with significant spatial, temporal and depth variations found in nitrate concentrations. However, the study has had to simplify the system to a single unit that responds as one across the area
- The study uses mean nitrate inputs from only three conventional and three organic fields over only two years. These fields represent a variety of typical crops and grass found in the area but the data set is small and highly variable between sites. Even so, the mean estimate taken is in line with other academic studies and previous local monitoring

Aquifer 'catchment':

- There was no unanimous agreement with South West Water on what represented the 'aquifer catchment', i.e. the area across which any leaching nitrate will reach the aquifer. Initial investigations looked at the source protection zones around the borehole but on advice from WTW the Lower Otter catchment was identified as a more reasonable extent.

- It is also noted that further north the Otter Sandstone becomes more fractured and there are clearer boundaries between aquifer units (Allen et al., 1997)
- Parts of the bedrock in the Lower Otter catchment are Mercia or Aylesbeare Mudstone or Upper Greensand and were not considered to play a role in contributing to the aquifer.
- The area assumed to be the aquifer catchment was therefore taken as the area defined as Otter Sandstone or Budleigh Salterton Pebble Beds formations that lie within in the Lower Otter surface water catchment (Figure 1). It is recognised that combined these formations essentially act as one aquifer and share a common water table (Allen et al., 1997)
- This area also appears to be shown in the Lower Otter groundwater model outputs (provided by WTW) as receiving the highest density of rainfall recharge (the main pathway for nitrate entering the aquifer)
- The total area is identified as 81947054m2 or 8194.71ha (including some minor overlaps outside the surface water catchment) and is shown as the Otter Sandstone layer in Figure 1A.

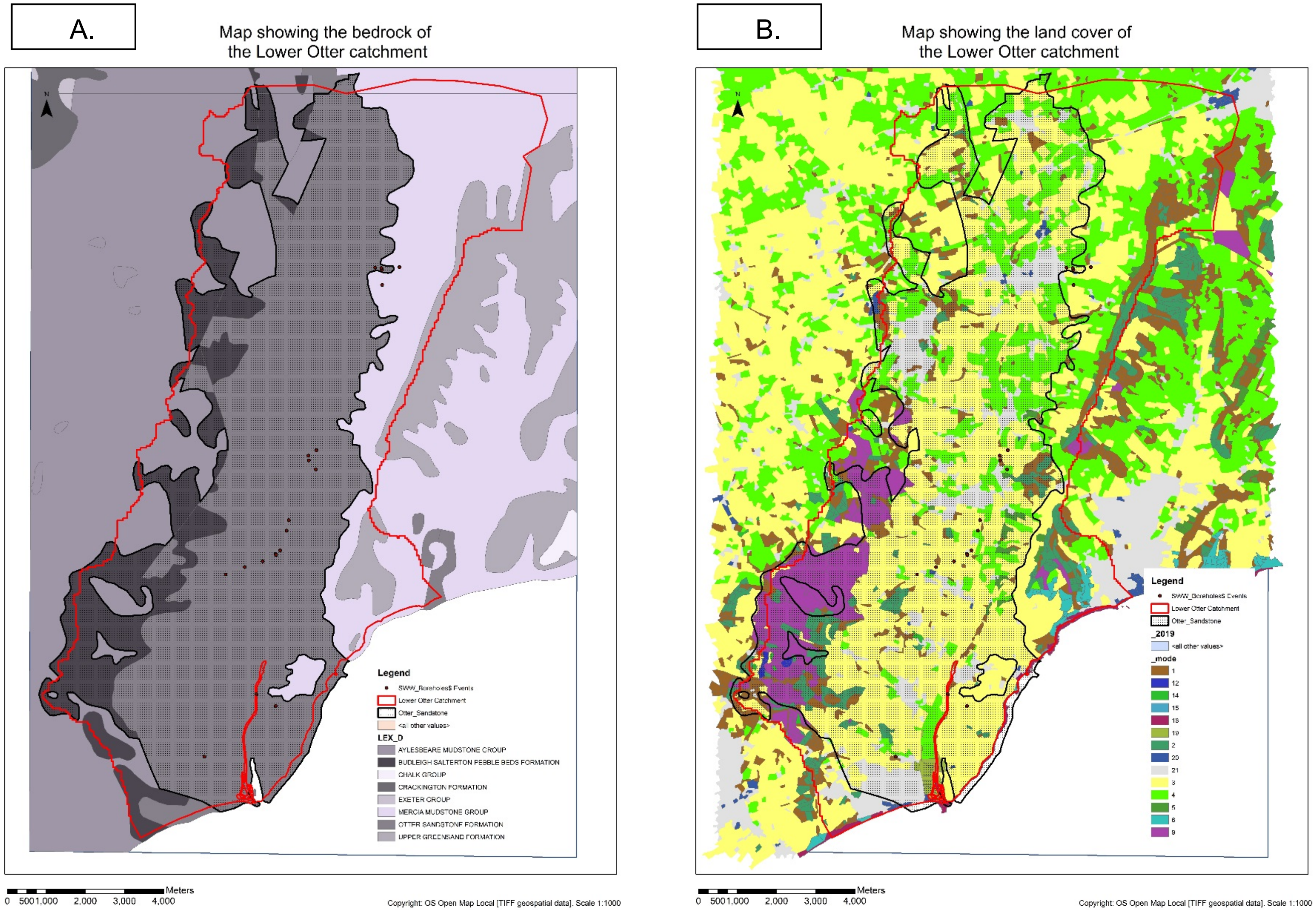


Figure 1: Showing spatial data used for the assessment of the aquifer 'catchment' and in determining land area that contributes to nitrate leaching.A. Shows the distribution of the otter sandstone and Budleigh Salterton pebble beds formations in the Lower Otter Valley used as the aquifer 'catchment'. B. Shows the 2019 land cover map (CED, 2020) used in the assessment of arable and grassland areas contributing leached nitrate.

Aquifer ‘capacity’:

- No definitive data is available on the ‘capacity’ of the aquifer, i.e. how much it is likely to be storing at any one time
- Bearcock and Smedley (2012) and Allen et al., (1997) cite pers. comm., information that suggests that the total allowable abstraction of 25,000m3 is estimated to be half of the rainfall recharge in the aquifer. This was contested by WTW during discussions however and is likely to be coarse as it was prior to more detailed groundwater modelling
- However, it is agreed that the main recharge to the Otter Sandstone is through rainfall recharge (Perl et al., 2004). See groundwater model outputs below, Figure 2.
- The MORECS effective rainfall for the area is identified as 427mm per annum (Perl et al., 2004), which is very much in line with drainage calculations made as part of my field work (mean 443.4mm)
- The Lower Otter groundwater model outputs support the information above and suggests that Total Available Water is aligned closely to rainfall recharge at around 400mm per year (see figures below)
- The aquifer capacity was therefore estimated by multiplying the aquifer ‘catchment’ by the annual rainfall recharge and came to 34,991,392m^3

Figure 2: DO NOT REPODUCE THIS FIGURE _ FOR INTERNAL GUIDANCE ONLY. PROVIDED BY WTW.

FIGURE REMOVED

Aquifer ‘response time’ to nitrate leaching:

- An important assumption for this study is that reduced nitrate leaching triggered by Clinton Devon Estates conversion of around 950ha to organic agriculture in the Lower Otter Valley in 2007 is yet to have had its full impact on the concentrations seen in the aquifer. This assumption appears valid in the context of the literature and in calculations predicting nitrate concentrations in the aquifer.
- The assumption requires an understanding of how long it takes for nitrate loading to reach the aquifer
- No standard response time for nitrate reaching the aquifer in the Lower Otter was provided/available and therefore estimates have to be made from the literature
- It is widely recognised that aquifers take time to respond to changes in nitrate leaching (Wang et al., 2012). The response time varies depending on a number of factors including bedrock characteristics, depth to the bedrock, effective rainfall rate and local fracturing that can facilitate rapid transport. It varies both between aquifers and spatially across the same aquifer.
- It is therefore reasonable to expect that the levels of nitrate we see in the aquifer today are reflective of inputs in the past….but how long ago?
- Wang et al., (2012) identify that most aquifers received nitrate loads within 20 years of surface leaching. They suggest that the Permo-triassic sandstones respond “fairly rapidly” to changes and that the majority of these types of aquifers had already achieved peak nitrate by 2012. Their national mapping of nitrate travel times indicates that response times in the Lower Otter Valley could be between ranges 1 – 5 years, 6 – 10 years, 11 – 20 years (though the map is hard to zoom in on) (Wang et al., 2012)
- Further work from Wang et al., (2016) suggest that the ‘turning’ point for the Permo-triassic sandstones of the South West was 2005. Coupled with their previous work which estimates a peak nitrate inputs ending in 1990 (Wang et al., 2012), this could suggest around about a 15 year response time.
- The key assumption here is that it is likely that it takes time to respond to nitrate changes, potentially in the range of 10 – 15 years (plus) and

therefore there is a good chance that the conversion to organic agriculture in 2007 across a large part of the Lower Otter Valley has not yet had a significant effect on the aquifer nitrate concentrations

Estimating nitrate inputs (pre-organic conversion):

- Field work conducted by this study identified mean nitrogen losses on three conventional fields and over a fairly dry and then very wet winter of 52.64 kg N ha^{-1}
- This estimate is in line with local data on leaching collected by Yog Watkins (Westcountry Rivers Trust) in the early stages of his monitoring of nitrate leaching (see 2010-11 and 2011-12). It is recognised that the mean leaching levels are considerably lower from 2012-13 and it is thought that this is due to advice provided by Yog to the farmers participating on how to reduce losses. It is for this reason that the data has not been used as a mean of leaching across the catchment

Table 1: Yog Watkins data on nitrogen losses by year (a minus represents a loss from the field)

Row Labels	Count of N Loss with crop est.	Mean N loss (kg N ha^{-1})	Min N loss (kg N ha^{-1})	Max N loss (kg N ha^{-1})
2010-11	9	-70.93	27.1	-234.1
2011-12	9	-42.32	22.6	-115.8
2012-13	9	-22.38	-10	-38.9
2013-14	8	-19.96	14.4	-81.8
2014-15	9	-12.98	5.9	-37.3
2015-16	8	-24.16	12	-88.1
2016-17	10	-5.75	34	-66.9

- The figure of 52.64 kg N ha^{-1} is also in line with estimated inputs from Wang et al., (2012) on national nitrate losses in the mid to late 2000s, although they predicted that nitrate losses by 2020 would have reduced to around 40 kg N ha^{-1}.
- A figure of 52.64 kg N ha^{-1} was used as the inputs per ha of agricultural land within the area covered by the Otter Sandstone and Budleigh Salterton Pebble Beds formations. It is recognised that this is a significant simplification given the variability in leaching from different land management, under different farms and in different seasons. However,

such a simplification is in line with other studies (Wang et al., 2012) and, using the data available, allows a fairly rapid overview to be made about the likely accuracy of the figure in light of the actual mean aquifer concentration

- Agricultural land was derived from the Land Cover Map 2019 (CEH, 2019) (Figure 1B). Agricultural land was taken as the total of arable and horticultural land, improved grassland and calcareous and neutral grassland (although the latter only made up around 50ha)
- All other land uses, including heathland, forestry, urban and suburban land were not included in the area used to estimate nitrogen inputs (I.e. they were assumed to have a nitrogen input of 0 kg N ha^{-1}). It is recognised that this is not necessarily accurate as such land uses do leach deposited nitrogen but these losses are typically much lower than on agricultural land (Herrmann et al., 2005)

<u>Putting it all together:</u>

Does our leaching estimate, assuming blanket conventional land use across the Lower Otter catchment align with current aquifer concentrations?

- Table 1 shows that the estimate of aquifer nitrate concentration (37.27 mg NO3 l^{-1}) made using a 52.64 kg N ha^{-1} input function and the assumed aquifer 'catchment' and aquifer 'capacity' is reassuringly close to the actual mean nitrate concentrations from all boreholes sampled in 2019 (37.16 mg NO3 l^{-1})
- It seems reasonable therefore to suggest that, if the aquifer capacity assumption holds, leaching in this order of magnitude might be responsible for the concentration of nitrate in the aquifer
- It also suggests that impacts of a significantly reduced nitrate input (from organic conversion) are unlikely to have had a significant impact on the aquifer concentration yet
- NOTE: NOT ALL BOREHOLES ARE MONITORED EACH YEAR or consistently throughout the year (so some might be sampled more than others in one year or not at all). There is a good chance that without this systematic monitoring it is hard to get a good picture of comparable

concentrations year on year. Reassuringly however in 2019 all 14 WTW boreholes were sampled at least once.

Table 2: Showing the key figures used in the estimation of current nitrate concentration in the aquifer and compared with actual mean data from WTW on the pumped water from boreholes

Component	Value	Units	Calculation method
Aquifer catchment	81,947,054	m2	Otter Sandstone and Budleigh PBs area
Annual rainfall recharge	0.427	m	MORECS data from Perl et al., (2004)
Aquifer 'capacity'	34,991,392	m3	Aquifer catchment * rainfall recharge
Agricultural area of aquifer (AA)	5,605.55	ha	Agricultural area (AA) from CEH LCM (2019)
Annual nitrogen input per ha	52.64	Kg N ha-1	Field work data (FWD)
Annual nitrogen input over AA	295,076.41	Kg N	AA * nitrogen input per ha
Expected aquifer concentration	0.008433	kg N m3-1	Total N input / aquifer capacity
Expected aquifer concentration	37.27	mg NO3 $^{l-1}$	Conversion from kg N m3 to mg NO3 l-1
Actual mean concentration from aquifer borehole samples	37.16	mg NO3 $^{l-1}$	Mean of all WTW borehole samples in 2019

Given the reassuringly close predicted vs actual figures for nitrate concentrations in the aquifer it is possible to use the same aquifer characteristic assumptions to predict what impact reducing nitrate losses across this area could have on nitrate concentrations

What impact will the conversion of the organic Home Farm have on nitrate leaching and aquifer concentrations?

- Table 2 shows the calculations and reductions generated by the reduced leaching from the organic Home Farm and Dalditch Farm
- The calculations use a mean organic nitrate loss mean of 19.85 kg N ha^{-1} which was taken from field work conducted in this study across three fields over a dry and then relatively wet winter season. The estimate is in line with Yog's study for an organic field from 2010-11 to 2017-18 of 19.61 kg N ha^{-1}
- The area of organic land is derived from Clinton Devon Estate maps and is clipped to only calculate the areas within agricultural use and within the aquifer 'catchment' area
- The calculation assumes that reduced loading of nitrate generated from organic land management has NOT yet manifested itself in significant changes in the aquifer

- The shift from conventional to organic agriculture across 896ha (15.97% of the defined agricultural area) suggests there could be a decrease in the mean aquifer concentration of 3.71 mg NO3 l^{-1}.
- The drop from 37.27 mg NO£ l^{-1} under the predicted conventional only scenario to 33.56 mg NO3 l^{-1} would reduce the mean aquifer concentration below the threshold level for treatment (37 mg NO3 l^{-1}).

Table 3: Showing the projected concentration shift in the aquifer nitrate concentrations on the basis of the shift to organic land management by Clinton Devon Estate

Scenario two (organic conversion):		Units
Conventional Nitrogen input	52.64	Kg N ha-1
Organic nitrogen input	19.85	Kg N ha-1
Organic agricultural area	895.32	ha
Remaining conventional area	4,710.24	ha
Conventional input over conventional area	247,947.00	Kg N
Organic input over the organic area	17,772.01	Kg N
Total input from both	265,719.01	Kg N
Expected aquifer concentrations:		
Expected concentration in the aquifer	0.00759384	kg N $m3^{-1}$
Expected concentration in the aquifer (converted to mg NO3 l^{-1})	33.56	mg NO3 l^{-1}
Conventional only scenario:	37.27	mg NO3 l^{-1}
Projected change in aquifer nitrate:		
Change in nitrate concentration	3.71	mg NO3 l^{-1}
Predicted mean aquifer concentration:	33.56	mg NO3 l^{-1}

Does the reduction in nitrate concentrations in the aquifer generate a financial benefit for WTW/society?

Changes in the aquifer nitrate concentration have the capacity to alter the costs of treatment in different ways. Minor changes in concentration that reduce the need for further treatment (above baseline) will show in gradually lower salt, power and potentially labour costs. More significant changes, i.e. shifting concentrations consistently below the baseline operation present opportunities for either running at baseline only (for cautions sake) or decommissioning the plant all together.

The predicted long term impact of reduced nitrate leaching from organic land at Clinton Devon Estate is relatively significant with a mean aquifer nitrate concentration reduction from above the treatment threshold (37 mg l^{-1}) to below it (estimated at 33.46 mg NO3 l^{-1}). Two possible scenarios have therefore been explored relating to the financial benefits of this:

A. The nitrate plant continues to operate at baseline (I.e. one cell is always running) with the expectation that it will rarely exceed baseline operation. The plant is still available should the need arise to increase treatment to tackle any occasional nitrate spikes.

B. The nitrate plant can be decommissioned and the parts re-used elsewhere. Blending becomes the only way to tackle any possible nitrate spikes (as conducted prior to plant installation in 2006)

<u>The potential cost savings generated under scenario A:</u>

Details on calculating the costs of nitrate treatment at the WTW can be seen in a separate document, titled: Overview of Treatment Cost Calculations at WTW. Table 3 provides an overview of the workings to estimate the reduction from current average annual operation to baseline operation. As it was only possible to predict the impact of running at baseline on salt use the percentage change in this was used to calculate changes in the associated labour and electricity costs. I.e. if the plant was running at baseline it would require around 50% less salt, it has therefore been assumed that this equates to 50% less electricity and 50% less labour required to run the plant.

The maximum reduction that could be generate by operating only at baseline level (using a mid-range estimate of labour costs) is estimated to be £11,681.58.

Table 4: Table showing workings to estimate the reduced costs associated with running the nitrate anion exchange plant throughout the whole year at baseline level (i.e. with one cell only).

Requirements for baseline operation:	Value	Units
Salt:		
Days operating per year	365	days
Hours for recharge for one cell	18	hrs
Recharge salt ammount	0.103	tonnes
Number of recharges of one cell in a year	486.67	recharges
Salt required for this amount of recharge	50.12667	tonnes
Cost per tonne of salt	130	£
Total salt costs estimated if running only one cell	6516.47	£
Percentage differences to actual salt requirements:		
Acutal annual salt usage (mean)	99.68	tonnes
Actual annual salt cost	12958.4	£
% differences between baseline and actual	50.29	%
If we apply the same percentage reduction to electricity and labour costs:		
Labour (mid) x 0.50	1828.8	£
Electricity x 0.50	3410.844	£
Revised changes if plant ONLY operates at capacity:		
Labour (mid)	1828.8	£
Electricity	3410.844	£
Salt	6516.47	£
Total	11756.11	£
Savings from current run costs:		
Current running cost (mid)	23437.69	£
Saving from current (mid)	**11,681.58**	£

The potential cost savings generated by scenario B:

The decommissioning of the nitrate treatment plant would result in annual savings on all operational costs. There is also the chance that the initial capital investment is not entirely sunk. That is, the components of the plant are not worthless. They might be able to be repurposed or sold on to another works. In this scenario the annualised capital costs are therefore included in the calculation of annual savings. The costs have been annualised over twenty years with a discount rate of 3.5%.

Table 5: Showing the estimated combined annual costs for annualised capital costs and operational costs. Capital costs are annualised using a discount rate of 3.5% and a time frame of 20 years.

	Cost	Comment
Capital Cost (£)	£1,250,000	Estimate
Annualised Capital Cost (£)	£87,951.35	r = 3.5%, T = 20
Operational Cost (£)	£23,437.69	Mid-range
Total Cost per year (£)	£111,389.03	Combined

Water treatment cost savings and assumption caveats:

Our study found that the conversion of 895ha of land to organic agriculture in the Lower Otter Valley in 2007 could have the capacity to reduce mean nitrate leaching below treatment thresholds and, depending on WTW's response, could generate considerable nitrate treatment savings (up to £111,389 per year). Given the complexity of the groundwater system, it was reassuring to find that the mean nitrate leached (52.64 kg N ha^{-1}) across our conventional fields was such an effective predictor of current aquifer nitrate concentrations (based on the assumptions made). Whilst based on limited replication this leaching estimate is in-line with simple nitrate input functions used in other studies. Wang et al., (2012) for example applied a basic variable nitrate input function from 1925 – 2050 (ranging between 25 kg N ha^{-1} – 70 kg N ha^{-1}) that changed in response to increased fertiliser use and regulations, to predict the arrival of peak nitrate concentrations across all UK aquifers. Harris *et al.*, (2006) use a central estimate of 40 – 50 kg N ha^{-1} for conventional arable practices in evaluating the benefits of different land use conversion scenarios.

We recognise that our assumptions simplify the functioning of a complex aquifer, with variable unsaturated zone depths, fracturing, saturated zone thickness and aquifer porosity, for example, all having an impact on nitrate transport and residence in the aquifer. The simple but justifiable assumptions, have however allowed us to forecast potential nitrate treatment scenarios and the costs associated with these. These costs of treatment are realistic and reflect real outcomes as opposed to hypothetical replacement cost scenarios. Other studies have also used simplified nitrate input functions and relatively basic information about aquifer functioning (e.g. unsaturated zone thickness and travel time through the unsaturated zone) in predicting peak nitrate arrival and aquifer concentrations across the UK (Wang *et al.*, 2011, 2013, 2016). Whilst a number of studies have either modelled or measured nitrate leaching under a range of different land management scenarios (including organic or conventional) no know studies have investigated how a specific local land use conversion scenario could impact on nitrate leaching and the costs of nitrate treatment in drinking water.

Total costs of nitrate pollution have been reported with Sutton *et al.*, (2011) estimating the social damage cost of nitrate losses to water in the EU at €15 – 70 billion yr^{-1} and Pretty *et al.*, (2000) reporting the annual UK cost of treating nitrate in drinking water at £20.1 m yr^{-1}, estimated to increase to £199 m yr^{-1} over the next 20 years. However, these calculations are not spatially specific or related to per unit area or marginal changes in nitrate treatment costs (Harris *et al.*, 2006). Harris et al., (2006) use a method of quantifying marginal costs of a loss of attenuation capacity of N in soils based on policy costs associated with WFD. They calculate costs associated with deviating from best practices or converting extensive grassland to arable between £1.2 - £253 ha^{-1} yr^{-1} but they do not place values on the potential benefits derived from scenarios that reduce nitrate leaching. They conclude that there is likely to be significant spatial variability in the costs of nitrate pollution and that there is a need for research on water company cost savings from the implementation of nitrate attenuation measures (Harris *et al.*, 2006). We present such research here and whilst we recognise that there are improvements that could be made to the understanding of the groundwater system, we feel that it provides a methodological base to be built upon on in the ES evaluation of other land use change scenarios.

Appendix E.3: Data outputs for Yield and Carbon plotting using all soil and yield data points (as used in Chapter 4)

Carbon plots:

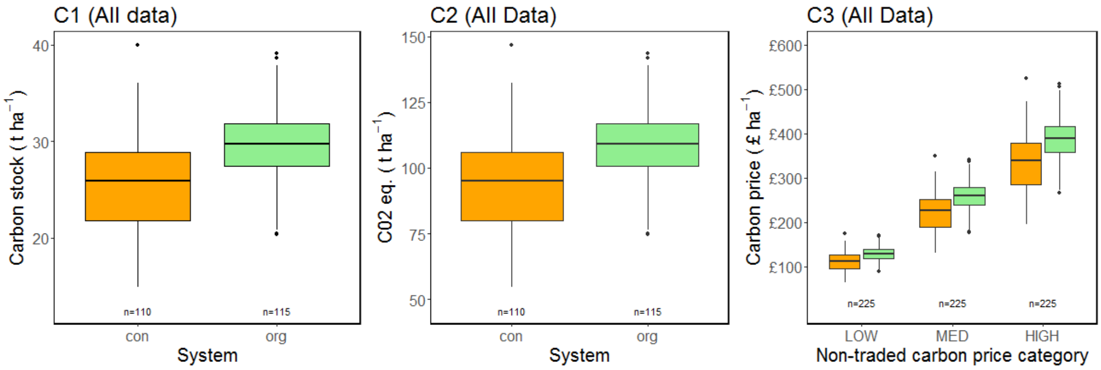


Figure 4: Showing carbon stocks (C1), CO2 eq. (C2) and carbon price (C3) differences between organic (green) and conventional (orange) across all field sites (as used in Chapter 4). It shows the same pattern as presented in the sub-set data showing in Chapter 7 with organic fields having significantly higher carbon storage than conventional fields.

Yield plots:

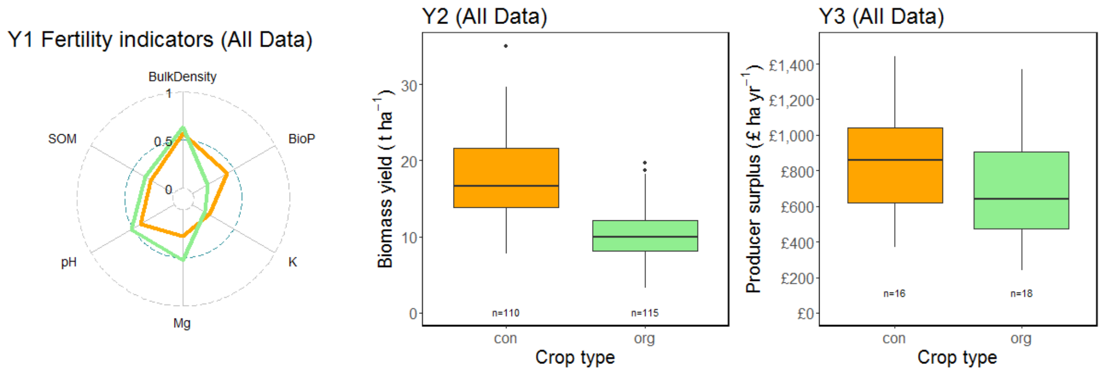


Figure 5: Showing fertility indicators (Y1), biomass yield (Y2) and producer surplus (Y3) differences between organic (green) and conventional (orange) across all field sites (as used in Chapter 4). The data shows the same pattern as presented in the sub-set data shown in Chapter 7 with limited significant differences in fertility indicators, organic having significantly lower biomass yield and no significant differences observed in producer surplus.

Appendix E.4: Detailed discussion on the specific challenges encountered when measuring the four pathways traced through the NC approach

Carbon storage and climate regulation:

In comparison to other ES flow pathways, measurements of carbon storage and the value of climate regulation are relatively straightforward to understand, a fact that has been noted by other studies (Harris *et al.*, 2006; Keeler *et al.*, 2012; Duncan, Thompson and Pettorelli, 2015). The only empirical data requirements are NC conditions; soil carbon, the density of the soil (bulk density) and the depth of the sample (all used to calculate carbon stocks). Acknowledgement of the importance of data on soil organic matter (SOM) and soil carbon is growing amongst land managers (Farmers Weekly, 2021) and simple conversion factors (e.g. 0.55, 0.58 or 0.52; refs) can be used to calculate soil organic carbon (SOC) from SOM. Analysis for SOM can be incorporated into other routine agricultural soil testing for P and K and adds relatively little to the cost (£7 - £10 per sample or field). In principle therefore SOC can be calculated easily and it is realistic that land managers could make appraisals of SOC (%) independently over time. It is important to note that some researchers disagree over the optimum depth for measuring changes in carbon storage (Baveye, Baveye and Gowdy, 2016) and some suggest the need to measure both labile and recalcitrant fractions of carbon in carbon storage calculations; distinguishing between those fractions that are more stable within the soil and those that more easily decompose (Yeluripati *et al.*, 2018) (more refs). Whilst this is not generally the norm in other ES studies it highlights that even apparently simple measurement of NC condition linked to EF and ES value are open to debate. Furthermore, measuring bulk density, whilst not necessarily requiring a huge amount of equipment or expertise, are particularly time consuming (especially on the stony soils at CDE). When it is even possible, sending soils off for BD analysis is expensive and for this reason, outside of academic studies, it is rarely measured by land managers. Available BD standards (e.g. for soil series or type) can be used in lieu of actual field data or modelled using other data but this does simplify the evident spatial and temporal variability of properties (Baveye, Baveye and Gowdy, 2016). Despite these complications the benefit of collecting data on carbon stocks is that once it has been quantified it can be easily converted to CO_2eq and then valued using non-traded carbon prices from the BEIS (2019) (suggested in the evaluation of schemes from sectors not covered by the EU Emissions Trading Scheme)[38]. These data can be compared and contrasted between different land management scenarios to evaluate change in the value of the ES climate regulation.

Crop yields and producer surplus:

In contrast to carbon stocks, understanding crop production requires information on EF and the additional calculations of the sellable proportion of the crop (e.g. converting to grain if previously measured as biomass)[39]. Measurements of NC condition were in fact not at all useful in determining crop biomass and whilst these conditions undoubtable impact crop yield these type of data are not necessary in determining crop production services. It is often projected that moving from measurement of yield to an understanding of ES value is straightforward and existing UK studies mostly use data on crop prices and variable costs of production from The John Nix Pocket Book (2019) (Bateman *et al.*, 2013; Fezzi *et al.*, 2014; Faccioli *et al.*, 2020). The same starting principles were applied here but what became immediately apparent was that data on farm gross margin presented in Nix (2019) only include costs on fertiliser, sprays and seed, assuming all other costs such as cultivation, drilling and harvest are fixed costs. This is in contrast to the management systems applied on many farms (in this study all case-study sites used farm contractors for crop establishment, management and harvest); In these situations costs such as preparing the field, drilling the crop, applying fertiliser and sprays and harvesting the crop are only incurred by the farmer if they proceed with the production of the crop. In effect they are variable costs and they have been included in this study in the calculation of producer surplus. It was identified that using only basic gross margins as calculated in Nix (2019) or in the organic equivalent, the Organic Research

[38] It is worth noting that whilst carbon storage in one part of the world is generally considered to deliver the same benefits in another part of the world (Keeler *et al.*, 2012) there is no consistent global method in the valuation of carbon storage (Bartkowski *et al.*, 2020). Different methods used such as marginal abatement costs methods (as used by the BEIS) and the social cost of carbon (Tol, 2019) can result in the use of different ES values for the same scale of EF (Bartkowski *et al.*, 2020).

[39] Conversion to grain is not necessary where data on grain yield is directly measured in the field/lab. This is however time consuming to do by hand. Technology exists to calculate real time grain yields during combining but this is not application where cereal crops are harvested for wholecrop silage feed.

Centre handbook (2017), inflated the value of conventional crops. These crops had higher yields and higher returns but the farm gross margin failed to account for the typically higher costs of production (e.g. labour and machine costs of applying the additional sprays and fertilisers). The producer surplus method has the advantage that the ES value of crop production is also more sensitive to changes in NC condition. For example, improved NC condition in the form of higher organic matter and biological activity could facilitate improved plant available nitrogen, reducing the rate of nitrogen fertiliser and the contract hours in spreading it, decreasing input costs and improving the producer surplus as a consequence.

It is worth highlighting here that despite the relevance of crop yield, crop production costs or crop returns to the farm business it was observed that these were not data already collected at the crop or field scale by the study farms. Whilst this is no doubt done by some enterprises it was not conducted by most other tenant farms on the estate. Gathering this data in the future will be critical if land managers want to better understand the influence of NC condition on crop performance and producer surplus and if they wish to understand the private benefits of a change in land management from a baseline scenario (i.e. changes in crop producer surplus).

Nutrient retention and the delivery of clean drinking water:

The information requirements for measuring nutrient retention and clean drinking water provision are a significant step up from measuring carbon and crop provision ES. Indeed, whilst measurements of carbon/climate regulation and yield/crop production are frequent within ES and NC studies (Greiner *et al.*, 2017) there are very few studies linking water quality contamination with ES delivery (Keeler *et al.*, 2012). This is the first study of which we are aware, to link nitrate leaching at the individual aquifer scale to drinking water treatment costs and explore change under different scenarios. It is widely recognised that these links are poorly understood, both in terms of the biophysical data linking leaching with aquifer nitrate concentration and the economics data showing how changes in nitrate levels will effect drinking water treatment costs (Keeler *et al.*, 2012). Chapter 7 Figure 7.8 shows the additional data required to go from an understanding of nitrate leaching to an understanding of aquifer concentration, incorporating data on the extent of the groundwater aquifer (based on geological data), land cover (based on CEH Land Cover Map) and rainfall recharge (based on literature sources). It is important to add here that this represents a very simplified model compared to other groundwater models (e.g. (Wang *et al.*, 2013)) and does not consider the complexities of more detailed aquifer properties or the spatial and temporal variability of nitrate levels across the aquifer area. Whilst the model works in so much as the input data aligns well with predicting the aquifer concentrations, increasing confidence in model outputs would require further comprehensive work with groundwater hydrologists and water quality specialists. Furthermore, understanding of the links to the aquifer nitrate concentration are only part of the picture and determining the consequences of changes in nitrate levels on the cost of nitrate treatment required extensive engagement with the local water company. It required a detailed understanding of water treatment protocols and treatment thresholds to understand how changes in levels would influence changes in costs. Exact data on the costs of treatment were not collected by the water company and these therefore had to be calculated based on material order receipts, information on the average annual power usage and details of the staff time spent operating the nitrate anion exchange plant at the water works. The key point here is that building an understanding of the service flows of clean drinking water are resource intensive, requiring input and collaboration with other stakeholders and spanning a range of expertise.

Pollinator stocks and pollination services:

Pollination services, which are frequently identified as being important in the delivery of ES, could not be valued in this study. This is in common with other studies who have eluded to or even measured pollination functions but have not valued the ES (e.g. Hardman *et al.*, 2016). Despite being able to model bumblebee population stocks under the two system scenarios, it became immediately apparent when producing farm and wider estate crop maps that very few, if any commercial crops are grown that depend upon insect pollination. Pollinator stocks therefore offer no/limited economic value to crop production in the local area. This issue has been noted by other researchers (Holland, 2017) and is likely to be a common problem across large parts of the UK, with the agricultural land area dominated by a combination of permanent grassland and cereal crops, 58% and 18%, respectively (Defra, 2020b). It presents a real challenge for valuing pollinators in these landscapes and recognising their importance in land management decision-making use of the NC approach. It can be recognised that their presence has an important opportunity value, that is if they were to be eliminated it could restrict the growing of future insect pollinated crops. They have also been recognised as being important in pollinating wild plants such as hedgerow fruiting plants which are critical in supporting farmland birds (Jacobs *et al.*, 2009). These wild plants and farmland birds, along with observations of the bumblebees, butterflies and more, are all likely to be enjoyed by a large number of people who access farmland. Valuing public enjoyment of wildlife and biodiversity, however, is a significant stumbling block for many studies applying the NC approach and remains a fundamental restriction in its holistic application (Faccioli *et al.*, 2020). A recent study looking at public support for pollinators does provide information on how individuals value pollinators, based on willingness to pay for theoretical bee protection but it does not transfer well to other case-studies considering land management decisions (Mwebaze *et al.*, 2018). Ideally information would be available that links bee or wild plant or farmland bird abundance or diversity with individuals' values (e.g. a value per bee, plant, bird), facilitating an estimation of the change in ES value as a response to change in the stocks of pollinators. This is perhaps an over simplistic outlook on valuing enjoyment of wildlife but such data would be extremely useful in connecting current wildlife monitoring or modelling with ES value.

It is worth noting here that, even in landscapes dominated by insect pollinated crops, measuring pollination services that link bee visitation with actual crop production is time consuming and requires expertise (e.g. see Knapp and Osborne, (2017)). Whilst BEE-STEWARD model outputs can provide both pollinator stock data and potential information that can be applied to support economic valuation of pollinator services its application is resource intensive. In principle the model can be run on the default semi-natural habitat data (flower data for habitats like hedgerows and woodland) and combined with basic crop data without the need for empirical data collection. Only map inputs are required. However, in the context of different land management assessments such as comparing organic and conventional crops it currently has insufficient background data. It was therefore necessary to collect data on the weedy flower rich organic arable fields and the species poor conventional arable fields as well as the grass-clover leys in order to offer a reasonable representation of the study farm. Applying the model also required input from the model development staff and experimental design required expertise from bumblebee specialists. In the context of pollinators, therefore, the information required is resource intensive to collect, is only partially available in some contexts (such as for our case-study) does not currently exist.