Quantifying and mitigating agricultural emissions in England and Wales in line with net zero ambitions

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Intellectual Property Rights and Publication Statement

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Abstract

With 25 years for the UK Government to meet the ‘net zero greenhouse gas (GHG) emissions by 2050’ target it set out in 2019, the pressure is increasing to find solutions to mitigate GHG emissions across sectors. Agriculture contributes 11% of UK GHG emissions, so this thesis aimed to further understanding around how the agriculture sector in England and Wales can reduce its GHG emissions to align with net zero targets using a combination of Life Cycle Assessment and scenario analyses of farm-level (University of Leeds farm) and regional-level GHG reduction and carbon removal practices.

Livestock feed, the application of manufactured fertilisers and manure management were found to be the main GHG emission sources for the University’s mixed farm. At farm-level, nitrification inhibitors applied with nitrogen fertiliser to arable crops and acidification of pig slurry were found to reduce the farm’s total emissions by 13%. At regional-level, increasing the uptake of practices including more efficient nitrogen fertiliser use on arable crops, introducing legumes into arable rotations, incorporating the methane feed inhibitor 3-NOP into cattle diets, and acidification of pig and dairy slurry, can reduce national-level GHG emissions each year to begin aligning with net zero targets.

However, key challenges around the quality of data available to perform these scenario analyses and the technical potential of these practices, such as practice additivity, economic and social implications of practice changes, was raised throughout this thesis. These factors have implications for farmers, food and drink businesses, and policymakers around the feasibility of a net zero farm, let alone a net zero agriculture sector. This thesis therefore highlights the need for higher quality farm data and better consensus on the appropriate practices to model but provides evidence supporting several GHG reduction opportunities and Government-aligned afforestation targets.

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Abbreviations

3-NOP	**3-nitrooxypropanol**
AFOLU	**Agriculture, Forestry and Other Land Use**
APHA	**Animal and Plant Health Agency**
AR6	**6th Assessment Report**
BECCS	**Bioenergy with Carbon Capture and Storage**
CAP	**Common Agricultural Policy**
CFC	**Chlorofluorocarbons**
CO_2	**Carbon dioxide**
CO_2e	**Carbon dioxide equivalent**
DCD	**Dicyandiamide**
DEFRA	**Department for Environment, Food and Rural Affairs**
DMI	**Dry Matter Intake**
DMPP	**3, 4-dimethylpyrazole phosphate**
ELM	**Environmental Land Management**
EU	**European Union**
EWCO	**England Woodland Creation Offer**
FAO	**Food and Agriculture Organisation**
FU	**Functional Unit**
FYM	**Farmyard Manure**
GGR	**Greenhouse Gas Removals**
GHG	**Greenhouse Gas**
GOR	**Government Office Region**
GWP	**Global Warming Potential**
GWP_{100}	**Global Warming Potential (100 years)**
IPCC	**Intergovernmental Panel on Climate Change**
LUC	**Land Use Change**
MACC	**Marginal Abatement Cost Curve**
N	**Nitrogen**
NBS	**Nature-Based Solutions**
NEE	**Net Ecosystem Exchange**
NFU	**National Farmers Union**

NI	**Nitrification Inhibitor**
NUE	**Nitrogen Use Efficiency**
OC	**Organic Carbon**
PA	**Precision Agriculture**
SFI	**Sustainable Farming Incentive**
SLCP	**Short Lived Climate Pollutant**
SOC	**Soil Organic Carbon**
SSP	**Shared Socioeconomic Pathway**
TFP	**Total Factor Productivity**
TMR	**Total Mixed Ration**
UI	**Urease Inhibitor**
UK	**United Kingdom**
UN	**United Nations**
UNFCCC	**United Nations Framework Convention on Climate Change**
WMO	**World Meteorological Organisation**

Chapter 1 - Introduction

The following sections introduce this doctoral research thesis, including the context and rationale behind the research topic, the focus of this thesis, as well as the aims and objectives.

1.1 Research context and rationale

This thesis delves into the science behind agricultural contributions to the changing climate, as well as explore science and policy driven mitigation opportunities unique to the agriculture sector in England and Wales. The following sections provide a brief overview of the agriculture-climate context behind this thesis and the rationale for choosing this topic, with further detail provided in the subsequent chapters of this thesis.

1.1.1 Climate change and the food system

A stable and equitable food system is critical to human survival. As a human society, we overuse planetary resources, which has caused us to surpass six of the nine planetary boundaries (e.g., climate change, nitrogen pollution and land use), which are the limits within with humanity can continue to thrive into the future (Rockström et al., 2020; Richardson et al., 2023).

The security of the food system is pressured by several factors. One of the most pressing concerns for food security is an increasing demand for food, owing to a growing population worldwide. The *Creating a Sustainable Food Future* report by the World Resources Institute (Searchinger et al., 2019) suggests that by 2050 food demand will rise by more than 50%, and there is still the challenge of malnourishment and hunger due to poor distribution of food resources.

In total, 46% of the habitable land on Earth is used for agricultural purposes, compared to 1% of land that is classed as urban and built-up (Ritchie and Roser, 2019). Roughly 12-14% of ice-free land worldwide is used for cropping, and close to 21% is used for pasture (intensive and extensive) (Shukla et al., 2019). However, on a global scale, the food system contributes roughly one quarter of total greenhouse gas (GHG) emissions (Poore and Nemecek, 2018), which enhances the greenhouse gas effect and thus, global warming.

There has already been at least a 1°C increase in global surface temperatures above pre-industrial levels (1800s), with a trajectory of 1.5°C by 2040 according to the Intergovernmental Panel on Climate Change (IPCC) (IPCC, 2018; Lal, 2020). Climate change has demonstrable consequences including temperature rise, increased precipitation variation and frequency of extreme weather events (e.g. droughts and floods), which impacts economies, people, biodiversity and food security (Vermeulen et al., 2012).

Worldwide goals and targets have been set to keep further temperature rise well below 2°C, i.e. the Paris Agreement (United Nations Framework Convention on Climate Change [UNFCCC], 2015), although climate change mitigation is also noted in older initiatives such as the Kyoto Protocol and Sustainable Development Goals (goal 13). In 2019, the UK Government became the first nation to set a target of achieving ‘net zero’ GHG emissions by 2050 under the advice of the Climate Change Committee (CCC) to keep in line with the Paris Agreement goals (CCC, 2019; UK Government, 2019). This means that the UK must reduce its GHG emissions 100% from 1990 levels by 2050, and suggests that any GHGs the UK produces after this point would need to be equal to or less than emissions (mostly carbon dioxide) removed (Brader, 2023).

Furthermore, an ambition for a net zero UK agricultural system was introduced by the National Farmers Union (NFU) in 2019 under their *Net Zero: Farming’s 2040 Goal* report, which aims to deliver net zero farming using three pillars of mainly production-phase change. These pillars focus on 1) improvements in productivity, 2) farmland carbon storage, and 3) renewable energy and bioenergy crops (NFU, 2019). The CCC have indicated that a 64% reduction in UK land use emissions is achievable by 2050, however with UK agricultural emissions being 46.4 million tonnes (Mt) CO_2e in 2020, this means that a 54.7% reduction is still required (CCC, 2020a; Buckingham et al., 2023)The term ‘net zero’ has gained significance in both climate science and climate policy in recent years. Net zero was originally used to describe efforts to stabilise atmospheric carbon dioxide (CO_2) concentration, by balancing CO_2 emissions with CO_2 removals (i.e., net zero CO_2). However, in agriculture, where non-CO_2 GHGs like methane (CH_4) and nitrous oxide (N_2O) are dominant, net zero CO_2 emissions is less relevant and some authors argue that the term ‘climate neutrality’ would be more useful as it is describing the situation whereby human activities neither increase nor decrease long-term global average temperatures (Allen et al., 2022). For the purpose of this thesis, the meaning of the term net zero is twofold. Firstly, it describes the reduction of CO_2 and non-CO_2 GHG emissions to stabilise and reduce the atmospheric concentration of these gases, and thus the increasing warming effect that has been occurring since the pre-industrial period as a result of anthropogenic activity. Secondly, any removals of CO_2 from the atmosphere offset CO_2 production and may not be able to compensate for the warming impact of residual non-CO_2 emissions. However, carbon removals can lead to a slight atmospheric cooling effect if done at scale (Parliamentary Commissioner for the Environment, 2022; Allen et al., 2022). Existing UK policy sees net zero purely as a 'balance the accounts' scenario, with the amount of "residual greenhouse gas emissions balanced by removals" (CCC, 2025) in carbon dioxide equivalents (CO_2e). However, evidence has shown that practices such as afforestation for carbon removal need to be at a scale larger than most countries could feasibly resource to achieve the same amount of atmospheric cooling as the ongoing warming effect of current CH_4 and N_2O sources (Parliamentary Commissioner for the Environment, 2022).

Whilst net zero has become a headline term for leading efforts across all scales and sectors to work towards a climate neutral future, the question of whether a net zero UK and, more specifically in relation to this PhD thesis, a net zero farming system in England and Wales remains unanswered. For example, there are uncertainties around the feasibility of the target timeframe, how long net zero could be maintained for if achieved, the innovation gap required to meet net zero and the many barriers that could prevent or limit the capability of a net zero system (Dyke et al., 2021; Hale et al., 2022; Rosa and Gabrielli, 2023). In agriculture, net zero is often discussed as if it were a blanket solution to reducing agricultural GHG impacts. However, the spatial and temporal complexity of farming and its interactions with the wider biosphere (water cycle, soils, biodiversity, society) makes finding the solutions to contribute to a net zero farming future challenging (Smith et al., 2008; Smith, 2012).

Therefore, this thesis will build upon this area of knowledge and scientific development to better understand the GHG impacts of a typical farming system and model scenarios across different farming systems to assess the feasibility of a net zero agriculture system in England and Wales.

1.1.2 Focussing in on agriculture

The rationale for the choice of topic stems from the unique disposition of the agriculture sector, which is capable of both reducing GHG emissions through more efficient practice and technology, and sequestering carbon dioxide (CO_2) from the atmosphere into biomass and soils, which reduces atmospheric CO_2 concentration, limiting further warming (Smith et al., 2008; Lal, 2020). Land-based sectors like agriculture present a potentially significant opportunity to mitigate the impacts of climate change through the adoption of better farming practices and increased carbon removals.

Agriculture as a sector contributes roughly 11% of UK GHG emissions. Other economic sectors produce almost twice or more the GHG emissions of agriculture, such as the transport (24%), energy (21%) and business sectors (18%) (Department for Business, Energy and Industrial Strategy, 2022a). However, these sectors have greater potential to decarbonise at a quicker rate than agriculture due to increasing use of renewable energy sources compared to fossil sources, minimising CO_2 emissions. A key part of this decarbonisation also suggests that land-based sectors like agriculture and forestry should be utilised to help other sectors offset GHG emissions. These strategies include Bioenergy with Carbon Capture and Storage (BECCS) and Nature-Based Solutions (NBS), such as afforestation, which stem from the concept of Greenhouse Gas Removals (GGR) (National Infrastructure Commission, 2021). This puts the agriculture sector back into the spotlight of scientific research, which has provided reason for this doctoral research project.

1.1.3 The agriculture sector's perspective on GHGs

Whilst the scientific and policy understanding of agriculture's contributions towards net zero is developing, the perspective of those more directly involved in the agri-food sector (e.g., farmers and landowners, food, and beverage businesses) is less concrete.

The Farm Practices survey by the Department for Environment, Food and Rural Affairs (DEFRA) examines the opinions and actions of the farming community in England towards GHG emission reductions on their farms. The survey reports annual trends in farmer opinions towards the importance of reducing GHG emissions on-farm, motivations for doing so, reasons for not doing so and the actions that are commonly taken. Survey results are available from 2013 to 2024 at time of writing (Defra, 2024b) and can be seen in Figure 1.1 for one of the main questions.

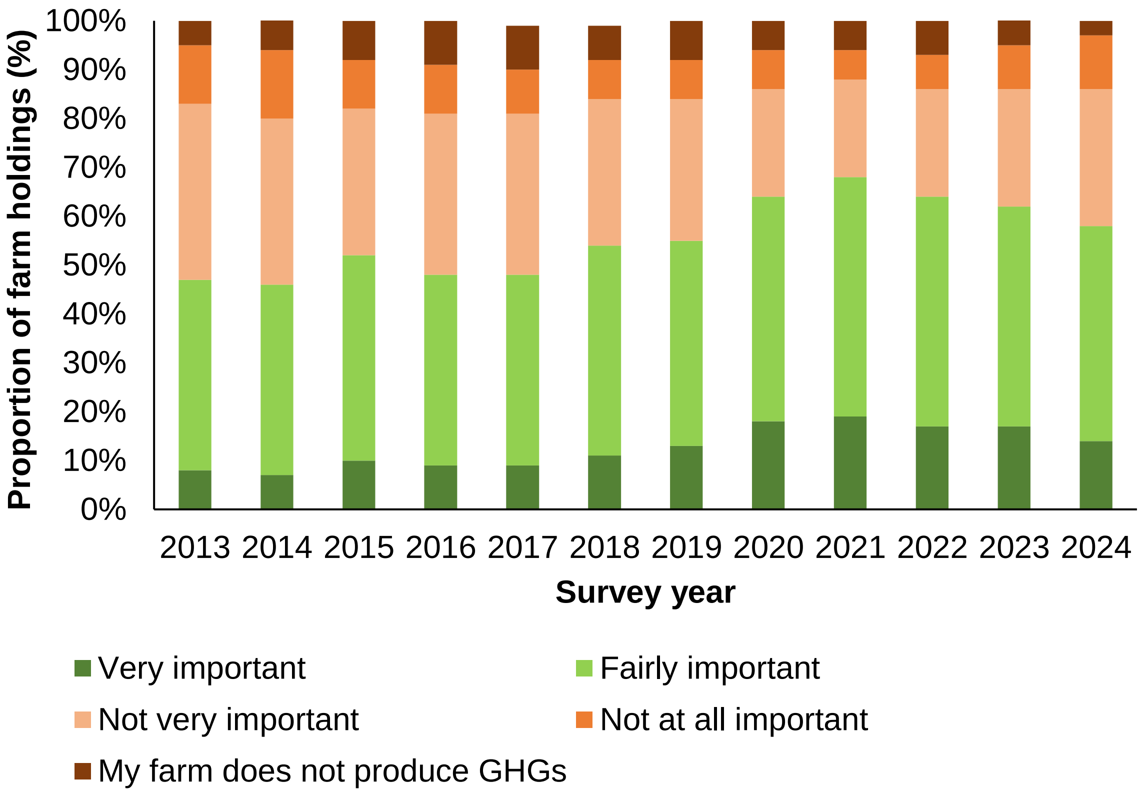

Figure 1.1 Annual responses of farmers in the Farm Practices Survey on the importance of GHGs when making decisions about their farm practices (Defra, 2024b). Responses include Very important (darker green), Fairly important (lighter green), Not very important (pale orange), Not at all important (darker orange) and My farm does not produce GHGs (brown).

Across the eleven years of survey data, the proportion of farm holdings considering GHGs to be 'very important' or 'fairly important' in farm practice decision making gradually increased over time by 11%, but 2021 was the peak in this trend. The proportion of 'not very important' and 'not at all important' responses increased year-on-year from a combined 26% in 2021 to 39% in 2024. On average over the 11 years, 7% of farm holdings said that their farm did not produce GHGs (Figure 1.1). Whilst these trends do indicate that some of the farming community are beginning to be more open to factoring in GHGs and climate change in their farm decision making, there is still a significant proportion who aren't on the same journey. Global geo-political factors also at play during this time could have influenced the opinions and priorities of farmers in England, such as the Covid-19 pandemic of 2020 and the Russian invasion of Ukraine in 2022.

The survey also records GHG mitigation practices taken by farmers over time. Between 2013 and 2024, an average of 58% holdings were taking action to reduce GHGs, but this has been declining from 62% in 2013 to 48% in 2024. The most common actions reportedly taken were recycling waste materials from the farm (11-year average of 84%), improving energy efficiency (75%), and improving nitrogen fertiliser application accuracy (68%). Whilst it is important that recycling schemes are extended to farms and farm buildings and equipment are as energy efficient as possible, these are ‘low hanging fruit’ options for farmers that carry little risk and minimal GHG savings (Innovation for Agriculture, 2021). Since 2021, the most common reason reported in the survey (41% to 44% of responses) for not taking action to reduce GHGs was that the farmer was ‘unsure what to do’ due to conflicting information (Defra, 2024b).

Overall, these results are concerning but an important insight into the current knowledge and appetite of the farming community in the question of a net zero agriculture sector, which fuels the rationale for this PhD work to help provide further scientific evidence for mitigation.

1.2 Aims and objectives

The overarching aim of this doctoral thesis is to better understand whether the agriculture sector could meet net zero GHG emissions by modelling agricultural emissions and GHG mitigation practices at varying scales. Outcomes are assessed against national-level legally binding targets (i.e., net zero GHG emissions by 2050), to discuss implications for policymakers and future research opportunities. However, the findings of this thesis are also applicable to sector-level targets (i.e., net zero agriculture by 2040) with implications for farmers on which practices to choose and where these should be taken up most.

Firstly, in Chapter 2 I present a literature review of the interactions between climate change and the agriculture sector to introduce key topics referred to throughout the thesis, such as the scientific basis of GHGs, the dominant sources of emissions from agriculture, and GHG mitigation and carbon removal options.

In Chapter 3, I investigate the main GHG sources of a commercial-scale mixed arable and pig case study farming system (University of Leeds farm), which is representative of arable and pig production in England and Wales, and model opportunities for GHG reductions at the farm-level. I use Life Cycle Assessment (LCA) to develop a baseline annual average GHG emissions inventory for the arable crops grown on-farm and for pig production and estimate the global warming impact of both production systems. I then model the GHG reduction opportunity of two key farming practices changes on the baseline data, including the use of nitrification inhibitors in nitrogen fertiliser applications and acidification of stored pig slurry. These practices are relevant to farm management during the baseline period, with crops receiving nitrogen without any inhibitors and the pig unit increasing its capacity with a need to manage increasing quantities of liquid slurry. The implications of the LCA analysis, data quality and farm-level practice choice for farmers are explored.

In Chapter 4, I transition from farm-level modelling to a spatial investigation of regional-scale opportunities for arable agricultural land and afforestation to contribute to net zero policy. I do this by identifying suitable land area for GHG reduction and afforestation and identify cropland-specific farming practices that mitigate emissions and for which there is data on the current level of uptake in each region. I then model these farming practices as scenarios of 10% uptake and a 1% one-off planting of new woodland to assess the potential contribution of arable agriculture and afforestation to net zero at regional and national scale. The implications of practice uptake and feasibility of afforestation to balance GHG emissions for policymakers are discussed.

In Chapter 5, I perform a similar regional-scale scenario analysis as Chapter 4 but with a shift in focus to the livestock sector and, in particular, methane reductions. Using UK GHG Inventory and livestock population data at a regional level, I estimate enteric fermentation and manure management methane emissions for cattle and pigs and model a 10% uptake of a recently approved methane inhibitor (3-Nitrooxypropanol) and re-model slurry acidification using the same factor as Chapter 3 to determine how much methane emissions could be reduced in line with net zero goals. The implications of practice choice and uptake potential are discussed.

Chapter 2 - Literature Review

In this chapter a literature review of agriculture-climate change interactions is presented, exploring the dominant greenhouse gas (GHG) emissions from agricultural activities and metrics used to evaluate their impact. This chapter also discusses key agri-policy at global and UK scale, and the scientific evidence behind farming practices that could reduce GHG emissions or remove carbon dioxide (CO_2) from the atmosphere to limit further warming. Lastly, an overview of the agriculture sector in England and Wales is given, as this forms the geographical extent of this thesis.

2.1 Introduction

In the current century, records are being broken worldwide for observed changes in climate that have strong connections with the effects of anthropogenic global warming due to the rise in GHG emissions. GHGs refer to atmospheric gases that are capable of trapping heat in the atmosphere, which creates a warming effect on the planet (the Greenhouse Gas Effect) (Ledley et al., 1999). The Intergovernmental Panel on Climate Change (IPCC) *6th assessment report (AR6)* suggests global climate scientists are more confident than ever that our climate has been changing and will continue to change as a result of anthropogenic activity (IPCC, 2021; IPCC, 2022).

Global surface temperatures have risen by more than 1°C above pre-industrial levels (1850 - 1900) and there is high confidence amongst scientists that the period of 2016 - 2020 was the warmest five-year period since 1850 to date (Arias et al., 2021). Current climate models suggest a trajectory of increasing global temperature change by roughly +0.2°C each decade, reaching 1.5°C by 2040 (Figure 2.1) (IPCC, 2018; Lal, 2020). The intensity and frequency of severe weather events, such as storms, heatwaves, droughts and flooding have increased in recent years (Abbass et al., 2022). Sea levels are rising largely in response to the melting of the polar ice caps due to the increases in air temperature, leaving low-lying areas exposed to the threat of flooding. Although there are several natural occurrences of a changing climate, e.g. ocean circulation patterns that impact regional weather, there are increasingly more frequent and intense occurrences that are anthropogenically influenced (McNutt, 2013) and these are not uniform across the planet.

With a changing climate comes vast and often problematic impacts to the environment, society, world economies and food security. Global efforts to mitigate and adapt to this changing climate are trans-disciplinary, enhancing the complexity of the solutions required. Most global leaders recognise the importance of setting and progressing towards targets that reduce the impacts of climate change, and therefore the impacts that a changing climate brings (Bernauer, 2013).

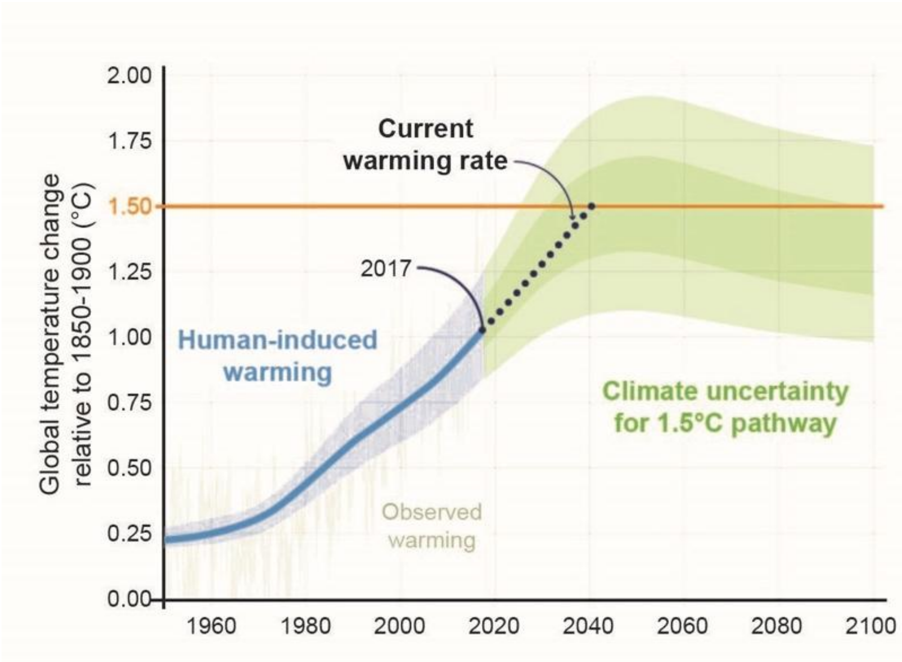

Figure 2.1 Anthropogenically enhanced warming has surpassed 1°C above levels observed in the pre-industrial era (1850-1900) and the trajectory is set to reach a 1.5°C increase by 2040 (IPCC, 2018).

2.1.1 Greenhouse Gas impacts on climate

To understand the climate influence of GHGs in the atmosphere on the GHG effect, and thus global warming, it is essential to first identify the properties of the main GHGs. There are several principal GHGs in the Earth's atmosphere. Some are produced naturally and some are produced industrially, and include CO_2, methane (CH_4), nitrous oxide (N_2O), water vapour and ozone (O_3) and others that are the result of industrial processes only include the fluorinated gases: chlorofluorocarbons (CFCs), hydrochlorofluorocarbons (HCFCs), hydrofluorocarbons (HFCs), perfluorocarbons (PFCs) and several other compounds (IPCC, 2013; IPCC, 2021). For the purpose of this literature review, only the main three GHGs (CO_2, CH_4 and N_2O) will be discussed henceforth as these gases are highly linked with agriculture (Smith et al., 2008).

The concentration of CO_2, CH_4 and N_2O has increased substantially since the pre-industrial era, which is undoubtedly the result of human activity. Between 2011 and 2019, when the IPCC's *5th Assessment Report* (AR5) and *6th Assessment Report* (AR6) were released, respectively, the concentrations of the main GHGs had further increased (Table 2.1).

Table 2.1 GHG concentration data comparisons between 1750 and 2019 for the main atmospheric GHGs.

GHG	Average concentration in 1750 (pre-industrial)[a]	2019 annual average concentration[b]
Carbon dioxide (CO_2)	278 ppm	410 ppm
Methane (CH_4)	719 ppb	1866 ppb
Nitrous oxide (N_2O)	270 ppb	332 ppb
PFCs	-	109 ppt
HFCs	-	237 ppt
HCFCs	-	1032 ppt

[a] Data from European Environment Agency (EEA, 2019)

[b] Data from IPCC AR6 (IPCC, 2021)

Emissions of each GHG have a different effect on the atmosphere and resulting climate impact. A change in the concentration of atmospheric GHGs alters the atmospheric energy balance, known as radiative forcing (Forster et al., 2021; FAO, 2023b). Each GHG also has a different lifetime in the atmosphere, which is the amount of time the GHG remains in the atmosphere before it is broken down and removed (Forster et al., 2021).

Emission metrics are used to compare the climate impacts from emissions of each GHG, typically non-CO_2 GHGs compared to CO_2 (Shine, 2009). Without a specific climate impact metric, radiative forcing is often used as a proxy measurement for emission metrics to compare the various GHGs (FAO, 2023b). One such emission metric is the Global Warming Potential (GWP), which “compares the radiative forcing accumulated over a user-defined time horizon resulting from a pulse emission of a specific GHG compared to a pulse emission of an equal mass of CO_2” (FAO, 2023b). The most common time horizon used for GWP (e.g., by the IPCC) is 100 years (GWP_{100}) and is reported in carbon dioxide equivalents (CO_2e) (Shine, 2009; Forster et al., 2021). GWP_{100} has been criticised by scientists as it over-simplifies the actual climate impact and atmospheric interactions of Short Lived Climate Pollutants (SLCP), like methane (Shine, 2009; Lynch et al., 2021), which breaks down in the atmosphere after roughly a decade (Allen et al., 2018; IPCC, 2021). Figure 2.2 illustrates this challenge and is taken from Lynch et al. (2021). The left graph shows a change in emissions reported in CO_2e using the GWP_{100} metric between 1950 and 2050. The right graph then shows the resulting warming impact broken down by scenarios where each GHG represents 100% of CO_2e. This figure demonstrates that the GWP_{100} metric oversimplifies the complex interaction of CH_4 (yellow line) with the atmosphere, which has a greater initial warming effect than CO_2 when emissions are increasing and then the majority of the warming is reversed when CH_4 emissions are in decline (Lynch et al., 2021).

CO_2 is used as the reference to which all other atmospheric gases are compared, and so has a GWP of 1 (for any time period). However, the lifetime of CO_2 in the atmosphere is largely unknown (Archer et al., 2009; Myhre et al., 2013). By 2017, concentrations of CO_2 in the atmosphere had reached 405.01 ± 0.01 parts per million (ppm), an increase of almost 130 ppm since 1750 (Le Quéré et al., 2018) and the 2023 Global Carbon Budget estimates this to have increased to 417 ppm (Friedlingstein et al., 2023). The same Global Carbon Budget estimated that in 2022 2.8 ± 0.4 Gigatonnes (Gt) carbon (C) was taken up by oceans (25%), 3.8 ± 0.8 Gt C was taken up by terrestrial vegetation (34%) and 4.6 ± 0.2 Gt C remained in the atmosphere (41%) (Friedlingstein et al., 2023). This means when a pulse of CO_2 is emitted, just over two fifths of the quantity of gas stays in the atmosphere, having a warming impact lasting several centuries or millennia before it is broken down, thus contributing significantly to climate change (Allen et al., 2018). Therefore, any further contributions of CO_2 into the atmosphere will accumulate over very long timeframes, increasing the atmospheric concentration (the stock), which traps heat and further warms the planet.

In contrast, CH_4 has a GWP_{100} that varies between fossil and non-fossil sources of the gas, at 29.8 and 27.2, respectively (IPCC, 2021). Fossil CH_4 sources are those typically associated with industrial processes, whereas non-fossil (biogenic) CH_4 is produced naturally from biological processes. The GWP of CH_4 means that for every tonne of the gas produced (biogenically), this would be equivalent to almost 30 tonnes of CO_2. Another difference between CH_4 and CO_2 is the gas lifetime in the atmosphere; CH_4 is a Short-Lived Climate Pollutant (SLCP). The IPCC AR6 revealed that the lifetime of CH_4 is 11.8 years, with literature sources usually citing 10-12 years (Allen et al., 2018; IPCC, 2021). After around a decade, CH_4 is broken down in the atmosphere into water vapour and CO_2 through reactions with hydroxyl radicals (Voosen, 2021). This means that an increase in CH_4 emissions will not accumulate in the atmosphere for millennia like CO_2 but will initially raise the concentration of atmospheric CH_4 before being broken down. If emissions of CH_4 were to stabilise or decline, then the overall atmospheric concentration would remain stable or decline, respectively, with the latter possibly leading to a slight cooling effect on the planet. There is an active debate about the efficacy of GWP_{100} as a metric for estimating CH_4 contributions to climate change due to the gas being a SLCP and the GWP_{100} methodology not taking the lifetime of the gas into account. Researchers from the University of Oxford have developed an alternative metric, GWP*, which accounts for the *rate* of CH_4 increase or decrease to the atmosphere, rather than assuming the gas remains in the atmosphere for at least 100 years (Allen et al., 2018; Cain et al., 2019; Lynch et al., 2020). However, the GWP_{100} method is embedded in international climate policy so there is unlikely to be a shift to alternative metrics or models until there is extensive research supporting it.

The other main non-CO_2 GHG, N_2O, has the highest GWP_{100} of the three main gases having 273 times more of a warming impact on the atmosphere than CO_2 and it has a longer lifetime than CH_4 at ~109 years (IPCC, 2021). N_2O is mostly produced through natural sources that stem from the nitrogen cycle, which is described later in Section 2.3.2, but is heavily influenced by human activity including agriculture. Research has also shown that N_2O depletes ozone in the stratosphere, alongside CFCs which were largely responsible for the ozone hole over the Antarctic (Ravishankara et al., 2009; Müller, 2021). Ozone is important for life on Earth as it acts as a barrier to ultraviolet radiation from the sun, which if reduced can penetrate the skin and membranes of humans and animals causing damage to the genetic makeup of the organisms affected (EEA, 2021).

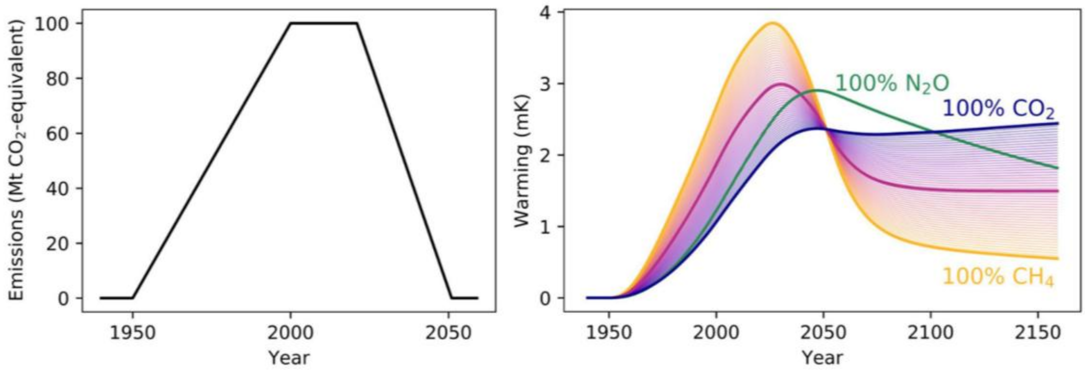

Figure 2.2 A comparison between an emission scenario and the resulting warming impact. The left graph shows the change in emissions over time using the typically reported metric of million tonnes (Mt) ‘CO_2e’ under GWP_{100}. The right graph shows the warming impact (mK) of N_2O (green), CO_2 (blue) CH_4 (yellow) if each represented 100% of CO_2e emissions, respectively. Taken from Lynch et al., 2021.

2.1.2 Anthropogenic climate change

The main GHGs outlined above do occur naturally in the atmosphere and are essential to the survival of life on Earth; without the GHG effect the Earth would experience an average temperature of around -19°C. Considering Earth’s average temperature sits at around 14°C, this gives the greenhouse gas effect a warming potential of 33°C (Ledley et al., 1999; Kweku et al., 2018). However, accelerated anthropogenic burning of fossil fuels and poor land management since the pre-industrial era has resulted in a sharp increase in GHG emissions that has enhanced the GHG effect, culminating in a global climate emergency that could see average temperatures increase between 1.5°C and 4°C, compared to pre-industrial levels, in the next 30 years (Karl and Trenberth, 2003; IPCC, 2018; Le Quéré et al., 2018; IPCC, 2021).

Some of the main threats of changes in global climate include higher temperatures, more varied rainfall patterns and increased frequency of extreme weather events, e.g., heatwaves, droughts, storms, and floods. These threats have widespread impacts on a global scale, affecting people, biodiversity, economies and food security (Vermeulen et al., 2012). Some of the observed impacts of these climate threats noted in the IPCC special report *Global Warming of 1.5°C* (IPCC, 2018) include alterations to disease and pest species patterns, changes in natural cycles and seasons (e.g., plant flowering phenology), more widespread tree mortality, as well as impacts to society (e.g., business losses, displacement, health concerns). All of these threats and impacts affect the agriculture sector and will make farming more challenging with each year.

2.2 Climate change impacts on agriculture

Land-based sectors like agriculture and forestry are vulnerable to the impacts of a changing climate because of the biological, chemical, and physical links between the land (including soils), atmosphere and water. In turn, this puts the livelihoods of millions of farmers and landowners worldwide at risk of severe business losses and even failure as the effects of climate change worsen and uncertainty towards food security into the future increases (Godfray et al., 2010). There has already been noticeable changing in global climate, which have consequently impacted the food system, and this is predicted to get worse with further climate impacts. For example, by 2070 average warming and precipitation during summer in the UK is predicted to be 1.3°C to 5.1°C and range between a 45% decline in precipitation and a 5% increase in high emission scenarios, respectively (Met Office, 2019). Both of these factors will make cropping and livestock farming more unpredictable and challenging across the UK, including in England and Wales. The following sections outline some of the major climate threats and impacts on agriculture and the food system.

2.2.1 Temperature and CO_2 fertilisation

Eighty percent of energy supply in the human diet is derived from crops like cereals and grains, fruit and vegetables, oilseeds, roots, and tubers. Roughly six hundred million farms worldwide produce some sort of crop, providing a source of income and tradeable produce to meet global crop demand (Lowder et al., 2019; FAO, 2019). Plants photosynthesise to produce their own energy for growth by using light energy to convert the CO_2 drawn down from the atmosphere and water taken up through the roots into sugars and oxygen. Photosynthesis is therefore dependent on factors including CO_2, water and light availability, which are also linked to changes in temperature and thus, the changing climate (Chen et al., 2022).

There is growing confidence amongst the scientific community that the increasing global surface temperatures being experienced and projected to increase further could pose both threats and advantages to agriculture in particular regions. For instance, the latest IPCC AR6 report suggests that regions in higher latitudes have begun to see an expansion of available growing area closer to the poles and reduced effects of cold stress in parts of North America and East Asia. On the other hand, mid to low latitude regions that are usually warmer without the effects of climate change are having to adapt quickly to the increasing temperatures and have experienced reduced yields (5-27%) (IPCC, 2022).

Furthermore, when modelling the ratio of total agricultural outputs to total agricultural inputs (Total Factor Productivity; TFP), Ortiz-Bobea et al. (2021) found anthropogenic climate change impacts have resulted in just over one fifth of a reduction in agricultural TFP worldwide over the past six decades. The authors also found this trend to be more significant (roughly 26-34%) in low-latitude regions (e.g., Africa, central and Latin America) that naturally experience warmer temperatures. Zhao et al. (2017) analysed data on declines in yield for four of the world's most highly consumed crops, wheat (*Triticum aestivum*), maize (*Zea mays*), rice (*Oryza sativa*) and soybean (*Glycine max*), which have been linked to increasing global temperatures at multiple scales. The authors suggest that without appropriate adaptation, each incremental 1°C increase in mean temperature would result in average yield reductions of 6% for wheat, 7.4% for maize, 3.2% for rice and 3.1% for soybean (Zhao et al., 2017).

Changes in temperature also impact livestock animals being raised to produce meat, milk, eggs, and wool. A literature review by Cheng et al. (2022) outlined the predominant climate impacts on livestock production, noting increases in temperature (heat stress) to be a direct influence on higher mortality rates, lower feed intake and lower productive outputs and an indirect negative influence on livestock feed production (Cheng et al., 2022). Additionally, it has been observed that 8.5% - 12.5% of livestock species populations experienced at least one day of heat stress at the start of this century in temperate climates (6.8% - 12.5% in tropical climates). However, projections to the end of the century for two climate scenarios (Shared Socioeconomic Pathway [SSP] 1-2.6 and SSP 5-8.5) indicate substantial increases in these proportions, particularly for poultry, where an estimated 75% of the population is affected by one heat stress day a year in 2090 under SSP 5-8.5 (temperate and tropical) (Thornton et al., 2021).

With atmospheric CO_2 concentration accumulation through human activity, there is an opportunity for farmers to increase productivity of their crops through CO_2 fertilisation. CO_2 fertilisation is the phenomenon when plant uptake of CO_2 during photosynthesis increases as the concentration of CO_2 available in the atmosphere increases (Challinor et al., 2009; Challinor et al., 2018; Huntingford and Oliver, 2021). The potential opportunity from this mechanism is still being investigated through laboratory and field research, and there are other limiting factors for plant productivity (such as water and nutrient availability) making it difficult to draw definitive conclusions. Research from Chen et al. (2022) used global eddy covariance measurements to demonstrate that 44% of Gross Primary Production (GPP) enhancement is the result of CO_2 fertilisation since the start of the century, although limitations in soil moisture and humidity have created interannual variability.

2.2.2 Precipitation

Crop production is directly impacted by changes in rainfall as plants need sufficient water to grow and a reserve of water in the soil for during dry periods. In regions where irrigation is necessary due to naturally lower rainfall, more frequent precipitation associated with a changing climate may improve crop productivity and reduce the need for irrigation infrastructure. However, intense *and* frequent precipitation can cause flooding, particularly in areas with drier and more compacted soils, or under drought conditions water reserves could shrink, both of which could reduce yields or cause crop failure (Calzadilla et al., 2013).

If rainfall intensity increases, this could have damaging effects on the soils that grow our crops and feed for livestock. Soil erosion leading to sediment, and thus nutrient, losses could increase under intense rainfall events if soils are left bare (e.g., after winter crop harvest) and exacerbates the risk of water pollution, e.g., eutrophication from nitrates or phosphorous (Klik and Eitzinger, 2010; Fowler et al., 2015). Furthermore, agricultural machinery cannot operate on wet soils to avoid compaction, and especially when waterlogged, meaning farmers will have to be more adaptable in regards to sowing, harvesting and input applications, which may cause reduced yields and less food available to consumers (Hamza and Anderson, 2005).

2.2.3 Droughts

Yield losses relating to droughts have affected three quarters of the cropped area on the planet, with an increase in frequency of these drought-related yield reduction events in the last decade (IPCC, 2022). An analysis of long-term extreme weather events at global scale revealed that droughts reduced cereal production by at least 10% (9.9% – 10.2% confidence interval at 95%) and cereal yields by 5% (4.9% - 5.2%). When looking at the results regionally the authors found that more developed regions, including Europe, North America, and Australasia, experienced more significant drought impacts than lower-income countries and regions at almost 20% production deficiency and 16% yield reductions. This is likely due to the higher tendency for technically developed countries to employ monocultures to maximise profit, whereas less economically developed countries use more risk-averse farming strategies (Lesk et al., 2016).

2.3 Agriculture's contribution to climate change

Emissions of GHGs occur from nearly all agricultural activities, although some have the potential to sequester carbon. Roughly 12% of GHG emissions stem from agricultural activity worldwide (Smith et al., 2014), with production-phase agricultural GHG emissions estimated to contribute roughly 80% of total global agricultural emissions (Vermeulen et al., 2012). In the UK, agricultural contributions represent around 11% of the total footprint (Department for Business, Energy and Industrial Strategy, 2022a). CO_2 from agriculture only contributes around 1% of total CO_2 emissions in the UK, whereas agricultural CH_4 and N_2O contribute approximately 49% and 71% of total methane and nitrous oxide emissions, respectively (Defra, 2023a). Being a land-based economic sector, agriculture also has the potential to be a large carbon sink, which can sequester CO_2 from the atmosphere and store it through time in perennial vegetation (i.e., hedgerows, trees, grassland) and soils.

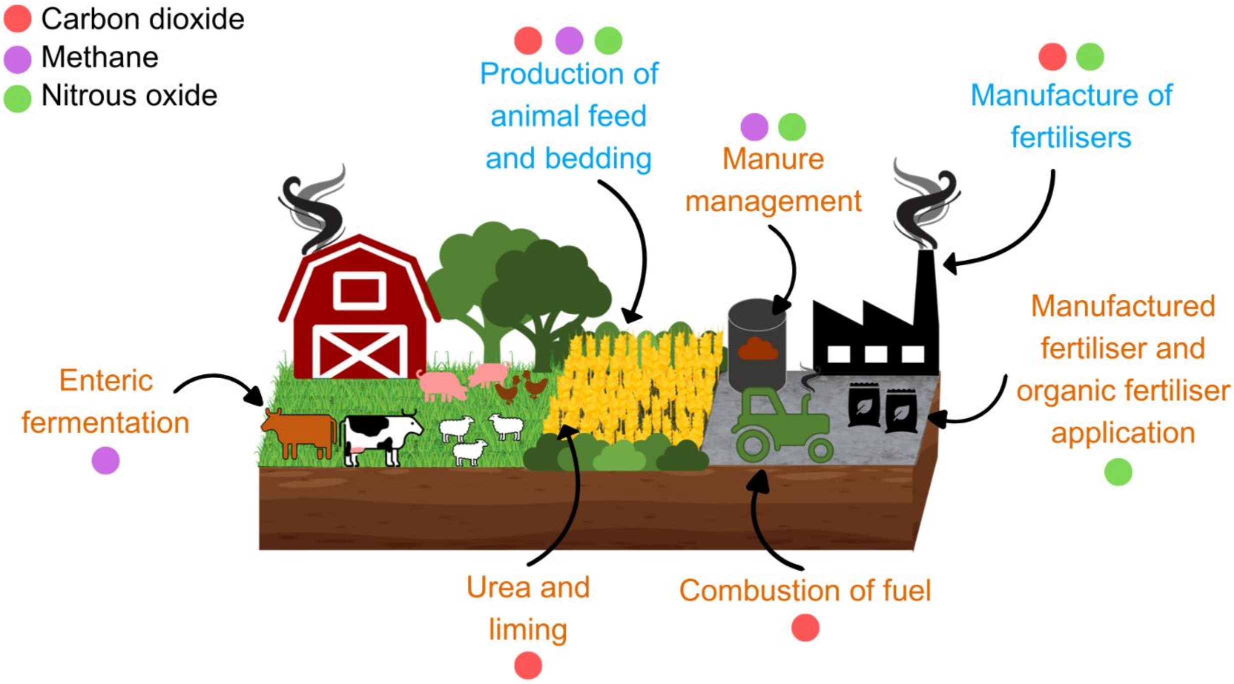

Figure 2.3 Main sources of GHGs on-farm (orange text) and off-farm (blue text) that would be necessary to calculate and reduce. The circles represent the main GHG(s) emitted from that activity. Red = Carbon dioxide, green = Nitrous oxide, and purple = Methane.

The main sources of GHG emission in agricultural production are outlined in the following sections, but a simple schematic of the main activity hotspots is found in Figure 2.3.

2.3.1 Land Use Change and management

Over time, land is changed and developed from its natural state to fit demand and purpose. Natural forested areas are cut down to increase agricultural area, natural grasslands are converted to woodland or enter into a rotation with arable crops, soils are disturbed using machinery to establish seedbeds and reduce weed presence (Smith and Conen, 2004; Dawson and Smith, 2007; Donkersley et al., 2021). Despite all of these activities being suitable to meet the demand of land use management and change, they all have an impact on the carbon that is stored within native vegetation and soils.

Earth has a land surface area of roughly 150 million km^2, covering close to 30% of the planet's surface with the remainder being water. Roughly one third of the land surface area (51 million km^2) is used for agriculture, just over one quarter (39 million km^2) is forested, 8% (12 million km^2) is shrub land, around 1% (1.5 million km^2) of land is urban or built up, and 1% is freshwater (Ritchie and Roser, 2019). As the global population increases, the demand for space to expand urban areas and food production to feed the growing population and livestock increases. This adds pressure to some of the world's biggest sustainability challenges: climate change, food security and biodiversity loss. Expansion of agricultural land through destruction of others is one of the most detrimental environmental concerns and is an example of Land Use Change (LUC). LUC has severe impacts on the ecosystem services and functions that natural habitats provide, with most changes being negative. For example, loss of biodiversity and habitats, reduced water quality, loss of cultural value and poorer climate regulation. Looking more closely at climate impacts of changing land use, there are concerns about CO_2 fluxes (Winkler et al., 2021).

Fluxes, or fluctuations, of CO_2 between the land and atmosphere is heavily influenced by land use change. Carbon dioxide is stored in terrestrial sinks that include biomass (both above and below ground) and soils as organic carbon (OC). When the soil is turned over and aerated, for example when converting a semi-natural pasture to cropland, or when biomass is harvested, burned, or chopped down, the OC is exposed to oxygen (O_2) which converts to CO_2 (Lal et al., 2018). The largest source of CO_2 emissions in agricultural systems is from LUC, which is typically the result of large-scale land clearance for agriculture, for example deforestation to make way for pasture systems. LUC sourced CO_2 emissions are estimated to contribute 14% of anthropogenic CO_2 each year (Lynch et al., 2021). As mentioned in Section 2.1.1, CO_2 is a long-lifetime GHG that has an accumulation effect in the atmosphere that drives the greenhouse gas effect, causing global warming. Alternatively, if land areas are converted to a more native state, such as forest, woodland, and permanent grassland, then a large area of continuously photosynthesising plants will be sequestering atmospheric CO_2.

Carbon dioxide is exchanged between plants, soils, and the atmosphere in various natural processes, which make up the carbon cycle. Plants, such as crops, photosynthesise by taking in CO_2 from the atmosphere and converting it into energy - glucose - to be used for plant growth. Plants also respire, which releases CO_2 and water when the energy generated during photosynthesis is used for growth alongside oxygen (O_2). Both the aboveground (ABG) and belowground (BG) biomass of plants utilise the products of these processes, meaning that carbon is therefore stored and sequestered in plant biomass and soils (from the BG biomass of root systems). CO_2 is also naturally released from vegetation and soils due to respiration from ABG biomass and plant roots, respectively, and both aerobic and anaerobic microbial respiration (Lovett et al., 2006; Oertel et al., 2016).

2.3.2 Soil applications

Soils can be sinks or sources of GHG emissions, with fluxes between states largely determined by soil management and applications (Smith et al., 2001; Oertel et al., 2016). As mentioned in the previous section, soils under natural vegetation such as permanent grasslands and forests are often carbon sinks, meaning that they sequester and store more atmospheric CO_2 as carbon than they emit. On the other hand, soils that are disturbed through LUC and management, such as in agricultural systems, are typically net sources of GHGs. Although the production of GHGs in soil is often the result of natural, microbial, or biological processes, these are exacerbated by anthropogenic activities that enhance pollution.

Nitrogen (N) is the primary gas in the atmosphere, constituting almost 80% by volume (Menegat et al., 2022). Much of the land biomass is unable to take up nitrogen (N) directly from the atmosphere, so N needs to be fixed through industrial processes or specialised N-fixing plants and bacteria (Smith, 2012). Agricultural land plays a significant role in the N cycle, with farm soils interacting with water and the atmosphere through biological activity. Key soil processes, including nitrification and denitrification, convert N into more available forms for plants, but also release varying quantities of N_2O as a by-product. Nitrification occurs when nitrifying soil bacteria break down ammonia (NH_3) from decomposing organic matter (e.g., leaf litter) into nitrites (NO_2^-) and subsequently oxidise nitrites to nitrates (NO_3). On the other hand, denitrification is a microbial process that reduces nitrates and nitrites to nitrogen gas (N_2) and N_2O under anaerobic conditions, i.e. when oxygen is limited (Bernhard, 2010; Fowler et al., 2015).

Significant emissions of N_2O are produced during cropland cultivation, mostly from direct soil emissions when manufactured nitrogen (N) fertilisers are added, such as ammonium nitrate (AN) or urea (Gerber et al., 2016; Carlson et al., 2017). Indirect volatilisation and atmospheric deposition of N stems from the application of manufactured N, organic N and N from livestock deposits. Alternatively, indirect leaching and runoff of N as nitrates can also result in N_2O production at a later stage when entering groundwater or water bodies (IPCC, 2006; Hama-Aziz et al., 2017). A meta-analysis from Menegat et al. (2022) synthesised global, regional and national soil emissions of direct N_2O emissions and emission factors across the supply chain of manufactured N fertilisers. The authors calculated that in 2018 1.13 Gt CO_2e of global emissions stemmed from this supply chain, contributing 10.6% of agricultural emissions. The manufacture of N fertiliser, which typically involved the use of fossil fuels, accounted for close to 40% of total manufactured N fertiliser emissions. However, the greatest contribution to total emissions was from the use of the N fertilisers in the field, accounting for ~ 60% of emissions. Almost two thirds of the total emissions were the responsibility of four contributing entities: China, India, the United States of America (USA) and the European Union (EU28) (Menegat et al., 2022). Therefore, there are opportunities for vast amounts of GHG emissions, particularly N_2O, to be reduced through the mitigation of manufactured N fertiliser use.

At a finer scale, it is estimated that roughly 42% of N_2O emissions from farmland soil is the result of inorganic fertiliser application, which is followed by almost one fifth of emissions from crop residues (19%) and 13% from manure application (CCC, 2018c). Other common field inputs that result in N_2O emissions include organic amendments, which can take the form of animal slurries and manures, crop residues and composts. Livestock animals naturally produce excreta (urine and dung) whilst grazing fields in grassland systems and when housed. Excreta contains a portion of the nitrogen that was in the diet of the livestock, as well as carbon and water. These nutrients and compounds react in natural processes to produce N_2O and CH_4 emissions (Chadwick et al., 2011). On-farm management decisions affecting the fate of manures and slurries impacts the magnitude of direct and indirect GHG emissions produced on-site. For example, it is a common farming practice to collect and spread livestock manure (farmyard manure, FYM) and slurries from housed systems on fields to improve nutrient uptake of crops (see Figure 2.4). However, if the soil being managed is saturated with water or is too dry, then the nutrients in the manure or slurry can be washed off the field or not be taken up completely by the crops.

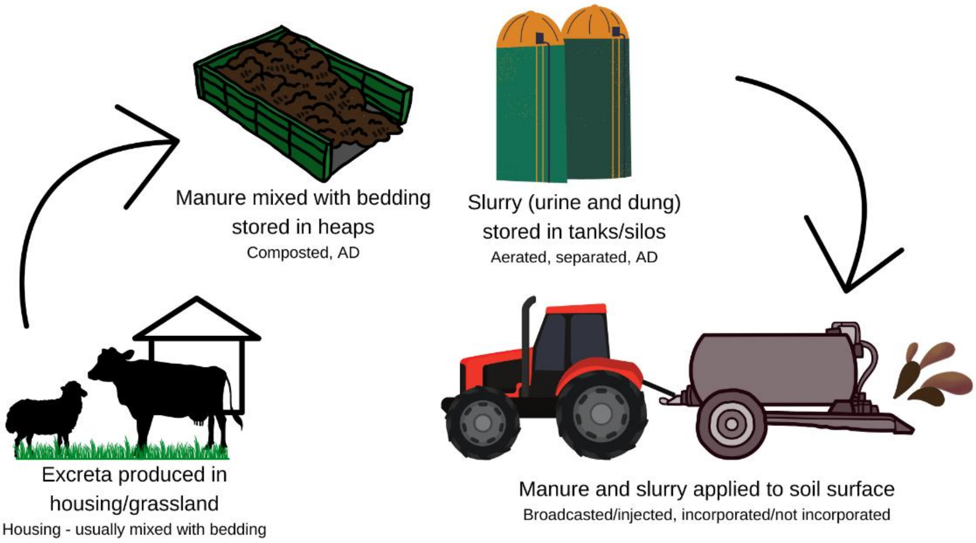

Figure 2.4 Schematic diagram of typical manure management systems for manure and slurry. Adapted from Chadwick et al. (2011).

Crop residues, the biomass left on the soil after harvest of the main crop parts, can be a direct and indirect source of N_2O emissions, which results in climate impacts. Long term studies of crop residue incorporation revealed a 12-fold increase in N_2O emissions on average, but also an average 7% increase in SOC concentration (Lehtinen et al., 2014). Although other studies have found no significant effects of crop residue retention or incorporation on N_2O release (Malhi and Lemke, 2007), and it is likely that there are soil characteristic effects that control the magnitude of N_2O production (Badagliacca et al., 2017).

There are large uncertainties around N_2O emission accounting in agriculture due to the nature of the N cycle, differences in methodology behind N_2O measurements, soil management and exogenous influencing factors that are rarely considered when estimating N_2O from soils, such as soil properties and climate (Brown et al., 2001). For instance, one of the most common methods for taking N_2O measurements from soils is using chambers, which allows N_2O gas to accumulate within a known volume to measure concentration. However, this requires air samples to be taken at regular intervals, which is labour intensive, so fewer measurements are often recorded for manual chamber use (Rapson and Dacres, 2014). Furthermore, N_2O measurements from UK experiments have demonstrated that the emission factor for direct N_2O release following N fertiliser application is roughly 33% lower than the IPCC global average of 1% (Sylvester-Bradley et al., 2015). This makes it difficult to obtain consistent and accurate research on soil based N_2O emissions in agricultural systems but provides interesting research questions surrounding the differences in soil management effects on N_2O.

In cropping systems, the net carbon balance varies with production cycles and management. European cropland soils have been shown to be losing carbon each year. For example, Ceschia et al. (2010) calculated this loss to be 2.6% (± 4.5%) of soil organic carbon each year. Furthermore, direct CO_2 emissions are produced when soils have lime, in the form or carbonates such as limestone ($CaCO_3$) or dolomite ($CaMg(CO_3)_2$), added to them to reduce acidity and optimise plant growth conditions. When the carbonate lime dissolves in the soil, it releases bicarbonate ($2HCO_3^-$), which converts into CO_2 and water (H_2O). The other main form of direct CO_2 release associated with soil inputs is through urea fertilisation, which is when CO_2 that has been fixed and stored in urea ($CO(NH_2)_2$) during the manufacturing process is released during a reaction with water and urease enzymes (IPCC, 2006).

2.3.3 Livestock

Livestock contribute a significant amount of GHG emissions globally every year, which is estimated at 7.1 Gt CO_2e per year (5.6 - 7.5 Gt CO_2e, Herrero et al., 2016) and accounts for roughly 14.5% of all anthropogenic GHG emissions (Gerber et al., 2013). The bulk of emissions that stem from livestock-related agricultural activities are from the direct release of CH_4 during enteric fermentation in ruminants and from manure management (Thorpe, 2009). The FAO reports that roughly 44% of global livestock supply chain emissions are the result of enteric fermentation and nearly 6% and 4% are from manure management CH_4 and N_2O emissions, respectively (Figure 2.5) (FAO, 2017a; Amon et al., 2021). Being ruminant animals, which emit far greater quantities of CH_4 than monogastric animals, cattle (in the beef and dairy sector) and sheep have some of the highest GHG emissions contributions of all (Williams et al., 2006; Bellarby et al., 2013; Amon et al., 2021). CH_4 is a by-product of the natural anaerobic process of microbial fermentation in the stomach of livestock animals. Methanogenic bacteria are specialised in digesting carbohydrates in the rumen under these conditions, and cause the release of CH_4 through the mouth and rectum of ruminants (Hristov et al., 2013).

Most LCAs and research literature account for the direct livestock emission impacts outlined above, but few consider the broader indirect consequences of animal production worldwide (Garnett, 2009; Weiss and Leip, 2012). Indirect livestock emissions that are often left out of GHG assessments include emissions associated with livestock feed, which accounts for approximately 13% of global livestock GHG emissions, and LUC at around ~9% (Figure 2.5; FAO, 2017a). One LCA study of GHG emissions from the European Union (EU) livestock sector revealed that LUC from imported feed production was predominantly found in non-European countries and represented between 9% and 33% of total emissions (Weiss and Leip, 2012).

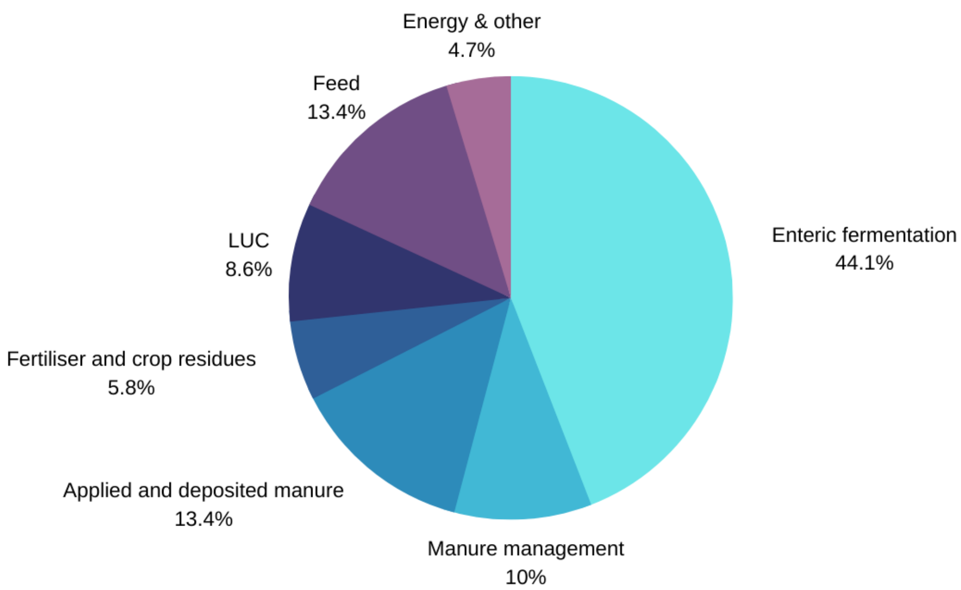

Figure 2.5 Sources of global livestock GHG emissions. Adapted from FAO (2017a).

For many animal products, the GHG emission intensity (quantity of gas per unit of product) is often skewed by large differences in emissions between producers. Poore and Nemecek (2018) reviewed the environmental impacts of various food items and revealed that the top 25% beef producers with the greatest GHG impacts represented 56% of the total beef herd footprint and slightly more than 60% of the land used for beef production. Similar patterns were found for other food products assessed by the authors, with an average of 53% of the environmental impact contribution from the top 25% of producers (Poore and Nemecek, 2018).

2.4 Agriculture-climate policy

To reduce the impacts of climate change and mitigate the contribution of agricultural emissions to further warming, global and national policies, pledges and agreements have been made and progress towards these targets is being monitored. The following two sections describe the key global policies and events that have led to worldwide efforts to mitigate GHG emissions, and the policy implemented in the UK, focussing on England and Wales, which has made the UK a leader in climate change mitigations efforts.

2.4.1 Global policy

There are several key moments in recent history where global policies and agreements have been initiated and enforced with the aim of limiting and reducing GHG emissions, some of which are outlined below.

The first World Climate Conference in Geneva (1979) was instrumental for bringing global climate change scientists together to understand the physical basis of climate change, how anthropogenic activity influenced climate change, and the impacts of climate on humans and our environment (including agriculture) (World Meteorological Organization [WMO], 1979). In 1988, the Intergovernmental Panel on Climate Change (IPCC) was set up by the WMO and the United Nations Environment Programme to be a trusted source of scientific advice for Governments across the globe to help create better climate-related policies. The IPCC produces several types of report, including national level GHG accounting methodology *Guidelines for National Greenhouse Gas Inventories* (IPCC, 2006; 2019a), Assessment Reports that provide updates on the direction of climate science, global climate impacts and mitigation opportunities (e.g., IPCC, 2021; 2022) and Special Reports, such as *Global Warming of 1.5°C* (IPCC, 2018). The Earth Summit of 1992 took place in Rio de Janeiro where the United Nations Framework Convention on Climate Change (UNFCCC) was signed by 166 Parties (countries) with the main aim of the UNFCCC being to respond to the increasing impacts of climate change (Kuyper et al., 2018). A few years later in 1997, the Kyoto Protocol was adopted to implement the Convention and support countries signed up to pursue and enforce measures to reduce and limit further GHG emissions (UNFCCC, 1997). In 2015, 196 Parties signed up to the Paris Agreement which aimed to keep global temperature rise well below 2°C and ideally limit any further temperature increase to 1.5°C (UNFCCC, 2015; Roe et al., 2019). 2019 was a critical year for setting ambitious climate targets around net zero GHG emissions (explored in Section 2.4.2) and climate neutrality (e.g., European Union climate neutral by 2050 target; European Commission, 2019). To tackle global methane emissions, including those resulting from agricultural activity, the Global Methane Pledge was initiated at the United Nations Conference of the Parties in Glasgow in 2021 (known as COP26). A 30% reduction in CH_4 emissions by 2030 (relative to 2020 emissions) target was agreed to by 158 Parties that represent around half of CH_4 emissions globally (United Nations Environment Programme, 2021; Malley et al., 2023).

2.4.2 UK policy

Under the advice of the Committee on Climate Change (CCC), the UK Government agreed to achieve a target of ‘net zero’ GHG emissions by 2050 (UK Government, 2019; Hale et al., 2022), which would deliver the UK’s contribution to the Paris Agreement, playing an important role in keeping global temperature rise well below 2°C (CCC, 2019; Roe et al., 2019). However, the term ‘net zero’ is fairly novel and has been used interchangeably with other terms like ‘climate neutrality’ (IPCC, 2018), although, as outlined in Chapter 1, the terms have slightly different meanings. At the sector-level, the National Farmers Union (NFU) have set a target for the UK’s agriculture sector to achieve net zero GHG emissions by 2040 (NFU, 2019). The agricultural sector contributes 11% of UK GHG emissions (Defra, 2023a), but also has the unique opportunity for carbon removals from the atmosphere and mitigating further GHG losses through more efficient farming practice without damaging the environment further (CCC, 2020b). However, the idea of net zero agriculture is still early, driving the need for research into this area to determine if net zero is actually possible and how future climate change factors may impact the resilience of this system.

There is understandably some uncertainty amongst the farming community about the resilience and performance of their farms particularly without basic payments and the creation of new initiatives since Brexit (Tyllianakis et al., 2023). The Environmental Land Management (ELM) schemes and underlying Sustainable Farming Incentive (SFI) have largely replaced the European Common Agricultural Policy (CAP), which used to provide £3 billion from the European Union budget (Hurley et al., 2020). CAP was centred around incentivising farmers to adopt ‘greener’ environmental farm practices, e.g. hedge and wetland maintenance (Helm, 2017). This new framework of ELMs has been designed to continue to support UK farmers in food production and will offer a “public money for public goods” approach for those that can offer environmental and animal welfare benefits (DEFRA, 2018). However, farmer engagement research has shown younger, more receptive farmers may be more willing to sign up to these new contracts as long as the remuneration isn’t too different to before (Tyllianakis et al., 2023).

2.5 Strategies for GHG mitigation

In order to contribute to national and sectoral net zero ambitions and reduce the environmental impact of agricultural activities, it is essential that the agriculture sector mitigates its GHG footprint. Many farmers in England are already taking action to reduce their GHG emissions (Chapter 1), although there is still a deficit, and further efforts are needed to meet climate change policies.

There are three overarching approaches to GHG mitigation in the literature: reductions, removals, and displacement of emissions. The first approach of reducing emissions requires hotspot activities to be identified and efforts made to reduce the CO_2, N_2O and CH_4 emissions (Smith et al., 2008). The second approach involves enhancing soil carbon sequestration through the increase in farm vegetation and improving soil health to enhance the soil carbon stock (Smith, 2004; Lal et al., 2018). The last approach focusses more on avoiding GHG emissions through the use of cleaner and renewable energy sources to displace fossil fuel usage, such as solar, wind and bioenergy (Brack and King, 2020; Styles et al., 2015). For the purpose of this chapter, and owing to the scope of this thesis, only reductions and removals are discussed in the following sections as they are most relevant to the research questions.

2.5.1 Reducing GHG emissions

As a key question of this thesis centres around how agriculture can reduce its contribution towards climate change, the following sections outline some of the key strategies available to farmers and landowners to reduce their farm's GHG footprint, broken down by practices suitable to arable production and livestock production.

2.5.1.1 Arable production

The most effective practices to reduce GHG emissions in arable systems centre around reducing direct and indirect N_2O emissions as these represent the largest sources of non-livestock emissions in the UK GHG Inventory at 18.4% and 4.5% of agricultural emissions, respectively (Brown et al., 2023). As mentioned earlier in this chapter, N_2O emissions in arable systems primarily stem from nitrogen-based fertilisers that are applied to soils, making nitrogen (N) management a key lever in crop production .

One immediate opportunity to reduce the N_2O footprint from cultivations is to either use a lower N concentration fertiliser or ensure that the application of the nutrient is only what is required by the crop, thereby reducing N losses (Snyder et al., 2014; Guo et al., 2022).

It is estimated that approximately 1% of manufactured nitrogen fertiliser applied to soil is emitted as N_2O, although this is a global default value, with actual N_2O emissions varying by soil and climate type (IPCC, 2006; IPCC, 2019a), and some research has suggested that this might be an overestimate of N_2O emissions (Sylvester-Bradley et al., 2015). For a UK average tillage crop application of 130 kg N ha^{-1} (Defra, 2022b), roughly 1.3 kg N ha^{-1} is released directly as N_2O into the atmosphere, which corresponds to a warming equivalent of 355 kg CO_2e ha^{-1} using a 100-year timeframe (GWP_{100}). Experiments investigating the effects of different fertiliser types have observed slightly lower direct N_2O emissions from urea-based fertilisers than AN-based, however, the indirect N_2O emissions associated with ammonia volatilisation from urea balanced N_2O emissions out (Smith et al., 2012). This paper, as well as other experimental studies have also found soil moisture and temperature to be important factors affecting N_2O production (Snyder et al., 2009; Smith et al., 2012; Bell et al., 2015). Both of these findings indicate that to reduce N_2O emissions from manufactured N applications, the type of fertiliser and the conditions in which it is applied are essential to get right.

Manures and slurries are produced abundantly in livestock and mixed farming systems and are frequently used to supplement or offset synthetic fertiliser requirements on croplands and grassland. Organic materials contain significantly less nitrogen than their synthetic counterpart, for example cattle farmyard manure (FYM) at 25% dry matter contains 6 kg N per tonne (AHDB, 2021), whereas the same amount of AN fertiliser would contain 345 kg N. Therefore, it is unlikely that land managers that rely heavily on synthetic fertilisers would be able to make a complete switch to organic materials as much larger quantities of manure would be required to make up the nutrient deficit. However, the co-benefit of introducing organic materials to farm soils is the increase in soil carbon, which improves the overall condition and health of the soil in terms of porosity, drainage, and nutrient availability. This can result in longer term reduced requirements for synthetic inputs as soil retention and distribution of N is linked with higher soil carbon contents (Kibblewhite et al., 2008; Lal, 2016; Minasny et al., 2017).

A great deal of previous research has focused on the benefits of precision agriculture (PA) on GHG emissions (Stavi and Lal, 2013; Balafoutis et al., 2017; Soto et al., 2019), as well as wider environmental impacts (e.g., biodiversity and pollution) (Oliver et al., 2013; Duhan et al., 2017). PA is the concept of increasing farming efficiency and precision through instrumentation that can monitor and implement site-specific changes in management. It has been around since the mid-1980s, although has received more technological advancement in recent years (Gebbers and Adamchuk, 2010). The technology behind PA allows farmers to increase the efficiency of fertiliser, water, and agrochemical application in fields, thereby avoiding unnecessary emissions, wastage or leaching into soils or water sources. This is done through the use of sensors, satellite imagery and global positioning system (GPS) data to regulate where and when these inputs are applied based on weather forecasts, soil and air moisture content and canopy cover (Gebbers and Adamchuk, 2010; Godfray et al., 2010). The UK has a temperate climate, so typically receives enough rainfall to yield crops and maintain grassland areas, however under a changing climate the frequency of precipitation may change (IPCC, 2018). Precision technology will be useful for monitoring soil moisture content and configuring irrigation to coincide with future drier periods. Conversely, if precipitation frequency increases there will be a higher chance of soils becoming waterlogged, which releases N_2O emissions (Lal and Stewart, 2010), and PA technology will help prevent this.

As well as contributing to direct and indirect N_2O emissions upon soil application, manufactured nitrogen also has an embedded footprint (CO_2 and N_2O) associated with the manufacture process (Brentrup et al., 2018; Menegat et al., 2022). Whilst Brentrup et al. (2018) observed that the emissions associated with the production of fertiliser in Europe have decreased by 60% (1990s to 2014) due to increased use of renewable energy in the fertiliser production process and nitric acid abatement technology, other production regions are still producing AN fertiliser with three-times the carbon footprint of Europe (Brentrup et al., 2018).

Nitrification inhibitors (NIs) and urease inhibitors (UIs) are chemical compounds that are applied to N-based fertilisers to reduce N_2O emissions and increase the Nitrogen Use Efficiency (NUE) in cropping systems. The mechanism behind NIs and UIs involves delaying the process in which soil bacteria oxidise ammonium (NH_4^+) into nitrite (NO_2^-) and hydrolyse urea, respectively (Abalos et al., 2014). In the case of NIs, reducing the amount of NO_2^- in the soil means that there is less available NO_2^- for denitrification to occur, which further reduces the possibility of N_2O emissions (Lam et al., 2017). Two common NI products are dicyandiamide (DCD) and 3, 4-dimethylpyrazole phosphate (DMPP). Meta-analyses have determined that NIs can achieve a 31-48% reduction in N_2O emissions, particularly in uplands, grasslands and rice paddies (Qiao et al., 2015). In the UK, Rothamsted Research conducted field trials to test the effects of DCD and DMPP NIs on N_2O emissions when applied with N fertiliser, cattle slurry and urine (Misselbrook et al., 2014). The authors found that DCD effectively reduced N fertiliser and cattle urine sourced direct N_2O emissions by 39% and 69% for AN and urea fertilisers, respectively, and 70% for cattle urine. Whilst NI effects on cattle slurry were lower, an overall mean direct N_2O reduction of 56% was found across the treatments (Misselbrook et al., 2014).

An increasingly popular option for mitigating GHG emissions, whilst simultaneously improving soil health and nutrient availability through improving carbon storage and nitrogen availability is the planting of cover crops (Kaye and Quemada, 2017). Cover crops can be legumes or non-legumes, with leguminous plants being able to fix nitrogen directly from the atmosphere (Tribouillois et al., 2016; Kaye and Quemada, 2017). This is an advantage in modern agriculture and ties in with the idea of reducing synthetic N fertiliser usage, as leguminous cover crops typically fix and store enough N in the root zone to significantly reduce synthetic nutrient requirements. Examples of cover crops include clovers (legumes), vetches (legumes), turnips (non-legumes) and triticale (non-legumes). Some combinable crops can also be considered cover crops if being used to keep the soil covered between main cash crops, for example winter wheat, oats, spring wheat and rapeseed (Kaye and Quemada, 2017; Büchi et al., 2018; Abdalla et al., 2019; Storr et al., 2019). However, a trade-off with using N-fixing cover crops is that the additional N entering the N cycle will cause the release of some N_2O through nitrification and denitrification (Chapter 2).

Other common cropland-based practices for GHG mitigation include the retention and incorporation of crop residues and changes in cultivation to reduce soil inversion. Crop residues are a by-product of cash crop production, where the main grains and oilseeds are harvested and the stems, leaves and roots are left behind on the field. Crop residues have multiple uses, however, as they are often removed from the field as biofuel feedstock, livestock feed and bedding (Glithero et al., 2013; Ma et al., 2019).

Farm machinery often runs on fossil fuels, which releases CO_2 into the atmosphere (CCC, 2020a). Reduced and no till management options are increasingly used in agricultural systems worldwide, with the main benefit being reduced soil disturbance and thus limited soil carbon losses from soils and reduced fuel combustion CO_2 emissions, although this practice requires the use of herbicides, such as glyphosate (Lal, 2004; Smith et al., 2008; Lal, 2010). Furthermore, passes of each type of machinery, e.g., for ploughing, spreading muck and fertiliser, in-field can result in soil compaction, reducing the aeration and movement of water into the soil depths with the possibility of causing waterlogging. Therefore, use of machinery also indirectly influences N_2O release from the soil. Reducing hours of machinery in-field and adopting strategies such as controlled traffic farming, where all machinery can follow a set track to avoid soil compaction in productive soil areas (Antille et al., 2015), can increase farming efficiency and reduce GHG emissions.

2.5.1.2 Livestock production

With the main source of methane in agricultural systems being ruminant livestock enteric emissions (Thorpe, 2009), much of the mitigation research has focussed on altering livestock diet, improving productivity and genetics to reduce CH_4 emissions. The metric behind the efficacy of CH_4 mitigation changes from study to study, with common metrics being total CH_4 production (i.e., grams of CH_4 per day [g/d]), the yield of CH_4 (g CH_4 per kg Dry Matter Intake [g kg DMI^{-1}]) and CH_4 intensity (g kg $product^{-1}$) (Beauchemin et al., 2022). For the purposes of this chapter, only reductions in CH_4 production will be discussed in depth as this is most commonly reported.

Livestock feed plays an important role in the production of CH_4 during the anaerobic fermentation process that occurs in the rumen of ruminant livestock animals, e.g., cattle and sheep. There is growing evidence suggesting that manipulations to livestock feed digestibility, protein and fat content or the addition of concentrates, inhibitors or novel additives can reduce CH_4 production whilst maintaining productivity (Hristov et al., 2013; Arndt et al., 2021; Beauchemin et al., 2022). Methane production provides a general estimation of emissions that can be used in this scenario analysis; however, the other metrics may be more useful when looking at individual farm sites and product supply chains. In some cases, CH_4 yield may decrease at the expense of total CH_4 production increasing. For example, increasing the quantity of feed intake has been linked to higher absolute CH_4 production due to the additional emissions associated with the higher quantity fed, but lower CH_4 yields per unit of product due to the lower microbial action from quicker passage rates (Beauchemin et al., 2022).

A recent meta-analysis ranked dietary and management mitigation strategies for intensively housed and grazing ruminants based on absolute reductions in CH_4 without compromising productivity, reporting that the addition oilseeds to lactating animal diets could reduce daily CH_4 emissions by 20%. Feed additives of oils and fats, or using tanniferous forages (high tannin content) could also reduce daily CH_4 production by 19% and 12%, respectively (Arndt et al., 2022). A key advantage of high tannin forage is that it can be applied in both housed and grazed systems. Introducing more fats into the diet, such as oils, has been linked with the suppression of the bacteria that cause CH_4 production (Hristov et al., 2013). Recent research looking into the effects of adding canola oil into the diets of beef cattle demonstrated a 24% decline in CH_4 production (Gruninger et al., 2022). However, some authors have suggested that the literature on this topic has only provided evidence for *in vitro* applications of oils, particularly essential oils, on the inhibition of CH_4 production, and little evidence on *in vivo* applications exists (Benchaar and Greathead, 2011).

Furthermore, other researchers have revealed opportunities to combine dietary manipulation practices to decrease total CH_4 emissions further. Bayat et al. (2017)

observed additive effects with the replacement of grass silage with a high concentrate diet and the addition of sunflower oil, resulting in the lowest CH_4 production (335 g/d) compared to low concentrate only (492 g/d), high concentrate only (404 g/d) and low concentrate with sunflower oil (362 g/d). In the same vein, other dietary lipids have also been observed to reduce total CH_4 production with Arndt et al. (2022) reporting an overall decline in daily CH_4 emissions from vegetable oil addition to livestock diet in their meta-analysis.

Dietary additives are also a popular area of research for the mitigation of enteric emissions, with compounds such as 3-nitrooxypropanol (3-NOP), chloroform and bromochloromethane (BCM) amongst the more successful findings when tested *in vivo* (Hristov et al., 2013). 3-NOP has shown consistent promise for reducing enteric CH_4 emissions although most studies have been performed on cattle (mainly dairy) (Martínez-Fernández et al., 2014; Dijkstra et al., 2018; Reisinger et al., 2021; Gruninger et al., 2022). 3-NOP is a highly soluble compound, which means that it's efficacy is perhaps only relevant in intensively housed cattle systems where feed ration is highly controlled and constant. The recent meta-analysis from Arndt et al. (2022) suggests daily CH_4 reductions of 35% and when tested with sheep CH_4 emissions were reduced by up to 29% at the highest dose (Martínez-Fernández et al., 2014). When 3-NOP was used in combination with canola oil in the diet, a 51.3% reduction in beef cattle CH_4 production was observed due to both compounds having distinct mechanisms for targeting methanogenesis in the rumen (Gruninger et al., 2022). Seaweed supplementation (e.g., *Asparagopsis taxiformis*) has also presented another pathway to reducing enteric emissions, which could potentially inhibit daily CH_4 production by 80% as shown in beef steers (Roque et al., 2021). However, as this is still a fairly under-researched inhibitor, further evidence is required about the long term health effects of feeding seaweed, with the key ingredient bromoform, to cattle to be able to apply this practice at scale. Other lines of evidence for reducing daily CH_4 production include the addition of nitrates into livestock feed to act as electron sinks, which reduces the amount of hydrogen available to rumen bacteria for fermentation. Nitrate added to dairy and beef diets resulted in roughly a 20% and 10% decline in daily CH_4 emissions, respectively, (Feng et al., 2020) with other evidence suggesting a 17% reduction across livestock in general (Arndt et al., 2022).

Alternatively, improved genetic selection of individuals could lower CH_4 emissions, especially in extensive production systems (Beauchemin et al., 2022). Sheep that were considered low CH_4 producers after genetic selection over 10 years were also leaner and therefore more profitable to farmers, and had smaller rumens (Rowe et al., 2019). Similar findings have been observed for cattle, with low CH_4 emitting animals converting feed to mass more efficiently. However, the authors also found CH_4 production to correlate with other economically significant characteristics of cattle (e.g., feed intake and milk production) which complicates the applicability for genetic selection of lower CH_4 producing individuals in cattle (Manzanilla-Pech et al., 2021).

Recent research has described the increased potential of slurry acidification in pig (Dalby et al., 2022) and cattle systems (Misselbrook et al., 2016), which offers promising options for reducing CH_4 and N_2O emissions from indoor production system that collect and store slurry. In laboratory batch experiments, Dalby and colleagues (2022) used organic and inorganic acids to treat pig slurry and found the most effective inhibitor of methane emissions to be nitric acid with a >99% reduction at a pH of 5.5. Cattle slurry CH_4 emissions were reduced by an average of 61% when acidified, although this is reliant on the maintenance of a low slurry pH whilst in storage, which may not be seen as practical on commercial scale farms (Misselbrook et al., 2016). A preferable option for farmers to reduce their manure management GHG emissions could be to improve manure storage, which is typically heaped outdoors making the nitrogen and carbon molecules in manure vulnerable to oxidation to N_2O and CH_4 (under methanogenic bacterial processes), respectively. A study by Hansen et al. (2006) observed N_2O losses of almost 5% in uncovered pig manure heaps and N_2O reductions of 99% when the heap was covered by an airtight material.

2.5.2 Carbon removals and storage

Carbon sequestration and storage is a key process in the net zero debate. Enhancing the removal of carbon from the atmosphere and the long-term lock up of that carbon in vegetation and soils reduces the concentration of CO_2 in the atmosphere and benefits soil health. The effectiveness of these opportunities will depend on the size or area of the vegetation and the current health of the soil, variables which are strongly influenced by financial, social, political and environmental factors (Lamb et al., 2016). In recent years, attention has been shifted towards the benefits of improving soil health as a measure to increase carbon sequestration and storage (Minasny et al., 2017; Lal et al., 2018; Amelung et al., 2020). Whilst it is possible for agriculture to play an important role in this movement, with soil health being heavily degraded from decades of intensive conventional practices (Lal, 1997; Webb et al., 2017), many scientists and stakeholders agree that efforts should first be made to mitigate GHG emissions from agriculture as sequestration and storage of carbon from CO_2 has its limitations (Powlson et al., 2011; CCC, 2018a; Poulton et al., 2018). For instance, the 4 per 1000 initiative (Minasny et al., 2017) is a global effort to increase soil carbon stocks by 0.4% per year in the top 30-40 cm soil, which would limit the rise in atmospheric CO_2 concentration. Whilst some researchers have supported this ambition (Rumpel et al., 2018; Soussana et al., 2019), others have outlined its caveats and limitations, such as GHG leakage, feasibility without policy support and a lack of resources to farmers (see Poulton et al., 2018). GHG leakage is the increase in emissions elsewhere when efforts are made to reduce or remove GHG emissions within a system or project (Oldfield et al., 2021). In the context of 4 per 1000, where agricultural land could be put into forestry or reducing soil disturbance through minimum or zero tillage practices, GHG leakage could occur to compensate for the reduced productivity of land in these practices by needing to grow food crops elsewhere.

Afforestation and reforestation around the UK are key strategies in the Land Use: Policies for a Net Zero UK (CCC, 2019) report, with particular attention falling on the potential for agroforestry, i.e. planting trees on agricultural land. Agroforestry is likely to play a key role in meeting net zero targets by creating and maintaining woodland copses and tree lines in the farmed landscape, which is predicted to achieve GHG savings of 0.7 $MtCO_2e$/year (NFU, 2019) and up to 6 $MtCO_2e$ by 2050 (CCC, 2020a) through storage of C in above ground woody biomass. Tree plantations can also improve the health of the soil and provide a buffer against events such as flooding, which could become more frequent in some areas of the UK under future climate change (Lal et al., 2018) and a challenge for UK farmers wanting to avoid waterlogged soils and therefore excess N_2O emissions (Smith et al., 2010). In the recent Land Use: Policies for a Net Zero UK report (CCC, 2020a), it was suggested that up to 30,000 ha of trees be planted around the UK to increase carbon removals and storage. However, the feasibility of increasing agroforestry depends largely on the attitudes and behaviours of farmers, i.e. whether they are content with trading off land for food production for woodland, and whether there will be adequate policy and incentives to achieve this goal (Smith et al., 2007). The report further explains the potential for a carbon trading scheme, whereby the costs of planting and maintaining agroforestry plots could be covered by larger emitters, e.g. transport sector businesses, in return for carbon credits to those companies for their fossil fuel usage (CCC, 2020a).

In the UK, agricultural land is typically separated into fields by semi-natural features, such as hedgerows and scrub (Graham et al., 2018). The above ground woody biomass of these elements can sequester and store carbon from the atmosphere, as well as provide optimal habitat for farmland biodiversity (Falloon et al., 2004) and can be incentivised under agri-environment schemes (AES). Increased planting of hedgerows and enhancement of hedgerows through management are strategies that both the NFU and CCC discuss in their net zero reports, with estimates of ~181,000 ha of hedgerow area to be achieved by 2050 and up to 0.5 $MtCO_2e$/year in GHG savings (NFU, 2019; CCC, 2020a).

Bioenergy with Carbon Capture and Storage (BECCS) is a process in which the CO_2 that is emitted when bioenergy crops are burned for fuel or energy is captured and stored underground, rather than being released into the atmosphere (Brack and King, 2020). In 2018, 94,000 ha of farmland was used for growing bioenergy crops (Defra, 2019c). This technique is beneficial for carbon removals, but also provides biofuels for vehicles, displacing fossil fuel combustion and the associated emissions, and will provide income for biomass growers (Fajardy et al., 2019; Brack and King, 2020). However, growing bioenergy crops for CCS removes potential cropland for food production and its impacts on biodiversity have been mixed, but there are possibilities for 2nd generation biofuels from animal and crop wastage, as well the potential to grow bioenergy crops in marginal land to reduce the pressure on food production space (Fajardy et al., 2019) See a review of the environmental impacts of bioenergy crops by Wu et al. (2018) and further BECCS information in Fajardy et al. (2019).

Other potential strategies to aid in carbon sequestration, but not covered in detail in this review, include the use of biochar on soils (Smith, 2016), cover cropping (Lal et al., 2018) and increasing grassland area - i.e. grass strips and conversion of arable land to grassland (Falloon et al., 2004). The effectiveness of biochar is still being researched, so it’s feasibility as a potential net zero strategy within the next 30 years is debatable (Smith, 2016; Lal et al., 2018). Many farmers already employ cover cropping in and around fields to improve soil nutrient content and structure, to provide habitat for farmland biodiversity and importantly to facilitate carbon removal and storage in the soil (Lal et al., 2018; Dicks et al., 2019). Lastly, soil carbon can be increased by sowing grass strips in field margins (Falloon et al., 2004) and by converting unproductive arable land into temporary or permanent grassland. Although some emissions are to be expected from a change in land use type, grassland is beneficial for biodiversity and generally has a higher potential soil carbon stock than arable soils (Smith et al., 2010).

Chapter 3 - Estimating GHG emissions and modelling mitigation options for an integrated pig-arable farm

3.1 Introduction

There is growing pressure for the UK agriculture sector to identify ambitious yet feasible changes in farm practices that can deliver emission reductions in line with sectoral (National Farmers Union, 2019) and national (UK Government, 2021) net zero emission targets, and global (Verschuuren, 2016; Jacquet and Jamieson, 2016; Chan et al., 2018) efforts to minimise further global warming to well below 2°C. This is all whilst avoiding any potential reductions in productive land area and farm profitability due to producing 'net zero' food. The UK agriculture sector is responsible for 11% of the UK's total greenhouse gas (GHG) emissions, including 71% of nitrous oxide (N_2O) emissions and 49% of methane (CH_4) emissions (Defra, 2023a). Where other economic sectors are able to decarbonise through the use of renewable energy and carbon dioxide (CO_2) capture and storage (CCS) technologies (e.g., bioenergy with carbon capture and storage [BECCS], Brack and King, 2020), agricultural systems face a unique biological challenge of tackling N_2O and CH_4 emissions in crop and livestock systems. With N_2O and CH_4 having a greater GWP_{100} compared to CO_2 (273 and 27.2, respectively; IPCC, 2021; see Chapter 2), it is important for landowners and farm managers to understand their main on-farm emission hotspots and reduce their footprint to help the sector minimise its' impact on climate change. Land based opportunities for carbon removals, such as afforestation (CCC, 2020a; Staddon et al., 2021) and improving soil carbon sequestration (Oelkers and Cole, 2008; Lal, 2010; Lal, 2020) also present themselves as a unique route for the sector to mitigate further climate change, but this is explored at regional-scale in Chapter 4.

To better understand what impact agriculture is having on climate change, an assessment of the GHG emissions produced in a typical year is needed to benchmark and monitor progress where actions are being taken to reduce the footprint. At national level, annual GHG assessments are reported for all economic sectors in the National GHG Inventory to the United Nations Framework Convention on Climate Change (UNFCCC; Brown et al., 2023). The methodology used in national accounting is mostly Intergovernmental Panel on Climate Change (IPCC) national-level guidance for estimating GHG emissions (IPCC, 2006; IPCC, 2019a), which promotes consistency with other nations and is the widely accepted scientific methodology. This methodology is also used at farm-level and product-level by farm carbon calculators (e.g., Cool Farm Tool, Farm Carbon Calculator, Agrecalc) to estimate GHG emissions from whole farms, agricultural enterprises, and products. However, national level methodologies are not always appropriate to use at finer scales as they often lack the detail required for on-farm accounting and don't account for emissions incurred upstream or downstream within the same methodology (Colomb et al., 2013; Sykes et al., 2020).

The solution to this is to use a more robust and granular methodology that accounts for the impacts of a product through its life cycle (i.e., from the acquisition of raw materials to disposal) to understand fully how much of an environmental impact it has. Life Cycle Thinking (LCT) is an institutional logic that allows users, typically researchers, stakeholders, and policymakers, to estimate and evaluate the environmental burden of a product by accounting for both direct and indirect impacts throughout the products' life cycle (Heiskanen, 2002). Life Cycle Assessment (LCA), sometimes referred to as Life Cycle Analysis, is a methodological framework for evaluating the environmental and social impacts of a product through its life cycle. Environmental impacts include burdens such as global warming potential, eutrophication, acidification, land use and energy use (Caffrey and Veal, 2013). A lifecycle includes all stages leading from the acquisition of raw materials to the manufacture, distribution, use and disposal of a good or service, including any energy emissions of pollutants to air, soil, and water. A typical lifecycle value chain would have pre-production, production, and post-production stages. In the context of agriculture, pre-production could be the extraction and mining of raw materials to manufacture fertilisers and pesticides, or the production of animal feed. Production would then be the activities that occur on farm, such as the application of that fertiliser or pesticide, and the enteric methane and manure produced due to the diet of the animal. Post-production includes the transport, processing, packaging, retailing, consumption and disposal of the product (Roy et al., 2009; Vermeulen et al., 2012; Basset-Mens et al., 2022).

Agri-food LCAs are frequently reported in the academic literature (Williams et al., 2006; Roy et al., 2009; Notarnicola et al., 2017; Poore and Nemecek, 2018) and completed by businesses like retailers, food manufacturers and food service providers to account for their supply chain environmental impacts (World Resources Institute and World Business Council for Sustainable Development, 2014; Waste & Resources Action Programme, 2024). LCAs can help to highlight emission hotspots in a product's lifecycle, which can then be used as target areas for GHG reduction modelling (Caffrey and Veal, 2013) and are important for understanding the net zero potential of the food system.

3.1.1 Aims of this chapter

The aim of this chapter was to assess the main GHG emission sources associated with the production of arable crops and pigs and model options for reducing those emissions, using the UoL farm as a commercial-scale case study farming system, representative of arable-pig farming in England and Wales. The UoL farm presents a useful case study for assessing the global warming impacts of some of the most widely consumed farm products globally, namely pork (Ottosen et al., 2021) and cereals and oilseeds (FAO, 2023a).

This chapter consists of a baseline assessment of GHG emissions from wheat, barley, oilseed rape, grass silage (2015 to 2018) and pig liveweight (2016 – 2018) using LCA methodology, which is the first to be done for the UoL farm. Using the results of the LCA, two scenarios were modelled and presented to assess the efficacy of reducing the dominant emissions from arable and pig production using nitrification inhibitors (NIs) and slurry acidification, respectively. See Section 3.2.6 for details on these practices.

3.2 Methodology

3.2.1 Study site

The University of Leeds (UoL) farm is a 317 hectare commercial-scale research mixed livestock-arable farm near Tadcaster, United Kingdom (Lat: 53.861239, Long: -1.344573). This site was chosen as it is a commercial-scale farm allowing some representability to other pig, arable and integrated pig-arable farming systems in England and generally has high spatial and temporal data quality compared to non-research farms. This improves the applicability of the findings to other similar systems compared to studies that have used solely secondary data.

Arable & grassland

The farm is owned and operated by the University, however, fields are contracted out to three companies, which will remain anonymous as it does not impact the findings of this chapter. Each company handles either combinable crops, vining peas, or potatoes. Only data for combinable crops was available for this study.

Roughly three quarters of the land is used for arable farming, with the main combinable crops being winter wheat (*Triticum aestivum*), spring and winter barley (*Hordeum vulgare*), and oilseed rape (OSR; *Brassica napus*). Between 2015 and 2018, an average of 102.8 (Standard Error [SE]: ± 8.1) ha of wheat, 38.7 (± 9.1) ha of barley and 37.9 (± 6.7) ha OSR was grown. An average of 21.9 (± 4.8) ha of potatoes and 26.3 (± 4.9) ha of vining peas was grown, although no potatoes were grown in 2016. This represents a much smaller area (21%) of unknown crop emissions from the arable enterprise (i.e., potatoes and vining peas) compared to combinable crop area emissions (79%).

The UoL farm also had an average of 30.9 (± 1.7) ha of rotational and permanent pasture (PP) between 2015 and 2018. The farm management company indicated that at least 4 ha of grassland is cut for silage each year in general, which is then baled and sold off-farm. The grassland has been grazed by sheep from another business for many years.

Pig Unit

The farm is also home to the Centre for Innovation Excellence in Livestock (CIEL) National Pig Centre, herein referred to as the pig unit. This is the UK's largest pig research facility, with common research themes including pig nutrition, behaviour, welfare, productivity, and health. The pig unit is comprised of an outdoor breeding herd, and an indoor breeding and feeding herd at commercial scale. The outdoor pigs are bought in or born and reared outside only, moving between several fields on the farm every 2-3 years to allow the land to recover. This herd is used for breeding purposes, with piglets being sold after weaning. The stocking rate is 2 sows per hectare. During farrowing and weaning, a moveable fence is erected to mark off 10 m by 15 m plots of land for each sow and her litter, which also includes an arc, the pig shelter. Once the piglets have been sold off, the sows are either retained for further servicing or are sold for slaughter. The outdoor herd was set up in 2016, part way through the proposed baseline period of this study and thus was not at full capacity in the following years. Between 2016 and 2018 the outdoor herd occupied two of the fields on-farm; one is predominantly arable and the other is a permanent pasture. The indoor herd was also only initiated in 2016 and owing to the advanced facilities of the pig unit, the capacity is much larger than the outdoor herd. Full capacity was not achieved until 2022, and so the baseline period of 2015-2018 is not reflective of a full capacity pig production unit. The population numbers are based on estimates provided by the pig unit manager using the same numbers for 2016 and 2017 and a 10% increase in both populations in 2018. Typical weights were from industry average data and estimates from the farm manager. The pig unit does record weight of animals more regularly than non-research commercial pig farms, so at full capacity better weight data records would be available.

Both the indoor and outdoor herds are fed using purchased feed from a local feed company. The feed purchased is different for the two herds, and also for different life stages. Data on feed was limited, but the pig farm manager and feed company provided some data on ingredient origins and protein content. For example, the wheat, barley, limestone, salt, rape meal and biscuit meal in feed originated in the UK. Alternatively, hipro (high protein) soya, full fat soya and soya oil originated from either Argentina, the United States or Brazil. Protein content of feed ranged between 13% and 19.5%. The bedding used by the outdoor herd (in the arcs) and indoor herd is mostly purchased straw (2016 and 2017), with some farm-produced wheat straw being used in 2018 to supplement the purchased straw.

Outdoor pigs will excrete primarily onto the ground in their field, and some will be mixed in with the bedding in the arcs. These are scraped after each batch of piglets has weaned and been sold (4 to 8 weeks). The indoor sows and gilts are housed on straw in a barn until they reach the end of gestation, at which point they are moved into the farrowing unit. The farrowing unit and finishing sheds are fitted with a slatted floor slurry capture system, which is emptied and flushed when the batch of pigs reach the end of that life stage and move into another room or are finished. This is estimated to be up to 16 weeks, particularly for the finishing stage.

Exclusions

Whilst potatoes and vining peas are grown in several of the fields year-to-year, data is unavailable for these plots. Therefore, for the rest of this chapter the methodology and results shall discuss only the combinable crops. The UoL farm has a longstanding partnership with the National Institute of Agricultural Botany (NIAB), who have run trials in several fields as both short term and long term experiments over the years of farm operation. Between 2015 and 2018, NIAB ran trials across three of the arable fields on-farm. However, the input data for these trials is unknown and has not been provided to the farm for analysis. Therefore, these trial fields in these years must be excluded from the main analysis.

It is acknowledged that whilst the sheep were on-farm, their enteric fermentation and manure deposition emissions would be part of the inventory of the UoL farm (*GHG Protocol Agricultural guidance* (WRI & WBCSD, 2014). However, the population and weight data of the sheep were unavailable for the baseline period discussed in this chapter. As the grass is harvested for silage and exported off-farm only, it is assumed that any benefit and associated emissions relating to the sheep depositing on the grass whilst grazing is carried off-farm upon sale of the silage and outside of the boundary and scope of this chapter.

3.2.2 Life Cycle Assessment

There are numerous methods for assessing the environmental impact of agricultural and land-use activities, however, in order to fully comprehend the intensity of emissions from a product of interest across both time and space, it is important to consider emissions and pollutants from the whole life cycle of that product. LCA is a well-recognised and regulated technique, with several protocols and guidelines to help users perform LCAs (e.g.,ISO 14040 and ISO 14044; 2006a; 2006b). The framework is iterative, consisting of four main stages that should be consistently checked and revisited until the LCA has been interpreted fully.

3.2.2.1 Goal and scope definition

The first stage aims to define the spatio-temporal boundaries of the assessment and the functional unit (FU) for referencing, as well as identify the main goal(s) and aims, potential assumptions and quality of the data being used. By defining these factors, the basic LCA method can be established to ensure transparency, consistency, and robustness in the following stages (Caffrey and Veal, 2013).

Aspects of the LCA goal that should be considered include the principal intentions of conducting the assessment, application of the findings and the audience who may have interests in the LCA. There are numerous requirements that should be outlined in the LCA scope; critical parameters to define include: FU, system boundary, Life Cycle Impact Assessment (LCIA) chosen, data quality and if an allocation procedure is to be used. The FU is the measurable value (e.g., production of 1 kg grain) related to the system function (e.g., food production) and can be a complex problem for agricultural LCAs. In systems with multiple output products, an allocation procedure may be used to differentiate environmental impacts between products, e.g., by economic value or weight. Both spatial and temporal system boundaries should be established during scope definition, as these hypothetical boundaries determine what is included in the Life Cycle Inventory (LCI).

Full LCAs are often cradle-to-grave, indicating a full product life cycle from the raw materials and energy used to make the product through to the disposal or consumption of it. However, partial LCAs can include gate-to-gate and cradle-to-gate assessments, where production-only or upstream and production emissions are accounted for, respectively. In some cases, a partial LCA may be more appropriate than a full one, for instance when the post-production processes are not decided by the production-phase stakeholder and will therefore be accounted for in the downstream LCA if one is completed. LCA time frame is also key as some systems produce products that extend past the normal calendar year or have varying time frames depending on other factors, e.g., agricultural production of food crops or ground cover vegetation.

Although conducted in the third stage of the LCA, the LCIA methodology should be outlined in the scope and will largely depend on the main goals and FU of the assessment. Other scope-bound factors include the data quality and assumptions made about the dataset(s); LCA data is often split between foreground (primary) data and background data, so these sources should be listed, and assumptions justified.

3.2.2.2 Life Cycle Inventory

Collection and collation of all system inputs and outputs relevant to the FU and system boundary, and analysis of those values is the LCI stage. Typical inputs are resources, e.g., energy and raw materials from the air, water or ground, and outputs are emissions, e.g., GHGs to the atmosphere, water, or soil.

Inventory data can come from foreground and background sources. Foreground data usually consists of primary data from historic records that have been collected by the inventory compiler, whereas background data is often from upstream or downstream data sources that are average or estimate data. Where the LCA practitioner is unable to find adequate primary data for the LCA, an LCI database can be used to supplement the analysis. LCI databases contain processes for pre-assessed products and services, which are usually from scientific literature or industry research. For example, Ecoinvent, ELCD (European reference Life Cycle Database) and Gabi are all well established and widely used generic databases, whereas Agri-Footprint and Agribalyse are most commonly used for agricultural footprints.

Output data, i.e., emissions, are either calculated manually by the compiler or are calculated during the LCIA depending on what emissions arise from background sources and the LCIA method chosen. If being calculated manually, the GHG Protocol typically recommends using information from the IPCC *Guidelines for National Greenhouse Gas Inventories* documents (IPCC, 2006; World Resources Institute and World Business Council for Sustainable Development, 2011; IPCC, 2019a).

3.2.2.3 Life Cycle Impact Assessment

After compiling the inventory of inputs and outputs, these values can then go through the LCIA whereby pollutants and gas emissions can be converted into an impact category, or multiple categories, using impact factors. This stage has two compulsory steps and other additional steps that can be used if the LCA scope allows for it. Firstly, inventory outputs are classified depending on the impact assessment method of choice, so only the relevant gases and pollutants are selected for analysis. Secondly, these classified outputs are characterised by assigning them an impact factor relating to the impact category. Using global warming as an impact assessment method example, the main GHGs selected would be CO_2, N_2O and CH_4 and the characterisation or impact factor applied could be the GWP_{100} values used in the latest IPCC Assessment Report, i.e., IPCC AR6 (IPCC, 2021). Other LCAs, including those relating to agriculture, have assessed the impact of global warming using alternative metrics to GWP_{100} including the GWP* metric mentioned in Chapter 2 (Section 2.1.2) and Global Temperature change Potential (GTP). GTP estimates the expected temperature change caused by the emission of a GHG for a specific point in time relative to CO_2 (IPCC, 2021). McAuliffe et al. (2023) suggest that LCAs of agricultural products and food systems should consider using metrics like GTP or GWP* to better account for the actual climate warming impact of GHGs, and in particular the nuance of SLCP's like CH_4 to make more informed decisions for agricultural practices. Alternatively, the Greenhouse Gas Protocol (GHGP) Product Standard (World Resources Institute and World Business Council for Sustainable Development, 2011) encourages businesses conducting product carbon footprints to use the GWP_{100} metric. However, this protocol has not been updated since 2011, so it is conceivable that this recommendation will be changed upon its next update.

In this chapter, the assessment is only for global warming although it is acknowledged that other environmental impact categories, such as eutrophication, energy use and land use change, would all be useful to measure. The analysis only focusses on global warming impact due to the research question being centred around GHG emissions and climate change.

Other steps in LCIA that can be used are normalisation, grouping and weighting, although these are typically preferred when trying to assess the importance of one impact category over others or when performing an endpoint analysis. Normalisation is used to check for data inconsistencies and for “calculating the magnitude of category indicator results relative to reference information” (ISO, 2006b).

3.2.2.4 Interpretation

Once the impact assessment has been performed, the results can be interpreted, and separate sensitivity and uncertainty analyses can be completed to lend support to the reliability and robustness of the findings. For instance, high, average, and low value models can be generated for some of the key hotspot activities to assess the sensitivity of emissions intensity between the groups. Some databases come with their own uncertainty estimates, such as Ecoinvent, although it is important for LCA users to perform some sort of Quality Assurance and Quality Control (QA/QC) on the data entered into the inventory regardless. This could be by getting another LCA practitioner or data provider to check over the inventory or by performing Monte Carlo simulations on the dataset to generate probability distributions.

3.2.3 Challenges of conducting LCA in the food system

Crop and livestock production is a challenging system to fully account for the environmental impacts of the activities associated with production. This is because of the highly variable and biologically driven management decisions that farmers make in order to maximise productivity. For example, the type of crop being sown should be different year-to-year to avoid depleting soil nutrient resources and for pest and weed management (Liebman and Dyck, 1993). This, however, results in varying soil amendments and operational requirements each year to manage the growth of each crop, which results in different emission sources and quantities. Therefore, some LCA's studying the impacts of crop production have moved towards a rotational approach to capture the annual variation in agricultural activities (Basset-Mens et al., 2022).

Other challenges with agricultural LCA involve the FU and where to set the system boundary of the study. Agriculture produces a wide range of commodities, such as cereals and grains, fruit and vegetables, meat, milk, eggs, nuts, coffee and more. It is thus very difficult to set a meaningful FU when comparing commodities and is normally set as a unit of mass (i.e., emissions per tonne of product). This detracts from the actual function of agricultural production, which is to provide nutritious food for humans and animals. Furthermore, it is challenging to determine what inputs and processes to include within the LCA system boundary for agricultural products, as the diesel combusted during use of the tractor is a direct responsibility of the farmer, but the responsibility for the emissions associated with the raw materials used to manufacture the tractor itself are less certain (Jolliet et al., 2016; Basset-Mens et al., 2022).

3.2.4 Applying LCA to the University of Leeds farm

Goal and scope

The overarching goal of this chapter and the use of LCA was to assess the global warming impact of a pig-arable integrated farming system and model farm management practices that could reduce GHG emissions to help the UoL farm become more aligned with its net zero ambitions. To be able to meet this goal, separate assessments were made of the main crop types grown on the farm and the two pig herds in the pig unit. The FU for the arable LCA was emissions in kg CO_2e per tonne of product, whereas the FU for the pig LCA was emissions (kg CO_2e) per kg of live weight. The FU for pigs was chosen as no records were available on carcass weight or pig meat weight.

The LCA was done as a partial cradle to farm-gate analysis to account for upstream and on-farm emission sources. Several methodologies were used in tandem to be able to account more accurately for data that was high quality (through IPCC calculations) and data that was sparse and required background EcoInvent inventory data (in openLCA). These two inventories are illustrated in Figure 3.1. The largest emission sources on-farm for the arable operation are soil application emissions and with high spatial and temporal quality primary data available at field-level, these calculations were made using the IPCC's *Guidelines for National Greenhouse Gas Inventories* (IPCC, 2006; 2019a) in an

Excel spreadsheet. Although primary data was not available on diesel consumption on-farm by crop type, it was possible to estimate this using fuel consumption rates by operation type data from Defra Project SCF0104 (Gooday et al., 2015) and the latest GHG Conversion Factors (DESNZ, 2023). Electricity consumption was not considered in this assessment, primarily because there are no meter readings available before 2019. Electricity usage is often minimal in arable systems if there is no grain drying on-site, which is true for the UoL farm, but in the pig unit electricity would be used for the lighting, heating, and ventilation.

Upstream data was accounted for using mostly Ecoinvent databases (Wernet et al., 2016) in openLCA, primarily because most farms would not have adequate records of this information. This included fertiliser manufacturing emissions, seed production, pesticide manufacture, and machinery manufacture and depreciation. There was limited data on pig feed, therefore, a UK feed emission factor for a conventional pig feed diet was used from Kool et al. (2009). To calculate the amount of food consumed by the pigs, and thus the total emissions from feed, industry average data on daily pig feed rations were taken from the AHDB *Nutrition Guide for Pigs* (AHDB, 2023b). See Figure 3.2 for a schematic diagram of the system boundary.

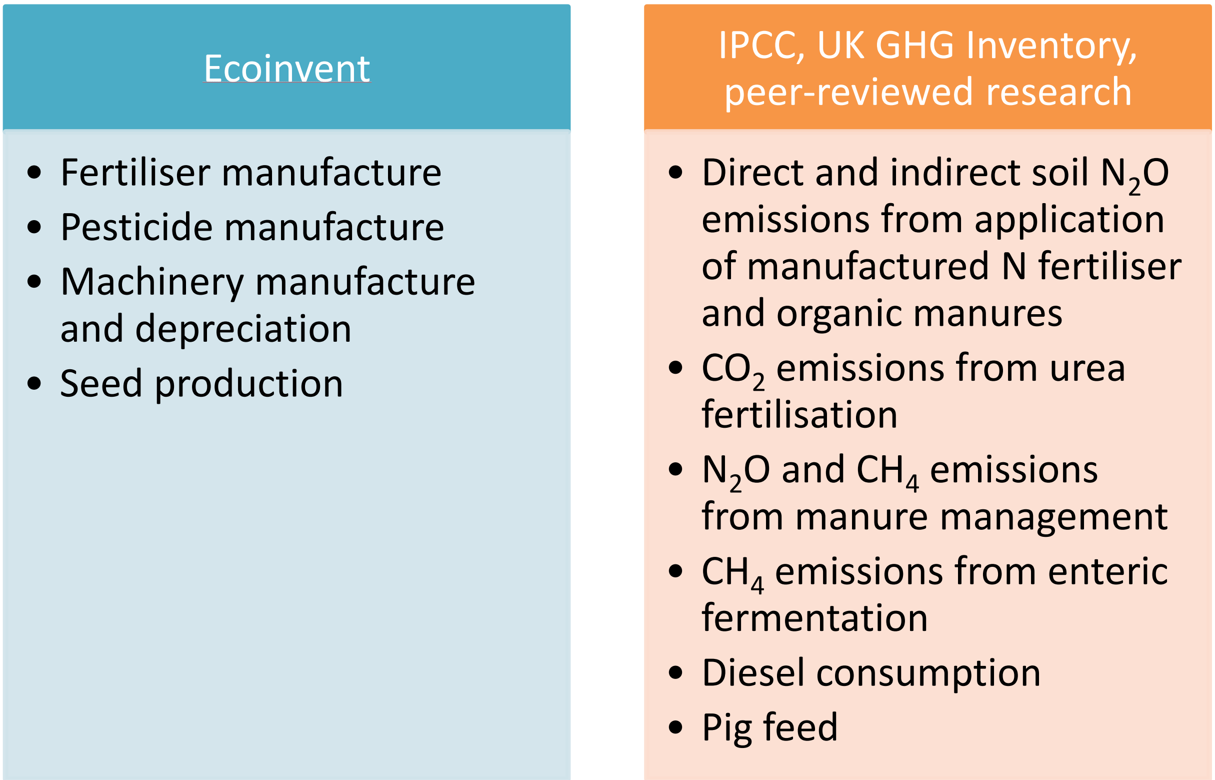

Figure 3.1 Emission sources estimated for the LCA. LCA calculations were performed using Ecoinvent and IPCC, UK GHG Inventory, and peer-reviewed research.

The UoL farm's pig unit was not finished until 2016, so no outdoor or indoor pig herd were present on the farm in 2015. The pig unit was not at capacity between the baseline years of 2016 and 2018, as this was only achieved fully in 2022. Therefore, baseline emissions from the pig unit are just a snapshot of the emissions at the time and are not a reliable benchmark for mitigation thus for the purpose of this chapter the mitigation options presented are for a full capacity pig unit only. During the baseline period, the outdoor pig herd occupied Poppy Field. After 2018, the outdoor pig herd was moved to a different predominantly arable field and thus any benefits the pig excreta provided the soil of Poppy Field for the arable crop from 2019 onwards (in terms of N inputs) would need to be accounted for in 2019's assessment. As this chapter is only dealing with the 2015 to 2018 period, this allocation is not required.

Due to the fact that most of the commodities produced or co-produced on-farm were sold off-site, it was assumed that at the point of sale the emissions from the downstream activities involving these products or co-products were out of scope. The main products exported as sales were wheat, barley and OSR grain, grass silage, and pig meat from the indoor herd and outdoor piglets. Co-products that were exported off-farm were wheat and barley straw (although this differed between 2015-2017 and 2018).

3.2.4.1 Economic allocation of straw

No official records for straw sales were available between 2015 and 2017 and considering the pig unit was not properly functional at this time; it is assumed that all straw was baled and sold off-farm between 2015 and 2017. However, in 2018 the pig unit retained 57% of the baled straw (approx. 727 bales) for bedding with the remainder being sold. Economic allocation was chosen to handle emissions of wheat production in 2018 to account for the portion of straw retained on-farm and the wheat and remaining straw sold at the farm gate (Table 3.1). OSR straw was incorporated into the soil each year and is accounted for under N_2O emissions from crop residues.

Table 3.1 Economic allocation of emissions for wheat grain (for sale) and straw (for sale and use as bedding in the pig unit) in 2018.

Commodity	Price (£/t)	Source	Economic allocation
Wheat grain	190	AHDB (2018a)	78%
Wheat straw	53	AHDB (2018b)	22%

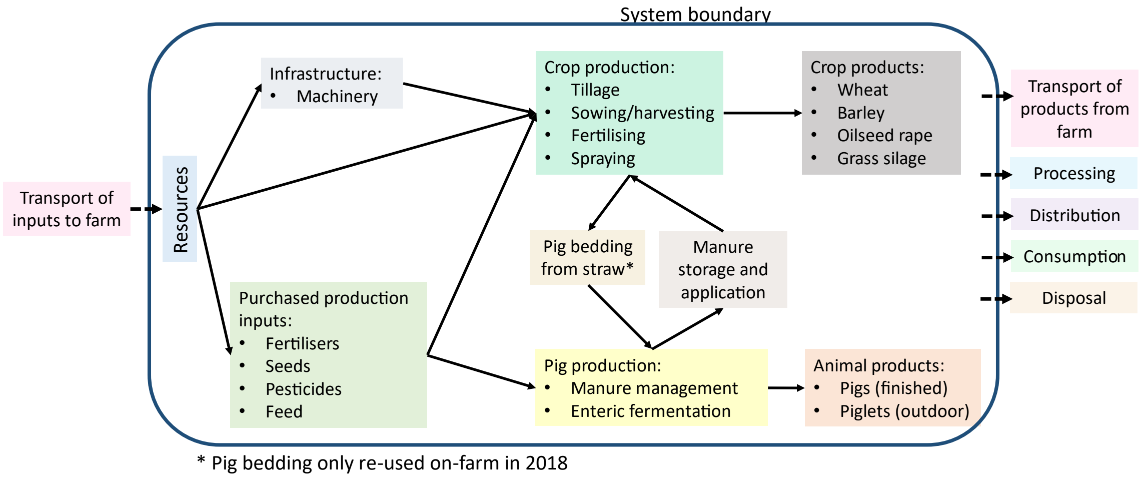

Figure 3.2 System boundary of the LCAs on the University of Leeds (UoL) farm.

Life Cycle Inventory

Data for the arable enterprise were of high spatial and temporal quality with only yield data for 2018 that had to be averaged using 2015-2017 data. Records of inputs (seed, fertilisers, and pesticides) and operations (establishment and application) were available through farm management software records in the inventory years. High quality fertiliser data meant that calculations could be done at IPCC Tier 2 level to estimate soil nitrous oxide emissions. IPCC Tier 1 and 2 methods were chosen for the pig data, which was sparse and not an accurate source of population numbers and weights but provided an estimate of enteric fermentation methane emissions and emissions from pig manure management.

Life Cycle Impact Assessment

The IPCC 2021 6th Assessment Report LCIA method was chosen for this assessment to meet the LCA goal of understanding the global warming impacts of the agricultural commodities produced at the UoL farm. This method takes the latest available 100-year period Global Warming Potential (GWP_{100}) values from AR6, which are 273 for N_2O, 27.2 for CH_4 and 1 for CO_2.

This method was used simultaneously in openLCA and through direct calculations using IPCC methods ensuring that all results are compatible.

3.2.5 Assumptions and uncertainty

As with all modelling, certain assumptions have been made to progress the LCAs given the mix of high quality temporal and spatial data, and data gaps (particularly around on-farm diesel use and upstream emissions).

The 2018 yield data was missing for all crops, so on-farm yields from 2015, 2016 and 2017 were averaged to provide an estimated yield for 2018. UoL crop yields were 14%, 37% and 11% higher than the UK average between 2015 and 2017 for wheat, barley and oilseed rape, respectively (Defra, 2024d). Therefore, using the farm average meant that the 2018 estimated yield was more representative of the higher yielding UoL farm system. Furthermore, the transport of inputs from the manufacturer to the farm were not included as the supplier data was unavailable. For a list of all of the emission sources included and excluded from the study, see Table 3.2.

In terms of methodological assumptions, the IPCC 2019 refinement calculations for Agriculture, Forestry and Other Land Use (AFOLU) were used to calculate product-level emissions, with a mixture of methodological complexity. Tier 2 calculations and emission factors (EFs; from the UK GHG Inventory) were chosen where possible to improve the accuracy of the analysis, and Tier 1 default methods and EFs where there were data gaps. For example, the soil N_2O calculations were calculated using IPCC Tier 2 methods with UK-specific EFs due to the high temporal and spatial quality of the arable data, however, Tier 1 methods were used for the pig manure management and enteric methane emissions due to a lack of good quality data.

3.2.5.1 Uncertainty and data quality

As explained in the above section, there is a mixture of high quality farm data and numerous assumptions in secondary parameter data, which introduces uncertainty into the modelling. Uncertainty can be qualified using a pedigree matrix, which is used in the Ecoinvent LCI database (Ciroth et al., 2016), and this is the approach taken in this chapter to demonstrate knowledge of uncertainty. The column 'uncertainty' in Table 3.2 gives a score for each of the following indicators: reliability, completeness, temporal correlation, geographical correlation, and further technical correlation as in Ciroth et al. (2016). A score of 1 would be representative of high quality, representative, reliable data, and a score of 5 would represent data that is non-qualified or has an unknown source.

Table 3.2 Life cycle emission sources included/not included in the UoL GHG assessment.

Life cycle stage	Emission source	Included? (Yes/No)	Details	Uncertainty*
Pre-production	Manufacture of fertilisers	Yes	EcoInvent data used in openLCA.	2;3;2;2;2
	Manufacture of pesticides	Yes	EcoInvent data used in openLCA.	2;3;3;3;2
	Manufacture of other agro-chemicals	No	Manufacture of growth regulators and adjuvants not included due to a lack of data.	
	Manufacture of machinery	Yes	Embedded emissions in the manufacture and depreciation of machinery included	2;3;3;3;2

			using EcoInvent data in openLCA.	
	Production of seed	Yes	EcoInvent data used in openLCA.	2;3;2;3;2
	Production of pigs	No	No UK pig data available in Ecoinvent and many pigs bred on-farm.	
	Pig feed	Yes	Emission factor from Kool et al. (2009) conventional England pig feed. Feeding requirements from AHDB (2023b).	4;2;3;3;3
	Pig bedding	Yes	Bedding requirements from Williams et al. (2006). Economically allocated wheat straw average emissions between 2015 and 2018 used for 2016 and 2017 bedding. Wheat straw emissions from 2018 used for 2018.	2;3;1;2;2
	Transport of inputs to farm	No	Transport amounts and distances unknown.	
On-farm production	Soil direct & indirect N_2O emissions	Yes	Raw data from farm Gatekeeper records. IPCC 2006 & 2019	2;2;1;2;2

			Tier 2 calculations and EFs from UK GHG Inventory used where possible.	
	Soil CO_2 emissions from urea fertilisation	Yes	Raw data from farm Gatekeeper records. IPCC 2006 & 2019 Tier 2 calculations and EFs from UK GHG Inventory used where possible.	2;2;1;2;2
	Enteric fermentation	Yes	Best estimate data from pig unit manager and records. IPCC 2019 Tier 1 calculations and EFs from UK GHG Inventory used where possible. Sheep enteric fermentation emissions not included due to a lack of data.	4;4;1;2;2
	Manure management	Yes	Best estimate data from pig unit manager and records. IPCC 2019 Tier 1 calculations and EFs from UK GHG Inventory used where possible. Sheep manure management emissions not	4;4;1;2;2

			included due to a lack of data.	
	Diesel consumption	Yes	Estimated using field areas per crop and default diesel consumption values by operation type from Defra Project SCF0104 (Gooday et al., 2015).	3;4;1;3;2
	Electricity consumption	No	No data for electricity readings available for the pig unit between 2015-2018. No grain drying unit on-farm for arable crops so electricity consumption likely to be low.	

*** Uncertainty categories in corresponding order: reliability**; **completeness**; **temporal correlation**; **geographical correlation**; **further technical correlation (Ciroth et al., 2016)**

3.2.6 Options for mitigating emissions

Soil N_2O emissions are often the main emission source for arable production (Rees et al., 2013; Bell et al., 2015; Cowan et al., 2020) due to the application of nitrogen fertiliser. UoL N fertiliser applications were 3.5-9.5% lower than the industry average nitrogen application per hectare across wheat, barley and oilseed rape (Defra, 2022b), so reducing the amount of nitrogen applied would not have been a feasible scenario to model. No nitrification or urease inhibitors were used on-farm 2015 – 2018, so the first scenario models the introduction of the NI dicyandiamide (DCD) to applications of Ammonium Nitrate (AN) and urea for wheat, barley and grass silage. There have been numerous studies and subsequent meta-analyses on the efficacy of DCD on different N additions (manufactured and organic); all agree that DCD reduces direct N_2O emissions and most also conclude that indirect N_2O emissions from ammonia volatilisation may increase with the use of NIs such as DCD (Misselbrook et al., 2014; Ruser and Schulz, 2015; Lam et al., 2017). The EF for AN applications chosen for this mitigation strategy was taken from Lam et al. (2017) at 27% reduction of direct N_2O (originally from Misselbrook et al., 2014) for wheat and barley production, as well as pasture. The same papers were used for a urea + DCD EF of 47% direct N_2O reduction. These reductions were applied to the direct N_2O EFs when calculating emissions for each crop. Urease inhibitors are increasingly used with urea-based fertilisers and target the ammonia volatilisation pathway leading to indirect N_2O release (Chapter 2). However, for the purposes of this chapter, only the effects of DCD were tested as this has been tested rigorously in arable systems.

For pig production, feed typically represents the largest proportion of emissions (McAuliffe et al., 2016), however, as data quality for pig feed is low in this chapter the mitigation of emissions from slurry management in the pig unit was modelled instead. Literature on the potential mitigation of CH_4 and N_2O emissions during manure storage show high variability in efficacy (see Ambrose et al., 2023 for a recent review). A widespread practice in Denmark is the acidification of pig slurry, mostly to reduce ammonia emissions, although reductions in CH_4 have been noted. The key challenges with acidification are around which acid to use, when to apply the acid and how frequently acidification is needed to maintain the emission mitigation (Fangueiro et al., 2015; Dalby et al., 2022; Vechi et al., 2022; Ambrose et al., 2023). In this chapter, a CH_4 mitigation efficiency of 91% from the acidification of pig slurry to pH 5.5 in the storage tank is used based on research from Vechi et al. (2022). This paper was chosen largely due to the similarities in the pig production systems studied to the UoL pig unit, although it is recognised that the pig unit was not fully operational until after the baseline period assessed in this chapter. This factor was applied to the indoor herd only covering CH_4 emissions from the slatted floor slurry system used by suckling sows and piglets, weaners (nursery), and bacon pigs.

These options were also chosen for the analysis as they should not have a direct implication on productivity and yields, and because the peer-reviewed research evidence base is a lot stronger for these options.

3.3 Results

The following sections outline the estimated emissions across the farm for the baseline period, which is then disaggregated into the arable footprint, pig unit footprint and an assessment of two options to reduce emissions from each enterprise.

3.3.1 Baseline average emissions

The average annual farm-level emission footprint was 991 (± 253) t CO_2e, 51% of which was from pig unit emissions and 49% from arable production (see Table 3.3). Of the pig unit emissions, 47% of the footprint came from the pig feed, with manure management at 41%, enteric fermentation at 11% and pig bedding for the remaining 1%. With arable emissions, the majority stemmed from the manufacture of fertiliser products (33%; majority being N-fertiliser at 27%) and direct N_2O emissions associated with applying N fertiliser (31%). Indirect N_2O emissions contributed 15% of the footprint with all other sources representing the remaining 20% of the footprint. Pesticide manufacture had the lowest footprint (2%), with seed production also very low at 3%. Diesel consumption and emissions from tractor manufacture and depreciation contributed 7 and 6% of the arable footprint, respectively.

Table 3.3 Average annual emissions (t CO_2e) and standard deviation for the arable and pig enterprises on the UoL farm between 2015 and 2018.

Enterprise	Emission source	Emissions (t CO_2e)*	Standard Deviation
Arable	**Diesel consumption**	**32**	**7**
	Soil direct N_2O emissions	**154**	**20**
	Synthetic fertiliser	89	17
	Organic amendments	29	4
	Crop residues	36	6
	Soil indirect N_2O emissions	**73**	**5**
	Synthetic fertiliser	45	5
	Organic amendments	22	2
	Crop residues	7	1
	CO_2 from urea	**13**	**14**
	Fertiliser manufacture	**162**	**21**
	Pesticide manufacture	**9**	**6**
	Seed	**16**	**3**
	Tractor	**29**	**0**
Pig Unit	**Enteric fermentation**	**57**	**33**
	Manure management	**208**	**121**
	Feed	**237**	**137**
	Bedding	**4**	**3**
	TOTAL	**991**	**253**

3.3.2 Arable emissions

The following section outlines the main results for the three key combinable crops grown on the UoL farm, and grass silage. Spring barley was grown only in 2016 and so is not a part of the main baseline footprint or conclusions of this chapter, but the results are presented in a section below. Appendix A has a breakdown of the results of the LCA per crop per year.

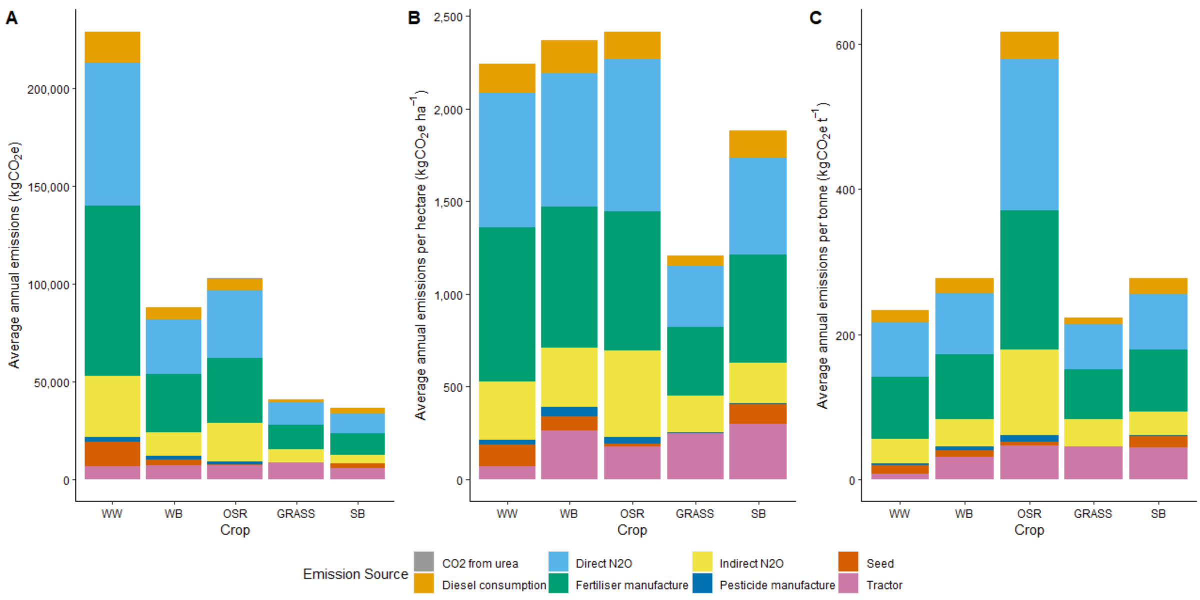

Figure 3.3 Average annual arable emissions (kg CO_2e) (A), average annual emissions per hectare (B) and average annual emissions per tonne of product (C) by key emission source type and crop grown. Averages taken between 2015 and 2018. WW = Winter Wheat, WB = Winter Barley, OSR = Oilseed Rape, SB = Spring Barley.

In all years, the combination of direct N_2O emissions from N fertiliser application and the manufacture of fertilisers (N and non-N) was the greatest source of emissions, although this fluctuated year-to-year and between crops (Figure 3.3). In 2015, direct N_2O emissions were greatest at (47% of the annual arable GHG footprint), whereas the following 3 years saw mostly declines in the contribution of emissions from synthetic N fertiliser sources (see Appendix A). The use of synthetic N fertiliser on the arable land was greatest in 2015, totalling 123 t of product, with the majority being AN (77%). The remainder of the synthetic N applications in that year were from a fertiliser called 'Sulphur +', which is an ammonium-sulphate-nitrate derivative. The lowest direct N_2O emissions from synthetic N applications occurred in 2018 (126 t CO_2e), representing 44% of the arable footprint in that year. A switch in the type of main N fertiliser used was made in 2018, converting from predominantly solid AN to liquid Urea Ammonium Nitrate (UAN), which has an EF of 0.006 kg N_2O released per kg N applied per year (0.6%), compared to 0.01 kg N_2O kg N^{-1} yr^{-1} (1%) with AN.

Indirect N_2O emissions were the third largest GHG source for arable farming at the UoL farm site between 2015 and 2018 (23% of emissions), although year-to-year the findings were fairly similar. The combination of atmospheric deposition and leaching or runoff was responsible for between 66 t CO_2e in 2018 and 105 t CO_2e in 2017. A higher quantity of urea fertiliser was applied in 2017 (39 t), which has an EF similar to UAN (0.006 kg N_2O

kg N^{-1} yr^{-1}) but has a higher fraction of synthetic N that could volatilise as ammonia (NH_3) and nitrogen oxides (NO_x) and be deposited on soils or water surfaces ($Frac_{GASF}$). The $Frac_{GASF}$ for urea is 0.15 kg NH_3-N + NO_x-N per kg N applied, whereas the value for AN based fertilisers is three times smaller (0.05 kg NH_3-N + NO_x-N per kg N applied; IPCC, 2019).

Winter wheat

Winter wheat was the most widely grown combinable crop on the UoL farm between 2015 and 2018, covering an average area of 103 (± 19) ha with production varying between 748 t (2017) and 1,366 t (2015). Yield varied between 8.3 and 10.7 t ha^{-1}, with an average of 9.7 t ha^{-1}. Average winter wheat emissions were 229 (± 42) t CO_2e, 2,245 (± 190) kg CO_2e ha^{-1}, and 234 (± 24) kg CO_2e t^{-1} grain across the four-year baseline.

In terms of total emissions for the crop, winter wheat had the largest GHG footprint of the crops assessed in all four years. 2015 winter wheat emissions were greatest at 298 t CO_2e, which declined through 2016 reaching a low of 186 t CO_2e in 2017 before rising to 219 t CO_2e in 2018 (

Table 3.4).

Table 3.4 Winter wheat total crop emissions (t CO_2e), emission per hectare (t CO_2e ha^{-1}) and emission intensity (kg CO_2e t^{-1} grain).

Year	Area (ha)	Yield (t ha^{-1})	Total emissions for the crop (t CO_2e)	Emission per hectare (kg CO_2e ha^{-1})	Emission intensity (kg CO_2e t^{-1} grain)
2015	128	10.7	298	2,334	218
2016	98	8.3	215	2,200	265
2017	75	10.0	186	2,481	248
2018*	112	9.7	219	1,964	203

*** GHG emissions in 2018 were economically allocated between the grain and straw as 500kg straw was retained on-farm for pig unit bedding. In all other years, straw is assumed to be 100% baled and removed from the farm.**

On a per hectare basis, winter wheat emissions were greatest in 2017, which is due to the fact that the area of wheat grown in 2017 was smaller than other years at 75 ha. The 2016 GHG footprint shows a greater emission intensity per tonne of product (265 kg $CO_2e\ t^{-1}$) than other years, likely due to the lower production output in that year.

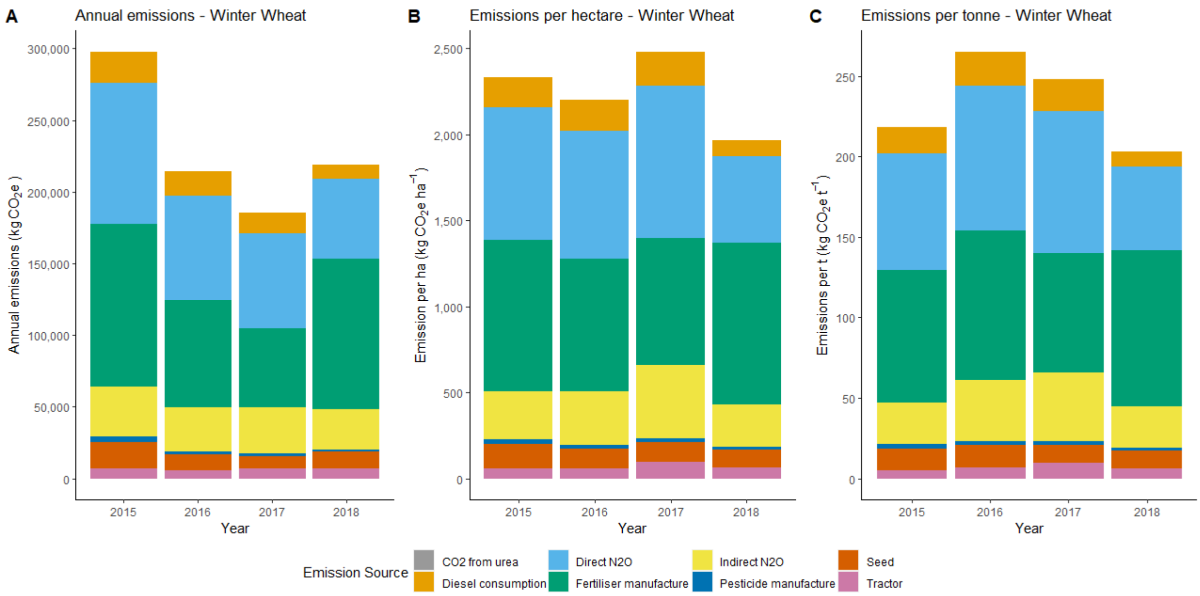

Figure 3.4 Total GHG emissions (A), emission per hectare (B) and emission intensity (C) associated with the production of winter wheat between 2015 and 2018 on the UoL farm.

Figure 3.4 illustrates that upstream fertiliser manufacture and direct N_2O emissions were the key emission hotspots for winter wheat. Fertiliser manufacture emissions made up 30-43% of the crop footprint over the four-year baseline and direct N_2O emissions from soil typically made up roughly one third of the footprint, most of which was due to manufactured nitrogen applications. Indirect N_2O emissions were lower than direct N_2O at between 12-17% of the footprint and diesel consumption made up between 5-8% of the footprint across the years.

Embedded emissions in wheat seed, pesticide manufacture, CO_2 emissions from urea and the manufacture and depreciation of the machinery made up a very small part of the footprint, typically no more than 10% cumulatively.

Winter barley

Winter barley was grown in all years of the baseline period, most of which was for feed. Only 38 (± 16) ha were used on average to grow winter barley between 2015 and 2018, with the most grown in 2017 (57 ha). Typical winter barley production was lower than winter wheat, with an average of 317 (± 121) t between 2015 and 2018. Yield varied between 7.3 and 9.9 t ha^{-1}, with an average of 8.5 t ha^{-1}. Average annual emissions from winter barley production were 88 (± 33) t CO_2e but average area emissions were smaller than winter wheat (2,368 ± 367 kg CO_2e ha^{-1}). Average emission intensity was slightly higher than winter wheat at 277 (± 13) kg CO_2e t^{-1} grain, likely due to the lower production of winter barley and smaller area on which it was grown in 2018.

2017 winter barley had the greatest absolute emission footprint (130 t CO_2e) and the following year saw a large decrease in emissions to only 37 t CO_2e. Emissions per hectare and emission intensity ranged between 1,943 to 2,959 kg CO_2e and 266 to 298 kg CO_2e, respectively (Table 3.5; Figure 3.5).

Table 3.5 Winter barley total crop emissions (t CO_2e), emission per hectare (t CO_2e ha^{-1}) and emission intensity (kg CO_2e t^{-1} grain).

Year	Area (ha)	Yield (t ha^{-1})	Total emissions for the crop (t CO_2e)	Emission per hectare (kg CO_2e ha^{-1})	Emission intensity (kg CO_2e t^{-1} grain)
2015	29	9.9	86	2,959	298
2016	50	7.3	98	1,943	266
2017	57	8.3	130	2,295	276
2018	16	8.5	37	2,285	269

The main emission hotspots for winter barley were also fertiliser manufacture (24-42%), direct N_2O (26-35%) and indirect N_2O (13-16%). Emissions from the tractor manufacture and depreciation totalled between 5 and 22% over the four years, the higher end of which occurred in 2018. Barley seed emissions were ≤ 5% of emissions across the baseline period and diesel consumption varied between 5 and 10% of the footprint.

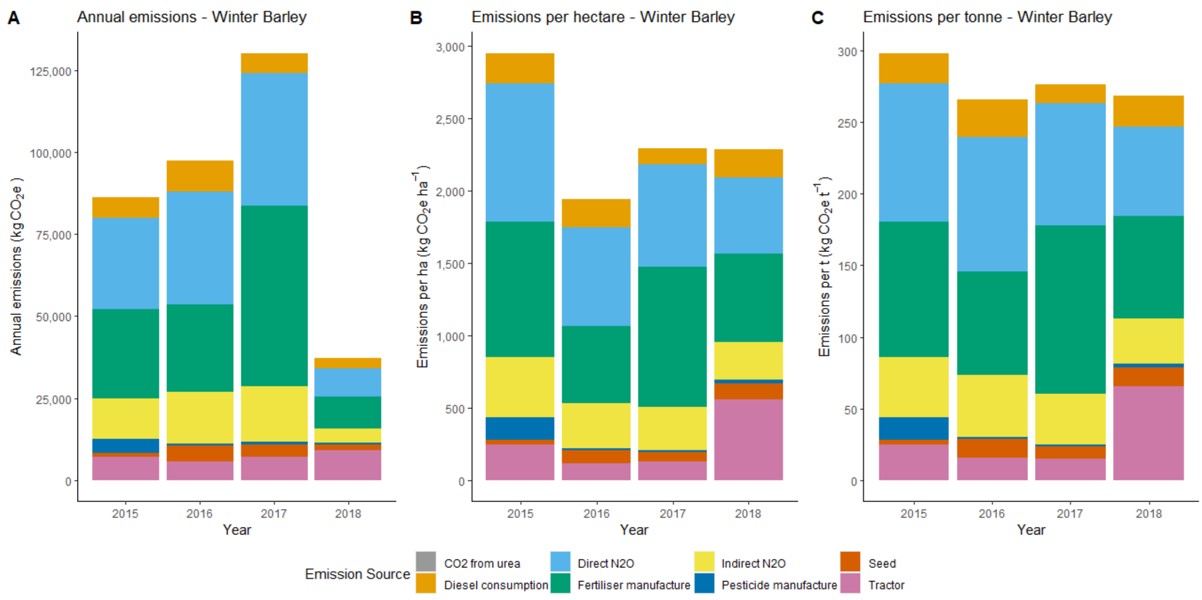

Figure 3.5 Absolute GHG emissions (A), emission per hectare (B) and emission intensity (C) associated with the production of winter barley between 2015 and 2018 on the UoL farm.

Spring barley

Spring barley was only produced in 2016 and appears to be a non-rotational crop in general, so the results in this section are purely for evaluating key drivers of emissions and have not been used in the mitigation section. 131 t spring barley was grown on 19 ha land in 2016, producing a yield of 6.8 t ha^{-1}. Emissions were 37 t CO_2e, 277 kg CO_2e t^{-1} grain and 1,882 kg CO_2e ha^{-1}.

Spring barley has similar emission hotspots to winter barley, with 31% of emissions coming from the manufacture of fertiliser, 28% from direct N_2O emissions, 16% from embedded tractor emissions, 12% from indirect N_2O, 8% from diesel consumption and 6% from seed production.

Oilseed rape

OSR is a break crop typically grown in a rotation with cereals, such as winter wheat and barley, to reduce pest burden. On average, 42 (± 11) ha of OSR was grown producing 170 (± 57) t of product across the four-year baseline period. Yield varied between 3.0 and 4.7 t ha^{-1}, with an average of 4.0 t ha^{-1}. As OSR is a low yielding crop, its emission intensity is higher than the other commodities grown on-farm. Average crop emissions were 103 (± 31) t CO_2e overall, 618 (± 57) kg CO_2e t^{-1} grain and 2,418 (± 210) kg CO_2e ha^{-1}. For an annual breakdown of emissions, see Table 3.6.

Table 3.6 Oilseed rape total crop emissions (t CO_2e), emission per hectare (t CO_2e ha^{-1}) and emission intensity (kg CO_2e t^{-1} grain).

Year	Area (ha)	Yield (t ha^{-1})	Total emissions for the crop (t CO_2e)	Emission per hectare (kg CO_2e ha^{-1})	Emission intensity (kg CO_2e t^{-1} grain)
2015	44	4.7	120	2,734	582
2016	29	3.0	63	2,144	715
2017	36	4.2	85	2,394	570
2018	60	4.0	143	2,400	605

Key emission hotspots for OSR are very similar to cereals, with fertiliser manufacture (27-39%) and direct soil N_2O emissions (31-37%) being the dominant sources in all years. Minimal emissions came from seed production and pesticide manufacture, contributing no more than 10% of emissions together in the four year baseline. Absolute emissions were lowest in 2016 at 63 t CO_2e but presented the greatest emission intensity at 715 kg CO_2e t^{-1} due to the low yield (88 t) (Figure 3.6).

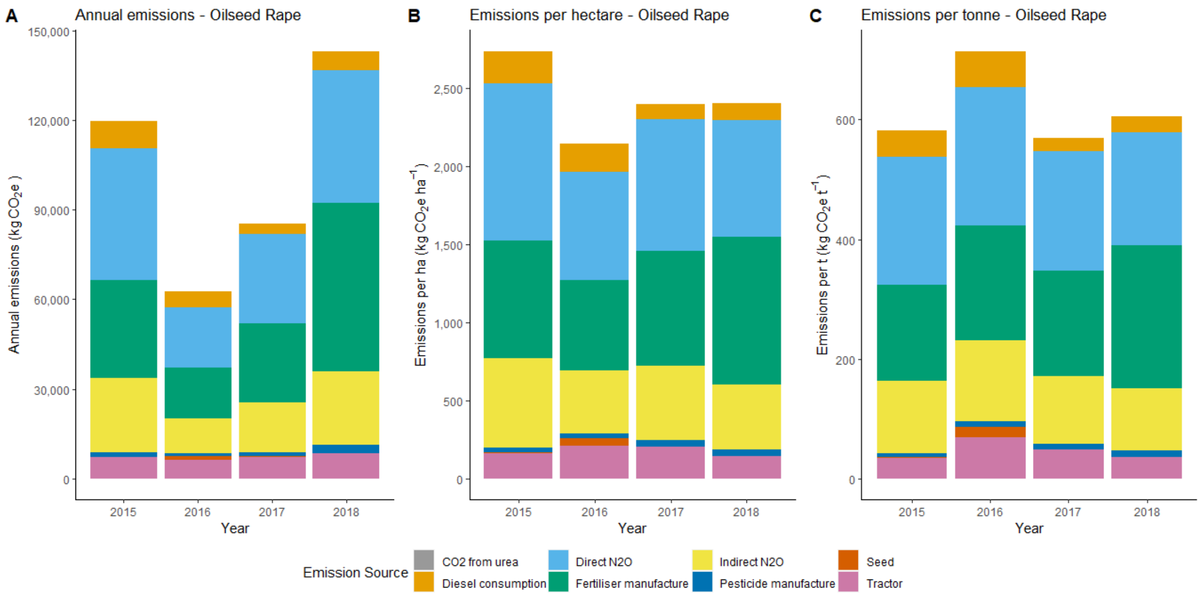

Figure 3.6 Total GHG emissions (A), emission per hectare (B) and emission intensity (C) associated with the production of oilseed rape between 2015 and 2018 on the UoL farm.

Grass silage

Although grass silage is not an arable crop, it was grown on the farm across the four-year baseline period and fairly high quality input data was available, so it has been reported in this chapter for completeness. Grass doesn’t require the same inputs and processes to be able to grow and be harvested for silage, so the emission footprint is lower than for combinable crops. Average annual emissions were 41 (± 13) t CO_2e overall, 223 (± 97) kg CO_2e t^{-1} silage and 1,206 (± 411) kg CO_2e ha^{-1}.

For grass silage, the majority of emissions in the footprint were from the manufacture of fertilisers, particularly N-based fertiliser (14-38%), and the subsequent direct N_2O emissions from application (6-36%). Fewer operations were required to produce the silage, resulting in a small diesel footprint (2-7%), and the fields must have been reseeded shortly before 2015 as no seed was used in any of the four years. This left the embedded emissions in the manufacture and depreciation of farm machinery at a higher average proportion in the baseline period (12-50%).

3.3.3 Pig unit emissions

Average pig unit emissions across the three years of data were 676 (± 29) t CO_2e, which equated to an average of 4.99 (± 0.01) kg CO_2e kg liveweight^{-1} (Table 3.7). The three main sources of emissions for the pig unit were feed, manure management, and enteric methane production, although the latter is very small due to pigs being monogastric animals.

Table 3.7 Annual pig liveweights and average emissions per kg liveweight per year (kg CO_2e kg liveweight^{-1} year^{-1})

	Liveweight (kg)	Emissions per kg liveweight per year (kg CO_2e kg liveweight^{-1} year^{-1})
2016	131,199.00	5.00
2017	131,199.00	5.00
2018	144,318.90	4.97
3-year average	**135,572.30**	**4.99**

Feed emissions were estimated to be an average of 237 (± 137) t CO_2e across the three years of pig data (Table 3.3), which equates to an average of 1.75 kg CO_2e kg liveweight^{-1}. The highest emissions were in 2018 (337 t CO_2e), which is to be expected considering more pigs were on-farm in that year.

Average manure management emissions across the three years for which pigs were present on the farm were 208 (± 121) t CO_2e (Table 3.3) and 1.54 kg CO_2e kg liveweight^{-1}. The majority of the footprint was associated with the indoor herd (233 ± 11 t CO_2e), which utilised a slatted floor slurry pit system (205 ± 9 t CO_2e) and separate solid storage system (28 ± 1 t CO_2e). The outdoor herd manure management footprint was 45 (± 2) t CO_2e on average, which was mostly from the solid storage of manure mixed with bedding in the outdoor arcs (43 ± 2 t CO_2e) with the remaining 2 (± 0.1) t CO_2e coming from CH_4 production from grazing deposits on the pasture (Figure 3.7).

Enteric methane emissions amounted to 57 (± 33) t CO_2e on average across the three years' of data (Table 3.3) and 0.42 kg CO_2e kg liveweight^{-1}. As enteric methane emissions are calculated using a Tier 1 calculation and emission factor, the typical animal weight and population size largely determines the quantity of emissions produced. Therefore, the indoor herd, being the larger unit had a higher average footprint of 54 (± 3) t CO_2e compared to the outdoor unit (23 ± 1 t CO_2e).

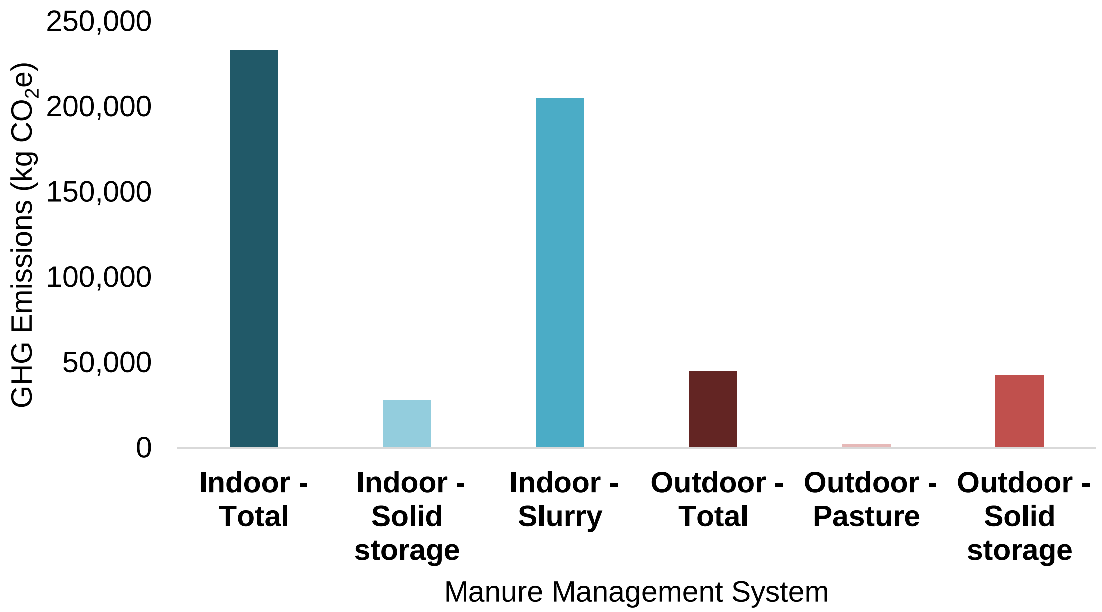

Figure 3.7 GHG emissions associated with the different types of Manure Management System present in the pig unit at the UoL farm.

3.3.4 Reducing emissions from arable and pig production

The UoL farm used mostly AN (23 – 33 t N yr^{-1}) and urea (12 – 13 t N yr^{-1}) between 2015 and 2017, but in 2018 the predominant N fertiliser used was a liquid UAN fertiliser called Omex (N26 + 5S). Despite applying 27 t N in the form of the UAN, emissions fell by 28% due to the lower direct N_2O EF associated with the use of UAN (UAN: 0.004 kg N_2O-N, AN: 0.006 kg N_2O-N; NAEI, 2020).

In the scenario with the NI DCD added to applications of AN and urea for wheat, barley and grass silage from 2015 to 2017, average annual GHG emissions for the farm would have been 2% lower and total arable emissions would have been 5% lower (Table 3.8). This stems from a reduction in direct N_2O from fertiliser applications of 17%. The majority of this reduction occurred in 2015 (21%), taking the annual emissions across crop types from 184 t CO_2e to 145 t CO_2e.

Looking at individual crops, the mitigation effect of DCD was strongest in winter barley in 2015, with a 48% reduction in direct N_2O emissions. This was closely followed by silage grown in 2016 with a 47% reduction and winter wheat grown in 2015 with a 43% reduction.

Table 3.8 Mitigation effects of a DCD nitrification inhibitor on average (2015 – 2018) GHG emissions at farm- and enterprise-level.

	Without NI (t CO_2e yr^{-1})	**With NI**[a] (t CO_2e yr^{-1})	**% decrease**
Farm-level	991	970	2.1
Arable	489	467	4.5

[a]DCD was applied to AN and Urea applications to wheat, barley and grass grown for silage between 2015 and 2017. In 2018 mostly UAN was used so this was not tested with DCD.

In the second scenario, the pig unit slurry management had slurry acidification applied to test the potential reduction in methane emissions. This was only applied to indoor pigs that are based on slurry system (with a slatted floor), and including the suckling sows and piglets, nursery pigs and bacon pigs. The scenario modelling revealed that average annual GHG emissions from manure management were reduced by 53%. This equates to a reduction of 11% at farm-level and 22% when considering the pig unit alone (Table 3.9).

Table 3.9 Mitigation effects of slurry acidification on average (2015 – 2018) GHG emissions at farm- and enterprise-level.

	Without acidification (t CO_2e yr^{-1})	With acidification[a] (t CO_2e yr^{-1})	% decrease
Farm-level	991	881	11.1
Pig unit	507	396	21.9

[a] **Applies** to acidification of slurry from the indoor herd where a slatted floor was in-place. This included the farrowing unit (suckling sows and piglets), the nursery unit (pigs up to 40kg) and the finishing unit (baconer pigs over 40kg).

With both practices combined, total UoL farm emissions would be 131 t CO_2e lower than the estimated 991 t CO_2e between 2015 and 2018, which is a 13% reduction in annual average GHG emissions.

3.4 Discussion and conclusions

This chapter presents the results of an LCA analysis of the key commodities produced on the UoL farm. The purpose of this research was to investigate the key GHG sources of the UoL farm as a case study farming system representative of arable and pig production in England and Wales, as well as model opportunities to reduce emissions on-farm that can be compared against the baseline. Pork is the most widely consumed meat globally (Ottosen et al., 2021), and the environmental impacts of pig production systems are substantial, potentially 9% of livestock-produced GHG emissions globally (Gerber et al., 2013). Furthermore, cereal and oilseed crops are the first and third most produced arable crops worldwide by land area at 732 Mha and 344 Mha, respectively (FAO, 2023a).

The UoL farm was deemed to be a representative case study of commercial-scale mixed arable and pig production in England and Wales for several reasons. Firstly, looking at the latest Animal and Plant Health Agency (APHA) pig population maps (2020/2021 data), Yorkshire has a greater pig density at > 200 - 455 pigs per square kilometre (km^2) than any other region in England or Wales and there are several other hotspot areas of higher pig density around these two countries where pig density is 20 – 200 pigs per km^2 (APHA, 2024). With the UoL farm having an average pig population of 1,874 individuals between 2016 and 2018, most of which are indoors, this would be representative of at least pig production in Yorkshire but also areas of East Anglia and other hotspot regions in England and Wales. Secondly, arable crop yields on the UoL farm were within a similar range to national averages. Average (with range in brackets) wheat, barley and OSR yields between 2015 and 2018 were 8.2 (7.8 – 9.0) t ha^{-1} in England and 7.4 (6.9 – 8.1) t ha^{-1} in Wales, 6.9 (6.4 – 7.6) t ha^{-1} in England and 6.5 (6.1 – 7.2) t ha^{-1} in Wales, and 3.5 (3.1 – 3.9) t ha^{-1} in England and 3.6 (3.1 – 3.9) t ha^{-1} in Wales, respectively (Defra, 2023b). As shown in Appendix A, the range of UoL main cereal and oilseed crop yields fall mostly within these national ranges.

3.4.1 LCA of arable and pig production emissions

The first and second objectives of the chapter were to develop a baseline GHG emission inventory of the arable crops grown on-farm and for pig production and then perform an LCA on that data to understand the GHG emissions of the production systems. The baseline LCA revealed significant contributions from multiple sources within each enterprise that were expected given the type of farming, but with some inter-annual variability.

Looking at the arable emissions, this highlighted the predominant role of manufactured N fertilisers in driving GHG emissions, particularly through direct and indirect N_2O contributions, and the embedded emissions associated with their production (Table 3.3). Wheat and barley emissions were 234 (± 24) kg CO_2e t^{-1} grain and 277 (± 13) kg CO_2e t^{-1} grain, respectively, and OSR emissions were 618 (± 57) kg CO_2e t^{-1} grain, all of which were lower than figures reported by other sources (Williams et al., 2006; CHAP, 2022). The CHAP benchmarking report is a meta-analysis of LCA studies across a broad range of arable and horticultural products both in the UK and globally. The authors report that average winter feed wheat GHG emission intensity in the UK is 340 kg CO_2e t^{-1} but had a broad range from 130 to 910 kg CO_2e t^{-1}. Likewise, winter barley had the same average emission intensity but ranged from 300 to 730 kg CO_2e t^{-1} and OSR was higher at 740 kg CO_2e t^{-1} (640 to 1,000 kg CO_2e t^{-1}) (CHAP, 2022). Williams et al. (2006) is an earlier LCA study of agricultural commodities and reports feed wheat, winter barley and OSR emissions as 731, 726 and 1,710 kg CO_2e t^{-1}, respectively. Both papers used the GWP_{100} methodology for reporting global warming impact, like this study, but they also used older GWP values, which were higher in IPCC AR3, AR4 and AR5 (IPCC, 2001; IPCC, 2007; IPCC, 2013). Additionally, the UoL farm OSR yield was 11% higher on average compared to the national average for 2015-2018 at 3.5 t ha^{-1} (Defra, 2024d) and 21% higher than the 3.3 t ha^{-1} yield reporting in Williams et al. (2006), which would lower the GHG intensity of OSR production and could explain some of the variation in emission intensities.

Direct nitrous oxide emissions from the application of nitrogen fertilisers represented 30% of GHG emissions on average, making this a significant emission source to tackle in the mitigation scenarios. These emissions were calculated using IPCC Tier 2 equations, UK GHG Inventory Tier 2 emission factors and high quality primary farm activity data from farm management software records. This is considered a higher quality estimate than using secondary activity data or Tier 1 calculations and emission factors according to Ciroth et al. (2016), but is still lower quality and not as accurate as measured data. For example, a recent paper by Lloyd et al. (2024) measured fluxes of N_2O from winter wheat plots at the UoL farm under different nitrogen treatments using automatic chambers, which analysed GHG data every seven minutes. Measurements of N_2O at that scale provides a more accurate and precise picture of actual GHG emissions from application of nitrogen fertiliser, whilst accounting for any site-specific weather and soil conditions at the time of measurement. On the other hand, the UK GHG Inventory Tier 2 emission factor for direct N_2O is an average across the country based on field trials in the MIN-NO project (Sylvester-Bradley et al., 2015), so whilst geographically more accurate than a global Tier 1 emission factor, there still lies some uncertainty in the modelling of N_2O emissions in this LCA.

The pattern in emission intensity differences between cereals and OSR between this study and those mentioned above is consistent, with OSR being a lower yielding crop and thus having less product to dilute emissions across. Much of the scientific literature assessing the global warming impact of cereals, such as those studied in this chapter, focus on the by-products (e.g., straw for biofuel) or processed end-product (e.g., bread or pasta) rather than the crop leaving the farm gate (Vinci et al., 2022). However, as the production stage of the supply chain can be responsible for up to 80% of GHG emissions (Vermeulen et al., 2012), this chapter builds on the scientific literature supporting this line of evidence of arable LCA up to the farm gate.

On the pig production side of the UoL farm, the key GHG emission sources were the feed and manure management, which made up 47% and 41% of the pig enterprise emissions, respectively. The FU chosen was kg CO_2e per kg liveweight for the pig unit as this is a commonly accepted unit for pig LCAs (McAuliffe et al., 2016) and without additional sales data on carcass weights or pig meat produced, this was the most robust option. GHG emissions from pig production were estimated to be 4.99 kg CO_2e kg liveweight^{-1}, which is higher than other studies (Basset-Mens and van der Werf, 2005; Pelletier et al., 2010; Stephen, 2012; Dourmad et al., 2014). For instance, both Dourmad et al. (2014) and Basset-Mens and van der Werf (2005) estimated non-UK European pig production emissions to produce 2.3 kg CO_2e kg liveweight^{-1}, and Stephen (2012) and Pelletier et al. (2010) reported slightly lower (2.03) and higher (2.47) emissions, respectively. On the other hand, a study from Italy comparing six pig farming systems found a wide range in emissions from 2.69 to 5.81 kg CO_2e kg liveweight^{-1} (Bava et al., 2017). A key factor to consider in the disparity between this study and GHG emission intensities reported in the literature is that in the period studied (2016-2018), the pig unit was not at full capacity and data on population numbers and live weights was not strictly recorded. Bava et al. (2017) also note the size and efficiency of pig system as a predictor of emissions per kg of liveweight, with smaller, less efficient farms having a higher emission intensity. Additionally, this study relied on proxy data for the UoL farm pig feed from a conventional English pig production system (Kool et al., 2009), so this combined with the lower capacity of the unit and poor data quality may have contributed to the higher emission intensity in this study. This discussion continues in Section 3.4.3.

3.4.2 Modelling reductions in emissions

A core objective of this study was to model scenarios of changes in farm practice on the UoL farm to assess how these would impact GHG emissions compared to the baseline period.

The assessment of pig unit emissions revealed the substantial impact of feed production and manure management on overall emissions. These findings highlight the significance of optimising feed formulations, or switching to home grown feeds (Garnett, 2011), and adopting efficient manure management practices to reduce emissions from pig farming operations (Chadwick et al., 2011; Montes et al., 2013). For example, using LCA to assess the environmental impacts (including GHG emissions) of reformulating feed for pigs, poultry and young bulls, Garcia-Launay et al. (2018) found a 20% decline in GHG emissions when feeds had been reformulated to include environmental impacts as a key consideration. However, as the pig feed data was largely from secondary data sources for the UoL farm case study, a manure management practice was chosen for the mitigation modelling and acidification was chosen specifically due to conversations with the farm management and research teams.

The application of slurry acidification techniques exhibited substantial potential for mitigating methane emissions from pig manure management on the UoL farm, more than halving manure management emissions compared to the 2016 – 2018 baseline period. This practice offers a plausible pathway for significant emission reductions at the farm level that may be applicable to other pig farms in England and Wales. In Denmark, up to 20% of animal slurry is acidified to a pH of 5.5 through policy mechanisms to lower ammonia emissions, which also subsequently reduces N_2O emissions (Pedersen et al., 2022). Previous research with pig slurry acidification has included laboratory experiments comparing organic and inorganic acids, with the most effective being found to be nitric acid, which reduced CH_4 emissions by >99% at an optimal pH of 5.5 (Dalby et al., 2022). Similar research has found pH 5.5 to be optimal for CH_4 reduction, but has used sulphuric acid and found similar reductions of 61% to 87% (Petersen et al., 2012; Misselbrook et al., 2016). This includes some research investigating the efficacy of sulphuric acidification of cattle slurries, in particular for dairy systems, which is where methane reduction reached 61% (Misselbrook et al., 2016). However, the authors do note that acidification may not be commercial scalable for all slurry-based farming systems due to the increased burden of health and safety assessments and potentially some upgrades to pipework to avoid corrosion from the acids (Misselbrook et al., 2016; Dalby et al., 2022; see also Ambrose et al., 2023). Pig slurry acidification modelled using LCA methodology in one study increased GHG emissions by around 8%, although the authors do not explain what this could be the result of, but lowered pollutants from other impact categories, specifically acidification and eutrophication potential (Pexas et al., 2020). This chapter did not assess the acid life cycle emissions, which could partly explain the opposing result in Pexas et al. (2020), or other LCA impact categories, despite agricultural systems being key to reducing non-GHG pollutants such as nitrate for eutrophication (Mackenzie, 2016). As the focus of this thesis was on improving understanding of net zero GHG emission opportunities, it was deemed non-essential to look at other impact categories, but future work could explore a broader range of environmental impacts to ensure conclusions are more holistic and assess life cycle implications of the practices themselves.

Emissions from the manufacture and application of nitrogen fertilisers were dominant emission sources on the arable side of the farm, so the practice modelled to reduce those emissions was the use of a nitrification inhibitor. This is an increasingly common practice, and more nitrogen and urea fertilisers are manufactured with an inhibitor (e.g., protected urea; Forrestal et al., 2019). Before investing in protected fertilisers, a simple practice farmers could do to ensure they are minimising their GHG emissions and fertiliser costs is to ensure they are not over-applying fertiliser beyond the crop's needs. Across the winter wheat, winter barley and oilseed rape grown on the University of Leeds farm, annual average (2015 – 2018) nitrogen applications were lower than the industry average (Defra, 2022b) and thus any further reductions to nitrogen applications may have resulted in yield changes. Introduction of the nitrification inhibitor DCD showed a 2% reduction in farm-level GHG emissions with a 17% reduction in direct N_2O emissions compared to annual average between 2015 – 2018. The GHG abatement potential used in this analysis was relevant to DCD use with AN and urea on wheat and barley cropping in the UK (Misselbrook et al., 2014; Lam et al., 2017), which matched the UoL case study farm well as these are two of the most produced crops on-farm. However, as oilseed crops and grass are also produced on-farm, and some of the grassland in particular would have had excreta deposited from the outdoor pig herd, it would be a useful development to this research to understand the N_2O reduction potential under those systems.

Although there are conflicting N_2O reduction potentials for NIs with slightly different systems (e.g., crops versus pasture) and N-based fertiliser being applied (e.g., AN versus slurry or urine) (Ruser and Schulz, 2015; Lam et al., 2017), all show a decline in N_2O with use. UK-based research into the efficacy of DCD as a NIs found direct N_2O emissions were reduced by 39% and 69% when applied with AN and urea fertilisers, respectively and even greater reductions were observed when DCD was applied with cattle urine (70% reduction) (Misselbrook et al., 2014). A review of NI efficacy by Ruser and Schulz (2015) found that whilst NIs like DCD are effective at reducing direct N_2O emissions from both arable and pasture land, the delay in nitrification caused by the inhibitor may increase indirect N_2O release through ammonia volatilisation if the fertiliser or organic material it is applied with isn't incorporated.

An interesting finding was that the farm changed the type of nitrogen fertiliser used during the baseline period, which caused GHG emissions to decline between 2017 and 2018.

From 2015 to 2017, the main fertiliser type used was AN and urea, however, UAN replaced most of the AN in 2018, which reduced direct and indirect N_2O emissions on-farm by 28%. UAN is a liquid N fertiliser and so is more quickly absorbed by soils and taken up by crops (Cowan et al., 2020). A study in maize systems found UAN to have a 52% lower GHG intensity in terms of N_2O emissions that straight urea fertilisers (Ren et al., 2021). However, other research focussed on UK arable systems has shown some differences in N_2O emissions between the use of AN and UAN in experimental crop sites. For example, Smith et al. (2012) compared different N fertiliser types across multiple sites growing winter wheat and found that the application of UAN produced 30% less N_2O than AN on one site (Bush Estate, Edinburgh), but between 63 to 68% more N_2O on two different sites in England (ADAS Boxworth and Terrington, East Anglia). In this study, the authors also found that when combining the UAN with a urease inhibitor (UI; Yara Agrotain) on the same three sites, Boxworth still exhibited an increase in N_2O emissions but only a 41% increase compared to the 68% without the UI, whereas the other two sites both showed a 43% decrease in N_2O emissions (Smith et al., 2012). This indicates that the UAN emissions on the UoL case study farm may have been reduced further if both NIs and UIs had been tested with the data, which could form a future development of this research.

3.4.3 Limitations

Whilst this chapter provides a strong assessment of crop GHG emissions over time, the GHG emissions from the pig unit have a high degree of uncertainty due to a lack of accurate herd population and weight data, as well as pig feed data. Population and weight data were derived from conversations with the pig unit manager in 2021/2022, with estimates based on memory and emails with the farm management company. More consistent and accurate population and weight data would vastly improve the confidence in the pig unit results, despite the consistent GHG emission footprint seen in other pig LCAs. Whilst the pig unit manager was able to provide some information about the pig feed (e.g., the specific product names), the complete ingredient list, composition and ingredient country origin was unavailable from the pig feed supplier, and neither was a carbon footprint for the product. It is not uncommon to have limited data on specific animal feed blend products, as each component varies in its composition in the feed, many ingredients are by-products of other processes (e.g., rape meal, distillers grains, biscuit meal; Röös and Nylinder, 2013) and if soy-based, accurate information on the sustainability of the production system (i.e., without deforestation) is unclear due to soy imports being mixed for transport (Fraanje and Garnett, 2020). To improve the accuracy and completeness of LCAs, such as the one conducted in this chapter, the pig feed industry should be supported in calculating the carbon footprint of their blended feed products, although this is only likely to come if demand for such data increases (e.g., by farmers). Additionally, any future LCA work for the UoL farm pig unit should have a higher degree of data quality around the population and weight of the two herds as the unit reached full capacity in 2022, so more accurate recording would have been put in place.

Furthermore, the results in this chapter do not represent a complete farm-level assessment as some crop and livestock data was unavailable. Potatoes and vining peas were grown on-farm between 2015 and 2018, although on a much smaller area than the combinable crops, and sheep were also periodically present on-farm in the pastures for grazing (on loan from Askham Bryan College). No data was available for any of these systems, so it was deemed appropriate to exclude them from the analysis with the caveat of understanding the scope of the analysis was not whole farm.

The two mitigation options chosen for the analysis were chosen based on the findings from the LCAs, but also the availability of GHG abatement data and from conversations with the farm managers. However, the economic cost of N fertilisers with NIs and lack of additional benefits (e.g., yield increases) may cause farmers to avoid adoption (Freeman et al., 2020). Furthermore, the scenario of introducing slurry acidification is based on conversations with the pig unit farm manager, however, the practicalities of introducing acidification are complex. For example, research has indicated that the acidification process itself can result in machinery corrosion (Fangueiro et al., 2015; Misselbrook et al., 2016). Thus, whilst the practice can reduce CH_4 emission by 90% (Vechi et al., 2022), the feasibility and cost of replacing pipework and machinery parts could be too high to be sustainable for the farm business.

Lastly, the UoL farm operates at commercial scale but there is a question of how representative the pig-arable system presented in this chapter is to the region, or wider systems found throughout England and Wales. Average farm size in Yorkshire is 94 ha (Defra, 2024a), so the UoL farm sits above average at 317 ha and soil on the farm is typically quite shallow (< 50 cm) and are mostly well drained and chalky, which isn't overly typical of the region (Cranfield Soil and Agrifood Institute, 2010). Therefore, the results in this chapter may not be completely representative of the region, however, they do contribute new data into the scientific literature and improve understanding around mitigation options for similar farms looking to transition towards net zero.

3.4.4 Conclusions

In conclusion, this chapter provides valuable insights into the typical emissions footprint of both arable and pig farming enterprises using new data from the University of Leeds farm. This work highlighted the key sources of emissions from several commodity products, supporting previous literature findings, and modelled two targeted mitigation strategies to reduce the calculated GHG emissions in the arable and pig enterprises. The findings highlight the importance of tailored approaches to emission reduction, considering the specificities of crop types and livestock management systems. For example, NIs were modelled in this study rather than simply modelling reduced nitrogen applications as average nitrogen use was already below the national average.

Moving forward, the adoption of innovative technologies and management practices, such as NIs and slurry acidification, holds promise for achieving emission reductions while ensuring sustainable agricultural production continues on the UoL farm. However, this research also illustrates a key challenge commonly faced in LCA around the lack of robust, high quality data. A key implication of this is that the pig unit findings are uncertain and future work should use higher quality population, weight and feed data in order to more accurately account for GHG emissions and model suitable scenarios for reducing those emissions.

Chapter 4 - Strategies for arable agriculture and woodland creation in England & Wales to contribute to net zero emissions

4.1 Introduction

Global greenhouse gas (GHG) emissions need to decrease substantially in the next few decades in order to keep global temperature increase below 1.5°C as stipulated in the 2015 Paris Agreement (Verschuuren, 2016; Chan et al., 2018). Several papers have assessed the feasibility of and governmental progress towards these climate change objectives (for example, see Jacquet and Jamieson, 2016; Peters et al., 2017; Roelfsema et al., 2020). Roelfsema and colleagues (2020) proposed an emission gap of between 22.4 and 28.3 gigatonnes of carbon dioxide equivalent (Gt CO_2e) by 2030 given current global policy implementation to keep warming below 1.5°C or 2°C. In the UK, the Climate Change Committee (CCC) delivered their progress report to UK Parliament in 2018 stating that "virtually no change in agricultural emissions" had occurred since 2008, with particular attention needed for non-CO_2 GHGs (CCC, 2018b). There has more recently been a surge in interest in setting targets for achieving 'net zero' emissions, a concept that emphasises the need to reduce GHG emissions across sectors with any residual emissions balanced by sequestration and storage of CO_2 from the atmosphere in soils, biomass and oceans (Hale et al., 2022). This includes the UK Government, which was the first major global economy to set the target of net zero emissions by 2050 in 2019 to replace the previous Climate Change Act 2008 target of 80% reduction in emissions by 2050. However, the ability to achieve this target is in question due to the technological and biophysical limitations that hinder efforts to move towards actual net zero emissions of GHGs (Dyke et al., 2021).

The Agriculture, Forestry and Other Land Use (AFOLU) sectors are part of this uncertainty, largely due to the numerous biological, physical and chemical interactions that occur between the land and atmosphere that influence GHG fluxes (Tilman et al., 2001; Hillel and Rosenzweig, 2010; Cantarello et al., 2011). These sectors are combined for the purpose of reporting land based GHG emissions to the United Nations (UN), however, there are contrasts between the individual AFOLU sectors. For example, the forestry sector is largely a net sink of GHG emissions, whereas agriculture is a net source. Between 2007 and 2016, global AFOLU emissions were estimated to be 12.0 ± 2.9 Gigatonnes (Gt) CO_2e yr^{-1} in the International Panel on Climate Change (IPCC) Special Report on Climate Change and Land, representing 23% of anthropogenic GHG emissions. Agriculture is estimated to contribute approximately one half of AFOLU emissions (6.2 ± 1.4 Gt CO_2e yr^{-1}) (IPCC, 2019b). However, the Food and Agriculture Organisation (FAO) report on '*Emissions from agriculture and forest land*' suggests that agricultural emissions were closer to 10.7 Gt CO_2e yr^{-1} in 2019 (FAO, 2021). The agriculture sector is also heavily influenced by agronomic management choices to improve yields and profitability. The combination of natural and anthropogenic influences on agriculture make it difficult for the sector to reduce emissions significantly without impacting productivity and profitability (Smith et al., 2007). It is estimated that as much as 80% of global agricultural emissions stem from the production stage of agri-supply, although significant amounts of GHGs are emitted at pre-production and post-production stages (Vermeulen et al., 2012); all three areas have the potential to reduce overall emissions with better management. At regional scale, 75% of Europe's agricultural GHG emissions were sourced from within the farm gate (FAO, 2021) and in the UK 11% of total emissions come from agricultural production (Defra, 2019a; Defra, 2021a). Between 1990 and 2022, there has been a 13% reduction in agricultural GHG emissions (Department for Energy Security and Net Zero [DESNZ], 2024). Whilst reductions in N fertiliser application and the number of livestock have contributed to this, there are further efforts to be made to reduce the overall footprint of agriculture in England and Wales in line with national net zero ambitions. In contrast, other economic sectors have seen more significant declines in GHG emissions in this timeframe, such as a 73% reduction in the energy sector from improvements to electricity supply and an 84% in industry emissions (DESNZ, 2024).

This means that for the agriculture sector, effort is needed to identify suitable changes in practice that can help to reduce GHG emissions across the sector and explore ways of enhancing carbon sequestration. These arguments form the basis of this chapter's research, shifting the focus to arable mitigation options here and exploring livestock mitigation in Chapter 5. Research using Marginal Abatement Cost Curves (MACC) analyses provide useful information about the mitigation potential of arable practices, as well as their cost-effectiveness and are often used in the context of developing new policies. MACC research from Fellman et al. (2021) found several practices that were low cost and had higher mitigation potentials, such as nitrification inhibitors in arable cropping and precision farming, whereas improving the timing of N fertiliser application may be very cost-effective but had a low mitigation potential (Fellmann et al., 2021). This is corroborated by earlier research from Eory et al. (2018), which also discusses the importance of including upstream and downstream emissions (i.e., off-farm) sources of emissions and carbon sequestration in MACCs to improve our understanding of the most effective mitigation options. Both papers emphasise regional heterogeneity as a key consideration for reducing GHG emissions in agriculture, agreeing that a flexible and more localised approach is needed, rather than a blanket, national mitigation strategy (Eory et al., 2018; Fellmann et al., 2021). Whilst this chapter is not a MACC of arable mitigation practices and carbon sequestration options, it does provide a regional-level scenario analysis of similar practices discussed in these papers.

4.1.1 Technical potential

There are important constraints to the feasibility and technical potential of some arable farm practices to farmers, for example in terms of the cost of implementation, appropriateness considering soil, climate, and previous management factors and land use rights (Smith et al., 2007; Bustamante et al., 2014). As mentioned in Chapter 2, the replacement of the European Common Agricultural Policy (CAP) with Environmental Land Management Schemes (ELMs) is providing farmers with similar and new incentives to reduce their farm's GHG emissions, whilst simultaneously improving biodiversity, water, and air quality (Tyllianakis et al., 2023). For example, payments of up to £589 per year for farmers who assess their nutrient management plans (NUM1) and establishing new hedgerows at £10 per 100 m (Defra, 2024c). However, some have criticised ELMs as being insufficient for farmers to take climate action and be able to afford their business as the payments do not cover the loss of direct payments in CAP (Swales, 2022).

Looking towards options for removing carbon from the atmosphere and storing it in soil and vegetation, there are some removal opportunities that have political limitations that need to be accounted for and may make them unpopular with some land owners. The creation of 30,000 ha of woodland per annum by 2025 and then 50,000 ha per year from 2035 to 2050 has been a recommendation from the CCC as a UK-specific strategy to offset GHG emissions (CCC, 2020a; CCC, 2020b). This would require agricultural land to be released in the range of scenarios outlined in their report (CCC, 2020b) and may only be applicable to CO_2 offsets rather than non-CO_2 GHGs (Parliamentary Commissioner for the Environment, 2022). For many land owners, agriculture is their primary business and income source, meaning land release for woodland creation is not possible unless financially incentivised. However, opportunities to regenerate woodland on unproductive farmland is an alternative possibility that can deliver carbon sequestration, and therefore GHG abatement, with appropriate funding mechanisms (Westaway et al., 2023).

4.1.2 Research gap

Much of the previous literature tackling the mitigation potential of farming practices has focussed on carbon sequestration potential, as opposed to GHG mitigation as a whole (Lal, 2003; Smith, 2004; Falloon et al., 2004; Powlson et al., 2011; Lal, 2011; Chambers et al., 2016; Lal et al., 2018). There has not been a regional scale attempt to quantify arable GHG mitigation through reduction or removal practices in England and Wales to identify the potential for the sector to reach net zero emissions by 2050. However, as the effects of the changing climate become more prominent in England and Wales, there is an increasing need for an attempt to answer this question to reduce the sectors' contributions to climate change.

Therefore, this chapter utilises a combination of spatial data on land under arable management and regional-level GHG emissions from arable sources, to investigate the regional-scale opportunities for arable agricultural land and afforestation to contribute to net zero policy. The objectives of the study were to identify suitable land area across regions of England and Wales for the GHG abatement and afforestation practices, as well as the practices to test in the scenario analysis itself and then test varying levels of uptake in each region to understand the potential to reduce soil-based emissions. This includes carbon sequestration potential associated with woodland planting in largely unproductive areas. This chapter will not provide an in-depth economic analysis of the strategies chosen, nor will it identify the 'best' set of strategies for GHG mitigation in each region, largely due to a lack of high-quality data to fulfil these needs.

4.2 Methodology

4.2.1 UK GHG Inventory - regional profiles

The UK GHG Inventory is a compilation of cross-sector GHG emissions, which is updated annually and submitted to the United Nations Framework Convention on Climate Change (UNFCCC) to report on any changes in GHG emissions. Whilst this data is useful for national-scale modelling, this chapter's focus was on regional-level emissions and modelling. Therefore, the dataset used in this chapter is from the July 2022 release of the UK GHG Inventory (covering emissions from 1990 to 2020) local authority emissions from 2005 to 2020 (Department for Business, Energy and Industrial Strategy [BEIS], 2022).

Using this supplementary dataset, it was possible to disaggregate emissions by Government Office Region (GOR) for England and the whole of Wales. London was combined with the South East to avoid smaller sampling areas. For the agriculture sector, emissions are reported for electricity, gas, livestock, soils, and 'other' emission sub-categories. The 'other' agriculture category mostly represents emissions from agriculture that are energy-related, such as fuel consumption, but not electricity or gas consumption. The data used in this chapter was the Agriculture Soil emissions (Table 4.1), which includes direct and indirect nitrous oxide (N_2O) emissions from the application of fertilisers (manufactured and organic), and CO_2 emissions from urea fertilisation and liming.

This regional data was used to calculate the average baseline Soil emissions (referred to as 'Soil emissions' henceforth) from arable agriculture over a three-year period (2018 - 2020). The next section describes the spatial analysis performed to allocate Soil emissions and land areas to purely arable land and arable land in mixed farming systems.

Table 4.1 Regional 3-year (2018 – 2020) average agricultural Soil emissions from the UK GHG Inventory regional and local authority dataset (BEIS, 2022b).

GOR	Agriculture Soils (kt CO_2e)
North East 3-year Average	238
North West 3-year Average	608
Yorkshire 3-year Average	768
East Midlands 3-year Average	968
West Midlands 3-year Average	631
East of England 3-year Average	1,710
South East/London 3-year Average	639
South West 3-year Average	1,178
Wales 3-year Average	752
TOTAL	**7,492**

4.2.2 Spatial analysis

The Centre for Ecology and Hydrology (CEH) multi-tier archetypes dataset was used to analyse the British agricultural landscape for potential areas of suitability for future regional-scale net zero options at a resolution of 1 km^2 (Goodwin et al., 2022). This dataset provides three rasters of different tiers of spatial information across Great Britain (except for Tier 3, which applies only to England and Wales): Tier 1 describes broad landscape features, including some information on soils and climate, Tier 2 describes farmland areas occurring in Tier 1 in more detail (e.g., altitude and terrain), and Tier 3 describes farm management in more detail. For further information about this dataset, see Goodwin et al. (2022). For this chapter, only Tier 3 spatial data were used. The objective of using the dataset was to identify areas of land for the scenario analysis of reducing Soil emissions in Table 4.1 in line with net zero.

Data was imported into QGIS (version 3.22.8 Białowieża), and the Tier 3 raster was polygonised (converted to vector format). The layer was clipped to each GOR region and Wales, and areas of the polygons were obtained using the field calculator. Tier 3 land classifications were grouped into broad Land Uses (LUs) based on the type of farm management described (see Table 4.2).

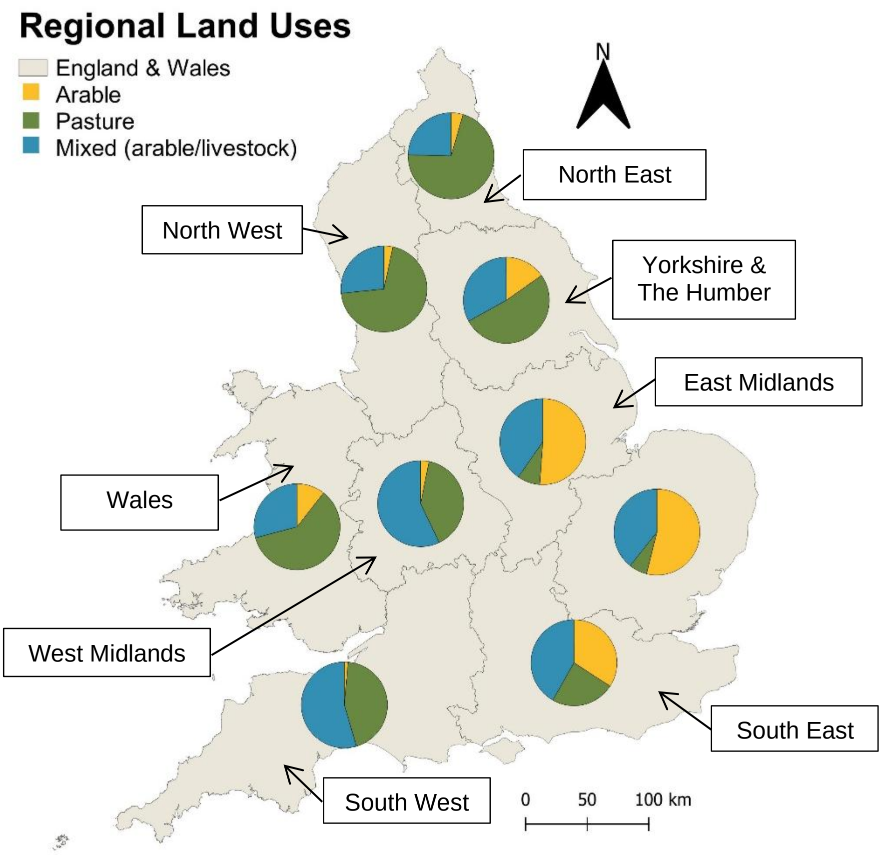

Figure 4.1 Map of England Government Office Regions (GORs) and Wales according to the proportion of main agricultural Land Use (LU), where yellow = arable, green = pasture (livestock only), and blue = mixed (arable and livestock). Data derived from UK CEH multi-tier archetypes dataset (Tier 3; Goodwin et al., 2022).

Table 4.2 Tier 3 Archetypes from Goodwin et al. (2022) and the Land Use (LU) categories used in this analysis.

Broad Land Use (LU)	Tier 3 Archetype
Arable	Broad acre arable; Broad acre vegetable growing
Mixed	Broad acre arable, pigs and poultry; Mixed farming; Beef, sheep and arable farming; Dairy with arable farming
Livestock	Paddocks; Rough grazing; Beef and sheep farming; Mixed pig and poultry farming; Dairy farming; Mixed livestock

Note: Diversified income farming and Agri-tourism Tier 3 land classifications omitted from analysis.

Tier 3 areas were summed to give total area (ha) per LU per region, which would be treated from here on as the available land area for the scenario analyses of farm practices. Two Tier 3 archetypes (diversified income farming and agri-tourism) were excluded as they did not fit into a broad LU and represented only around 7.7% of land area across England and Wales.

Mixed LUs will have some emissions associated with both soils and livestock, however, there is no publicly available data on mixed farming system land area splits (i.e., arable : livestock) so proxy calculations had to be used to estimate this split. The Land Cover Crop dataset for 2020 (CEH, 2020) was imported into QGIS and clipped to the extent of the Mixed LU layer for each region in England and Wales. Descriptive statistics in the software were used to generate area of each crop type in each region of the clipped layer. The LCC dataset included most arable crops, as well as grass, “other crops” and solar panels, the latter two of which were omitted from the analysis as they contributed less than 10% of the broad Mixed LU area. The area of grass in the LCC dataset was used as a proxy for the area of livestock on a mixed farm, although this assumes the land area for livestock housed in buildings is not included. See Table 4.3 for the proportion of arable and livestock land in mixed LU systems by GOR in England and Wales. Mixed LU area for each GOR in England and Wales was then multiplied by the corresponding proportion of arable land in Table 4.3.

Table 4.3 Mixed LU arable and livestock proportions (%). Figures sourced from CEH Land Cover Crops 2020 dataset.

Region	Arable	Livestock
North East	50%	43%
North West	32%	63%
Yorkshire & The Humber	76%	15%
East Midlands	66%	22%
West Midlands	46%	45%
East of England	79%	13%
South East	61%	31%
South West	45%	50%
Wales	5%	94%

Note: Proportions do not add up to 100% as the LCC dataset had other land area data (other crops and solar panels) that was excluded from the analysis.

Likewise for the Livestock LU, emissions would stem from both livestock (enteric fermentation and manure management) and soils (application of fertilisers and manure to grassland and any use of leguminous grass swards), the latter of which would contribute to the Soil emissions figure in Table 4.1. This does not include livestock systems that are completely housed, as there is no publicly available data on areas of land under this management system. To identify the proportion of the Livestock LU area that would be responsible for the Soil emissions average livestock supply chain emissions in Western Europe in 2015 from the FAO Global Livestock Environmental Assessment Model (GLEAM) model were used to identify the typical proportions of emission sources in livestock systems (FAO, 2017b; FAO, 2022). Roughly 31% of livestock emissions stem from enteric fermentation and 15% from manure management, and 9% and 13% of emissions were from manure applications and land-based fertiliser requirements (i.e., Soil emissions), respectively. Therefore, approximately 46% and 22% of emissions were from livestock (i.e., enteric fermentation and manure management) and grassland soils used for livestock, respectively. The 22% figure was then multiplied by the Livestock LU area for each GOR of England and Wales to give the Soil emissions associated with soil-related Livestock LU activities.

4.2.3 Arable farming practices for net zero scenario analysis

Numerous farm management practices to reduce GHG emissions or remove emissions in the agriculture sector have been outlined in Chapter 2 and some of these have been used in this scenario analysis. Table 4.4 shows the GHG reduction rate of each practice tested in the scenario analysis per hectare per year. For the two MacLeod et al. (2010) practices, these are described in Table 3 of their paper as the following:

- Reduced N fertiliser: "*An across the board reduction in the rate at which fertiliser is applied will reduce the amount of N in the system and the associated N_2O emissions.*"
- Avoid excess N fertiliser: "*Reducing N application in areas where it is applied in excess reduces N in the system and therefore reduces N_2O emissions.*"

Although no uncertainty estimates (high/low estimate, standard deviation/error etc.) were reported, the authors did have high expert agreement on the rates presented for these two practices (MacLeod et al., 2010).

For the two practices in the cover crops and legumes strategy, GHG reduction rates were taken from two papers that did have some level of uncertainty reported. Abdalla et al. (2019) estimated net GHG balance by modelling reductions in indirect N_2O and increases in soil carbon sequestration associated with planting mixed legume and non-legume cover crops in arable systems. Smith et al. (2008) provide a more general agronomy-based GHG reduction rate, with evidence from papers that assessed the GHG mitigation efficacy of legume crops and cover crops in arable rotations.

The practices chosen from Chapter 2 that are in Table 4.4 were chosen for this scenario analysis due to the availability of current uptake data from the publicly available Farm Practices Survey 2022 (Table 4.5). Although the survey only captures a portion of the farm businesses presented in England, the dataset has National Statistics status meaning that they are of high quality and robustness. The survey is sent to a representative proportion of the farm holdings in the country (according to farm type and size), with a response rate of 24% (n = 1,453) from eligible holdings in 2022 (see Defra, 2022b). Whilst this doesn't capture uptake for very small holdings, these are likely hobby farms with little agricultural activity and thus unlikely to contribute significantly to regional GHG emissions from agriculture. Wales does not have a Farm Practices Survey, so current level of uptake is unknown and assumed to be <10% as arable farming is not the predominant form of agriculture (11% of land area in the Goodwin et al. (2022) Tier 3 dataset).

Scenario results for each arable mitigation practice are reported for medium (best average), low and high abatement potentials where possible to better understand the uncertainty in mitigation potential range. An absolute increase in practice uptake (i.e., land area undertaking the practice) of 10% was modelled for this chapter as a more conservative scenario analysis, and when considering the current level of uptake in Table 4.5, takes some regions above 40% uptake (East of England).

Table 4.4 Farm practices modelled using scenario analysis to reduce regional Soil emissions. GHG reduction rates shown are reductions in emissions per hectare per year with values in brackets representing a low and high reduction rate estimate; a '-' indicates an increase in emissions.

Strategy	Practice	GHG reduction rate	Reference
Manufactured fertiliser	Reduced N fertiliser	0.5 t CO_2e ha^{-1} yr^{-1}	MacLeod et al., 2010[b]
	Avoiding excess N fertiliser	0.4 t CO_2e ha^{-1} yr^{-1}	
Cover crops and legumes	Legume and non-legume cover crops mix	2.06 (-0.04 – 4.16) t CO_2e ha^{-1} yr^{-1}	Abdalla et al., 2019
	Agronomy improvements, including cover crops and legumes	0.98 (0.51 – 1.45) t CO_2e ha^{-1} yr^{-1}*	Smith et al., 2008

*** The low estimate is a minus (i.e., increase in emission) value because legume plants release some nitrous oxide with regular nutrient cycling (Jensen et al., 2020).**

Table 4.5 Levels of uptake of GHG mitigation practices in England by Government Office Region in 2022. Data from the Farm Practices Survey (Defra, 2022a).

GOR	Improving N fertiliser application accuracy[a]	Increasing use of legumes in arable rotation[a]
North East	43%	5%
North West	28%	21%
Yorkshire & The Humber	41%	17%
East Midlands	51%	14%
West Midlands	42%	27%
East of England	52%	34%
South East	37%	11%
South West	30%	17%

As the two practices modelled for the cover crops and legumes strategy were from two different papers and modelled similar practices, an average Soil emission reduction was taken for each region and then summed to give total Soil emission reduction. On the other hand, the two practices modelled for the N fertiliser strategy could be used simultaneously, so results were summed per region and then summed across regions.

4.2.4 Woodland creation for net zero scenario analysis

There are opportunities for farmers to introduce woodland onto marginal land, or convert arable or pasture land to woodland to increase carbon removals, with the help of woodland creation grants and incentives such as the Woodland Carbon Code (WCC; Woodland Carbon Code, 2022). The WCC is the only certified standard for measuring carbon accumulation in the biomass of new woodland planting regimes for the purpose of selling sequestered carbon as credits. Tree biomass carbon sequestration rates were calculated by taking the average total rate of carbon sequestration (whole tree biomass and debris) for 5 time periods across the range of tree species, spacings, yield classes and management (thinning or no thinning) available in the WCC. Looking at this WCC data, trees appear to reach a peak carbon sequestration rate between 20-30 years post-planting (Figure 4.2), with a subsequent gradual decline in sequestration over time (up to 200 years). For the purposes of this chapter, the following sequestration rates were calculated: 3.1, 14.5, 18.0, 13.5, 9.9 and 2.4 t CO_2e ha^{-1} yr^{-1} for 0-10 years, 10-20 years, 20-30 years, 30-40 years, 40-50 years, and 50+ years since planting, respectively.

A soil carbon sequestration rate of 0.6 t CO_2e ha^{-1} yr^{-1} in the first 50 years since plantation of minimum intervention woodland on mineral soils previously under arable or rotational grassland was used. After 50 years, sequestration is expected to slow to 0.4 t CO_2e ha^{-1} yr^{-1}. These rates were originally taken from UK Short Rotation Forestry research (McKay, 2011), which suggests they are conservative estimates of soil carbon accumulation and are used in the WCC soil carbon guidance (West, 2011). The soil and vegetation carbon sequestration rates were then added together to give a final rate per ha per year (Table 4.6).

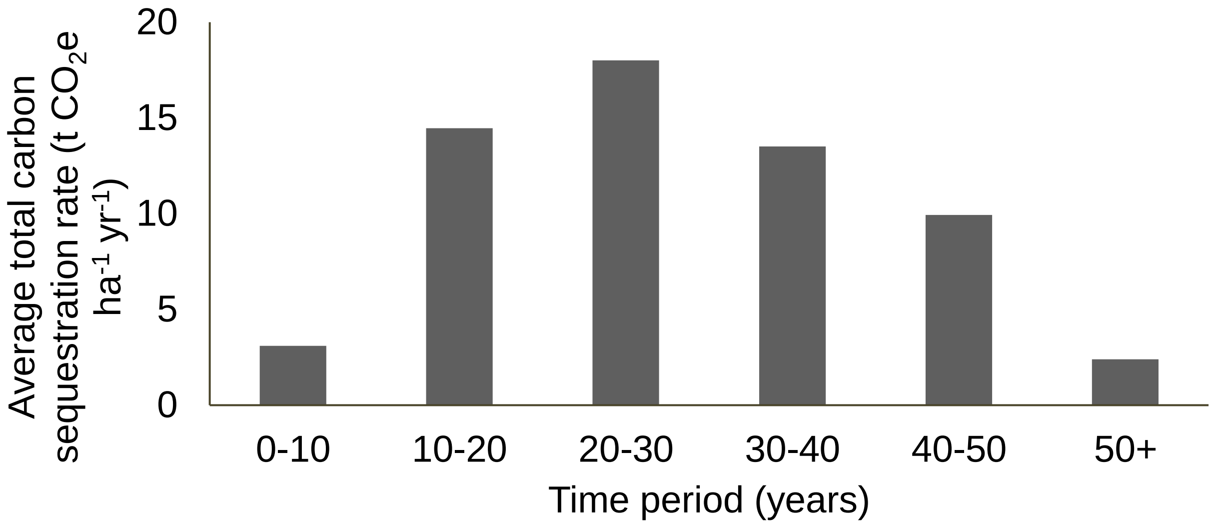

Figure 4.2 Tree biomass carbon sequestration rates through time. Data from the WCC (2022) representing average total carbon sequestration rate across tree species, spacings, yield classes and management (thinning/no thinning).

To determine the possible land area in each region for woodland creation, the Forestry Commission's *Woodland Creation Full Sensitivity Map v4.0* (2024c) was used. This map provides spatial data on low, medium, and high sensitivity areas for possible woodland creation in England only. The low sensitivity methodology includes 'agricultural land' as a broad category, which means that more productive and fertile Agricultural Land Classifications (ALC) grade 1, 2 or 3a land could be considered for woodland creation. For this chapter's analysis, medium sensitivity land was chosen (as shown in Figure 4.3) as this considered a wide range of land types to be more critical than for low sensitivity, including ALC of 1, 2 or 3a, and removed key land features such as National Parks, Sites of Special Scientific Interest and peatland. Full methodology can be found on the dataset page (Forestry Commission, 2024a).

As there was a high amount of available land from the medium sensitivity woodland map (5.4 Mha), the scenario analysis in this chapter focussed on a one-off 1% increase in afforested land in each region assuming planting occurred in 2025, with cumulative carbon reported in 5-year increments up to 2050. 1% of land was chosen as a conservative scenario given that woodland area in England and Wales has only increased 8% and 5% since 1998, respectively (Forest Research, 2024).

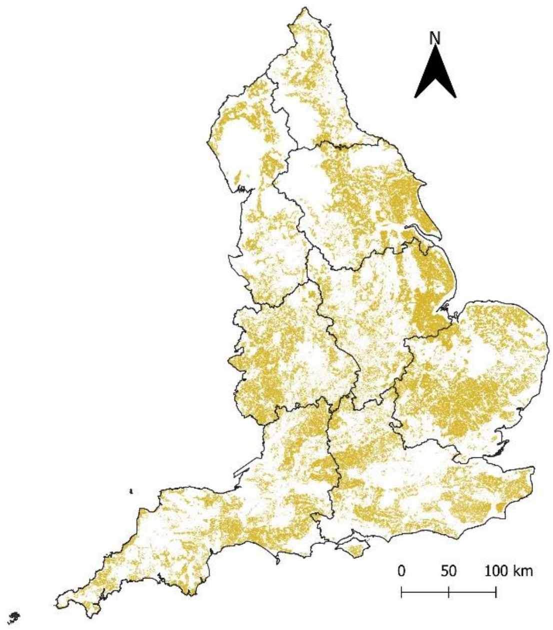

Figure 4.3 Map of medium sensitivity woodland creation opportunities in England (gold) with outlines of the Government Office Regions. Data from Forestry Commission (2024).

Table 4.6 Carbon sequestration potential for woodland creation (biomass carbon and soil carbon) to offset regional Soil emissions.

Practice	Carbon removal potential	Reference
Woodland creation on former arable land	0-10yrs: 3.7 t CO_2e ha^{-1} yr^{-1} 10-20yrs: 15.0 t CO_2e ha^{-1} yr^{-1} 20-30yrs: 18.6 t CO_2e ha^{-1} yr^{-1} 30-40yrs: 14.1 t CO_2e ha^{-1} yr^{-1} 40-50yrs: 10.5 t CO_2e ha^{-1} yr^{-1} 50+yrs: 2.8 t CO_2e ha^{-1} yr^{-1}	McKay, 2011; Woodland Carbon Code, 2022

4.3 Results

4.3.1 Area of land available for arable scenario analysis

An estimated total of 7,859,477 ha of land across England and Wales was available for reducing emissions from Agricultural Soils (Table 4.7), which was 7,492 kt CO_2e on average (2018 – 2020), according to the Tier 3 archetype dataset. Exploring the land area available, each GOR of England and Wales has a different amount of land available for mitigation practices, which broadly follows the East/West divide seen in agricultural practices.

The East Midlands and East of England had the most land available (across the Arable LU, arable Mixed LU, and grassland in the Livestock LU) for the scenario analysis at 1.9 million ha each, whereas the North East and North West had just over 230,000 ha available.

Out of the three types of land available for the analysis, Arable LU, arable Mixed LU, and grassland in the Livestock LU, the grassland represented the smallest land area (1.1 Mha), followed by arable land in Mixed LUs (3.3 Mha) and solely arable LU land (3.5 Mha).

Table 4.7 Available land area (ha) in England and Wales for reducing Soil emissions.

	Land area available for tackling Soil emissions			
Region	**Arable LU (ha)**	**Livestock LU grassland (ha)**	**Mixed LU arable (ha)**	**Total**
North East	31,925	111,513	88,432	**231,870**
North West	29,121	133,528	73,704	**236,352**
Yorkshire & The Humber	191,419	143,281	315,114	**649,814**
East Midlands	1,213,310	44,341	633,868	**1,891,519**
West Midlands	43,034	119,352	356,986	**519,372**
East of England	1,157,123	32,226	664,456	**1,853,805**
South East	703,822	109,127	525,673	**1,338,622**
South West	33,614	240,142	603,894	**877,650**
Wales	109,044	137,302	14,126	**260,472**
Total	**3,512,412**	**1,070,812**	**3,276,253**	**7,859,477**

4.3.2 Arable scenario analysis

The following sections demonstrate the regional abatement potential of improving the use of manufactured N fertiliser and introducing legumes into arable rotations, and how much regional uptake would be required year-on-year to reach net zero emissions by 2050. Appendix B contains the full range of uptake scenario models.

Improving the use of manufactured N fertilisers

Using arable land from Arable LU, arable Mixed LU land and grassland from the Livestock LU, two scenarios around improvements to manufactured N fertiliser use were tested. As can be seen from Table 4.5**,** overall uptake of N fertiliser management practices was high across England, but there were regional differences. 52% of farmers in the East of England reported they were already improving N fertiliser applications, whereas in the North West 28% of farmers were doing this practice. Evidently, the greater the uptake of the practice, i.e., the greater the land area the practice is applied to, the larger the mitigation potential (unless a practice has minor emissions at the lower abatement level), so not all uptake percentages were reported.

The two scenarios tested resulted in between a 1 and 10% reduction in Soil emissions across England and Wales with a 10% increase in uptake of practices in the first year. Reducing the use of N fertiliser in the East Midlands and South East led to a 10% reduction in annual Soil emissions (95 and 67 t CO_2e, respectively) based on the 2018 to 2020 baseline. When combined with the other practice of avoiding excess N fertiliser being used, the Soil emission reduction increases to 18% in both regions. The lowest mitigation potential was in Wales at only 3% (both practices) (

Figure 4.4).

Total Soil emission reduction from a 10% increase in uptake of improved N fertiliser use practices across the regions was 9% (from the 2018 – 2020 baseline), equating to 707 kt CO_2e. If the uptake was another 10% higher (i.e., 20% uptake on top of current uptake), particularly in the South East where current uptake is lowest of the arable-dominant regions at 37%, then Soil emissions for the region could be reduced by 38% rather than 18%, which would cut average annual national level Soil emissions by 828 kt CO_2e (11%).

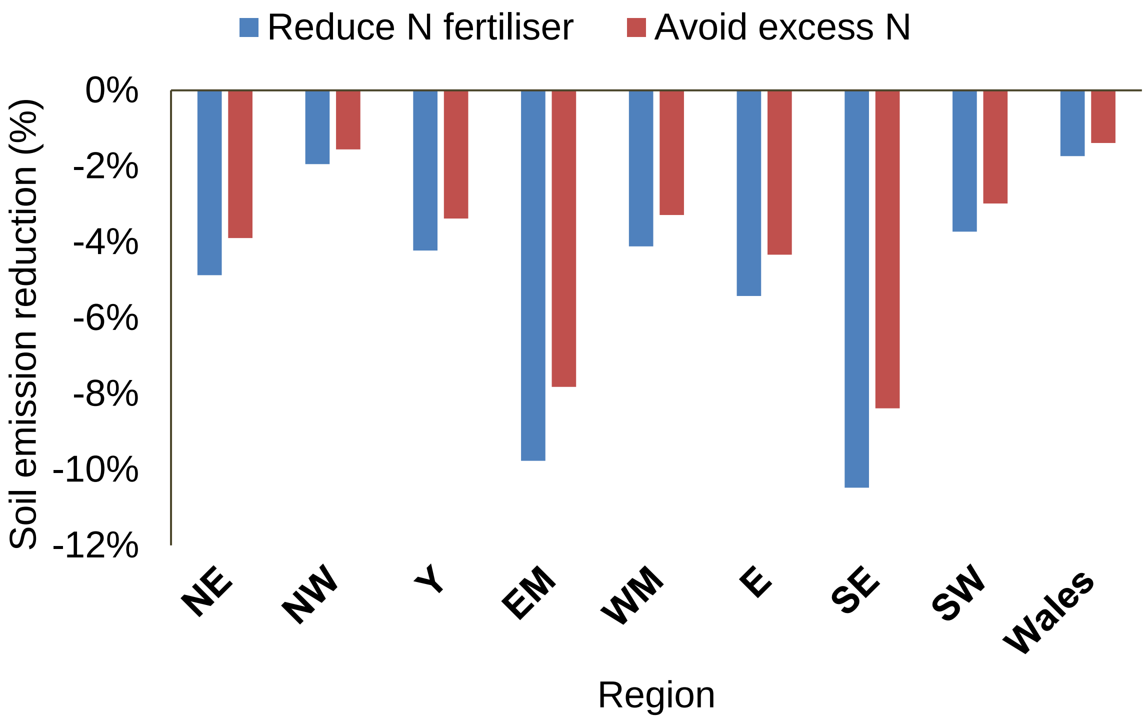

Figure 4.4 Percentage reduction of Soil emissions from a 10% uptake of practices improving the use of manufactured N fertilisers by either reducing N fertiliser (blue) or avoiding excess N use (orange) for each region.

Introducing legumes into the arable rotation

Using arable land from Arable LU and arable Mixed LU land, two scenarios using different GHG abatement potentials were tested in relation to the introduction of legume crops into arable rotations. As in the previous scenario, when looking at Table 4.5 there were regional differences in the level of uptake of the use of legumes in arable rotations. For example, 34% of farmers in the East of England reported they were already using legumes in the rotation, whereas in the North East only 5% of farmers were doing this practice.

For the purpose of these results, the medium (or best average) scenario findings are reported with the low and high results in brackets. The two scenarios achieved between a 2% and 40% reduction in Soil emissions across regions with a 10% increase in uptake of the practices from current levels. The greatest emission reduction of 40% (1% increase to 80% reduction) was seen in the South East, where 253 t CO_2e would be cut through implementation of legume and non-legume mixes into the arable rotation (Table 4.4; Figure 4.5). The East Midlands had a similar result with a medium reduction in Soil emissions of 39% (1% increase to 79% reduction). Looking at the other practice, improving agronomic practices such as through using cover crops, both regions had a 19% (10% to 28%) reduction in Soil emissions (Figure 4.6). The second practice had lower emission reductions overall as the abatement potential was 0.98 t CO_2e ha^{-1} yr^{-1} compared to 2.06 t CO_2e ha^{-1} yr^{-1} for the medium scenario (Table 4.4).

Total Soil emission reduction from a 10% increase in uptake of practices introducing legumes into the arable rotation across the regions was 14% (from the 2018 – 2020 baseline), equating to 1,032 kt CO_2e.

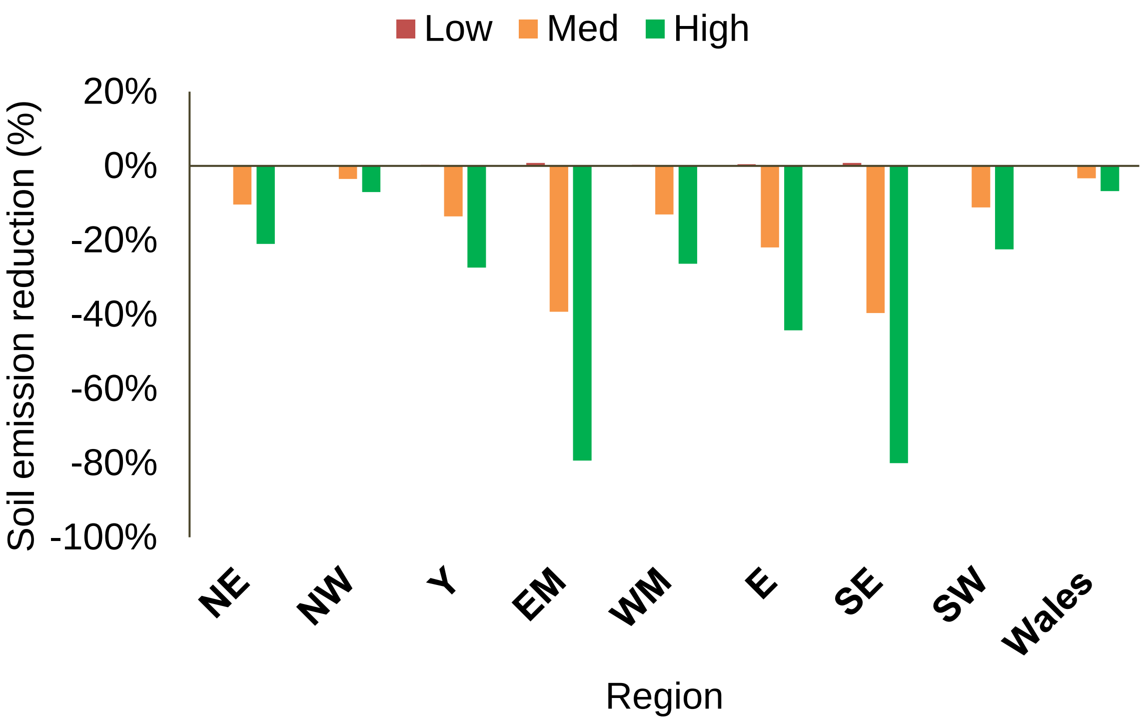

Figure 4.5 Percentage reduction of Soil emissions from incorporating a mixture of legume and non-legume crops in arable rotation (Abdalla et al., 2019) for three uncertainty levels: Low (red), Medium (yellow) and High (green). The uncertainty levels correspond with the values in Table 4.4, with medium being the average, and low and high being the lower value and higher values in brackets.

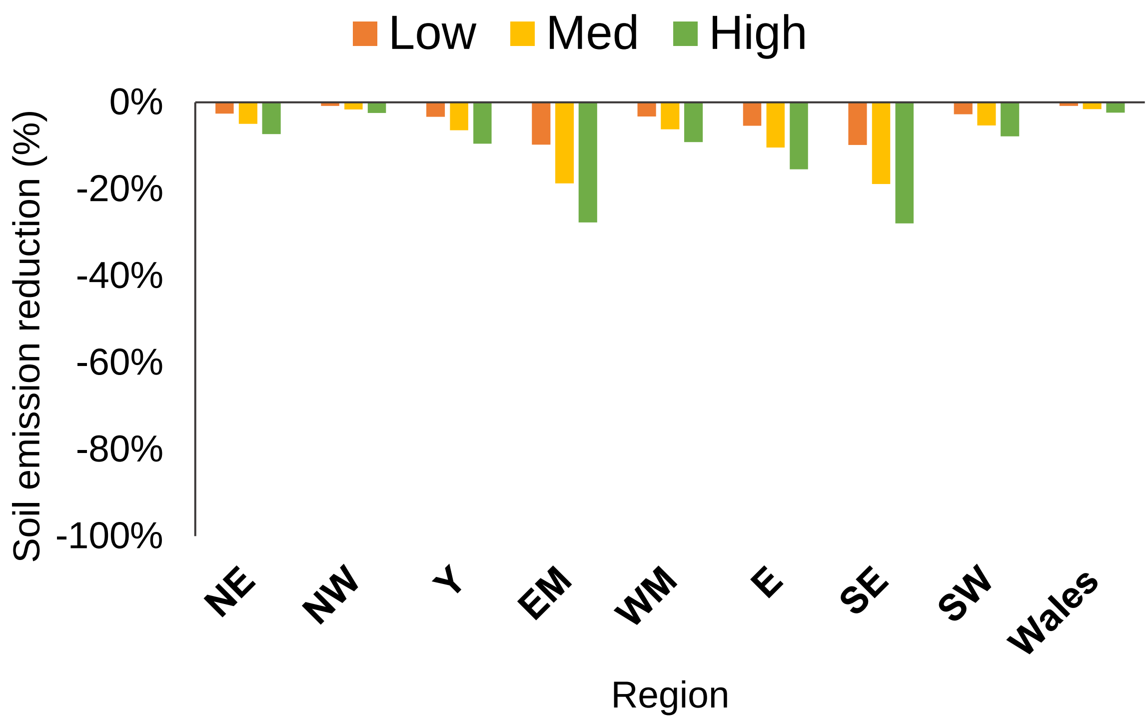

Figure 4.6 Percentage reduction of Soil emissions from improvements to agronomic practices (Smith et al., 2008), including use of cover crops and legumes in arable rotations, for three uncertainty levels: Low (red), Medium (yellow) and High (green). The uncertainty levels correspond with the values in Table 4.4, with medium being the average, and low and high being the lower value and higher values in brackets.

Net zero assessment using both strategies

The combined annual average Soil emission reduction from the two strategies modelled above, compared to the 2018 – 2020 baseline, was 23% (1,739 kt CO_2e). Soil emission reductions from each strategy and the combined strategies in this chapter compared to the baseline can be seen in Figure 4.7. It is clear that a 10% uptake of the practices assessed in this chapter alone is not enough to help arable farms reach net zero by 2050. Therefore, to reduce the annual average Soil emissions further in line with net zero, there either needs to be greater uptake of these practices within the technical limit (i.e., where lack of nitrogen becomes a limiting factor to high yields, or legume cover crops aren't able to establish in particular soils), or greater diversity of GHG reduction practices modelled that farmers can take up.

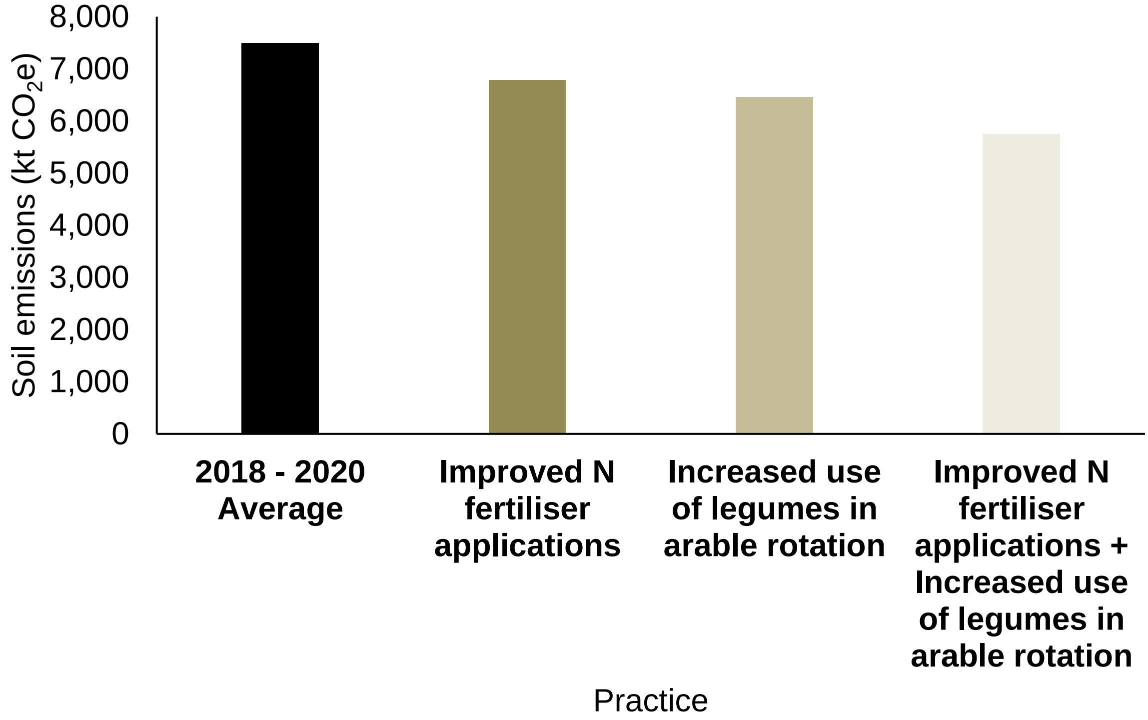

Figure 4.7 Modelled strategies for reducing annual average (2018 – 2020) Soil emissions in arable systems for England and Wales (see Table 4.1).

4.3.3 Woodland creation scenario analysis

A total of 5.4 Mha land was available in England for woodland creation according to the Forestry Commission's *Woodland Creation Full Sensitivity Map v4.0* (2024c) under the medium sensitivity scenario. The majority the suitable available land was in the South East (20% of the total available area). The North East had the smallest land area available for woodland creation at 4% of the area (Table 4.8).

Table 4.8 Land area available for woodland creation under the medium sensitivity scenario. Data from the *Woodland Creation Full Sensitivity Map v4.0* (Forestry Commission, 2024c).

Region	Area (ha)	1% of area
North East	192,808	1,928
North West	280,810	2,808
Yorkshire & The Humber	715,542	7,155
East Midlands	930,984	9,310
West Midlands	594,182	5,942
East of England	728,905	7,289
South East	1,071,750	10,718
South West	871,494	8,715
TOTAL	**5,386,475**	**53,865**

The total of 1% of the land area in each region of England in Table 4.8 is 53,865 ha, which is higher than the CCC's recommendation of 50,000 ha per year from 2035 to 2050 (CCC, 2020a; CCC, 2020b). Accounting for the specific sequestration rates outlined in Table 4.6, if new woodland was created on a one-off 1% of the available land area in each region in 2025, then in the first five years (i.e., up to 2030), 996 kt CO_2e would have been sequestered in tree biomass and soils. This equates to roughly 200 kt CO_2e per year in the first 5 years. By 2050, a total of 15,082 kt CO_2e or 15 Mt CO_2e would be accumulated in the 53,865 ha of afforested land across England (Table 4.9), assuming no woodland management or establishment failures. The breakdown of regional contributions to this are also found in Table 4.9, which illustrates the 5-year periodic changes in carbon accumulation. As the land area available for afforestation (from the medium sensitivity map) is the main factor causing differences in carbon accumulation between regions, the findings are the same as above with the South East having the greatest accumulation over the 25-year period.

Looking at these results, the amount of carbon accumulated in woodland biomass and soils on 1% of land in England by 2040-2045 would balance out the 2018 to 2020 average annual Soil emissions of 7,492 kt CO_2e. For 1% land area afforestation (53,865 ha) to remove enough carbon to balance out the average annual Soil emissions (7,492 kt CO_2e) in England and Wales each year, rather than in the next 15 years, then each hectare would need to be sequestering 139 t CO_2e each year. This indicates that the one-off 1% afforestation scenario may not be enough to meet net zero targets unless the quantity of annual Soil emissions is reduced, or more than 1% of land is afforested and preferably as mixed-age plantings owing to the differences in carbon sequestration rate outlined in Table 4.6. However, there are important implications of whether carbon removals can be used to offset non-CO_2 GHGs, which is discussed in Section 4.4.

Table 4.9 Carbon accumulation in vegetation and soil (kt CO_2e) from afforestation of 1% suitable land area in 2025. Cumulative carbon is reported in 5-year increments between 2025 and 2050.

Region	Period				
	2025 - 2030	**2030 - 2035**	**2035 - 2040**	**2040 - 2045**	**2045 - 2050**
North East	36	71	216	361	540
North West	52	104	315	525	786
Yorkshire & The Humber	132	265	801	1,338	2,004
East Midlands	172	344	1,043	1,741	2,607
West Midlands	110	220	665	1,111	1,664
East of England	135	270	816	1,363	2,041
South East	198	397	1,200	2,004	3,001
South West	161	322	976	1,630	2,440
TOTAL	**996**	**1,993**	**6,033**	**10,073**	**15,082**

4.4 Discussion and conclusions

One of the main purposes of this chapter was to identify potential GHG reduction options from farming practices at regional and national scale, and to understand how this might impact national ambitions of net zero GHG emissions by 2050 for the agriculture sector. Using the two strategies and the two practices within each strategy in Table 4.4, it has been possible to determine the potential GHG reduction from improving manufactured N use and introducing legumes/cover crops into arable rotation, and how this impacts the total regional soil GHG footprint (Table 4.1), and thus the national soil GHG footprint.

4.4.1 Improving the use of manufactured N fertilisers

There was a modest reduction in soil GHG emissions (1-10%) from the implementation of practices improving the use of manufactured N fertilisers regionally in England and Wales, with the East Midlands and South East showing the greatest potential. Combinations of practices reduced Soil emissions further and it was shown that targeting arable-dominant regions where N fertiliser use is likely to be higher, like the South East, can reduce regional and national soil GHG emissions by another 2%.

When modelled by MacLeod et al. (2010) in a marginal abatement cost curve, the practice of avoiding excess manufactured N use ranked higher than simply reducing N fertiliser overall, despite the abatement rate being higher for the latter (Table 4.4), as it had a negative cost (i.e., saved money). In this chapter, the analysis was simply assessing GHG abatement potential, so the practice of avoiding excess N use ranked lower than the alternative. Both practices have been modelled separately and in combination in this analysis as they form key parts of optimal crop nutrient management (AHDB, 2023a) and reduce reliance on manufactured N, which has embedded emissions (Brentrup et al., 2018). The 10% uptake modelled in each region assumes that some farms will be using nitrogen fertiliser in excess of industry standards (Sylvester-Bradley and Kindred, 2009) and other farms will be able to reduce their manufactured nitrogen use through supplementing with alternative nitrogen sources (e.g., organic manures, compost). However, the impact of choosing an alternative nitrogen source was not assessed in this analysis, and future research should consider the trade-offs with swapping nitrogen sources to reduce manufactured nitrogen contributions to global warming.

The scenario analysis used a flat 10% increase in uptake in each practice across regions, as well as a 20% increase in uptake of practices in the South East due to the lower current uptake levels. However, farming regions in the east of England are predominantly arable (Figure 4.1), so efforts to improve the use of manufactured N fertilisers in general should be prioritised in the east to achieve the greatest national impact. However, as can be seen in Table 4.5, the current level of uptake of nitrogen management practices on-farm is generally over 30% of the region, so the technical potential for these practices may be limited by economic and agronomic factors limiting farmers from applying non-manufactured nitrogen sources.

4.4.2 Introducing legumes into the arable rotation

Both the use of a combination of cover crops (legumes/non-legumes/mixed) and the general agronomy practice (legumes and cover crops) were effective at reducing regional Soil emissions in the South East and East Midlands at the 10% uptake level.

Cropland areas sowing a mixture of legume and non-legume cover crops would see greater net GHG abatement through the reduced requirement for manufactured N fertiliser and increased soil carbon storage, representing a win-win scenario for farmers throughout England and Wales (Smith et al., 2008; MacLeod et al., 2010). The GHG mitigation potential of cover cropping is limited in soils that have higher soil carbon content, such as grasslands, as saturation may be reached earlier than in arable soils (Powlson et al., 2011), so there is greater abatement potential in eastern regions of England.

Abdalla et al. (2019) suggest from their meta-analysis of cover crop research that the grain yield of the main crop could be impacted by the type of cover crop sown. For example, grain yield was reduced by approximately 4% in a non-legume cover crop system, although this effect was reversed if a legume and non-legume mix was used with yields increasing by ~ 13%. This has important implications for farmers, as any new practice taken up should not negatively impact the farm business, which relies on productivity. Furthermore, cover cropping requires the purchase and sowing of additional plants, which means that additional, and sometimes specialised, machinery use and potentially plant protection is needed (Paustian et al., 2001; Smith et al., 2007). This can increase the cost of overheads for farmers and may not be the most profitable scenario for reducing and removing GHG emissions from farm production.

4.4.3 Woodland creation

Using the scenario of a one-off instance of 1% of land area in England being afforested and modelling the carbon accumulation between 2025 and 2050, this analysis observed that it would take roughly 15 years for the afforested land to be removing as much carbon dioxide equivalent as the annual average Soil emissions between 2018 and 2020. By 2050, 15 Mt CO_2e would be accumulated in this woodland area, which is more than double the annual average Soil emissions. One challenge with this finding is that this analysis is comparing cumulative carbon sequestration over time with an annual Soil emission. For example, the average annual Soil emission (2018 to 2020) for England and Wales as a whole is 7,492 kt CO_2e and this analysis has shown that in the first five years since afforesting 1% of the land area the cumulative tree biomass and soil carbon sequestered was 996 kt CO_2e, which equates to roughly 200 kt CO_2e per year (although this increases after 10 years and then 20 years as per Table 4.6. This is representing around 3% of the annual average Soil emissions shown in Table 4.1. What this means is that either more than 1% of land area would need to be afforested in the first place, or each year another 1% of land is afforested as per the CCC's recommendation to Government, to ensure maximum feasible cumulative sequestration from those trees in a single year. The other main challenge with this research is that although the CO_2e numbers might be balanced 15 years after a one-off 1% afforestation scenario, this doesn't necessarily equate to the same climate and temperature effects. For example, research from New Zealand has demonstrated that to truly offset the temperature effects of methane from ruminant animals, millions of hectares of land would need to be afforested. For the dairy herd alone, 3.8 million hectares of new pine forest would need to be established, which equates to roughly 14% of the land area, which would impact other land uses (Parliamentary Commissioner for the Environment, 2022). Therefore, an important implication of this work is that although afforestation is being used as a national-level strategy for net zero efforts (CCC, 2020a), the climate change mitigation benefits of woodland are unlikely to be noticeable at the 1% land area scale.

The map used in this analysis was medium sensitivity, which means that only land that fell outside of ALC 1, 2 or 3a (as well as other non-ALC conditions) was considered. It is important that high agricultural productivity land (ALC 1, 2 and 3a) remains agricultural to maintain and improve the productive output of England and Wales (Staddon et al., 2021), which aligns with the land sparing approach. However, more marginal farmland, including field corners, awkwardly shaped fields and unproductive land can be converted to woodland to maximise land use effectiveness, improve water and air quality and importantly, remove CO_2 from the atmosphere (Staddon et al., 2021; Flack et al., 2022; Westaway et al., 2023).

The analysis found an 878,942 ha disparity between the region with the most available land area for afforestation (South East) and the area with the least area available (North East). The flat 1% of land area afforested scenario presented in this chapter may not be applicable to every region in England and the map used to identify area of land available did not present data for Wales, which could have further land area available for afforestation. On the former point, woodland planting has been a policy incentive for farmers in England through Countryside Stewardship, but with the new England Woodland Creation Offer (EWCO), farmers and landowners can be paid £1,100 - £3,300 per hectare to plant woodland on-farm (Forestry Commission, 2024b). This was updated in March 2024 to incorporate new ‘low sensitivity’ land payments, referring to the low sensitivity land areas in the *Woodland Creation Sensitivity Map* (Forestry Commission, 2024c). Whilst this chapter’s analysis looked at medium sensitivity woodland creation opportunities to be more cautious, low sensitivity land in England is estimated to be around 9 Mha, 3.5 to 4 Mha more land than what was considered available in this analysis. Farmers and landowners may be more receptive to woodland creation if more of their land is considered suitable and will be incentivised, which could exceed afforestation targets, such as the CCC’s 50,000 ha between 2035 and 2050 (CCC, 2020a; CCC, 2020b).

As the *Woodland Creation Full Sensitivity Map* (Forestry Commission, 2024c) was only released recently, there isn’t any academic literature to directly compare the results of this analysis to. However, research from Bradfer-Lawrence et al. (2021) shows an early iteration of this work as the authors looked at ’lower risk mineral soil’ and ‘higher risk organo-mineral soil’ land area available throughout the UK. The authors found that 4.6 Mha of land would be available for afforestation in the UK, 1.6 Mha of which would come from England (Bradfer-Lawrence et al., 2021), which is lower than the estimates in the Forestry Commission map used in this chapter. Another similar piece of research looked at afforestation targets and land area available in the UK for afforestation and found that ALC grades 1-3 were the biggest constraint on afforestation potential in England covering 65% of land area. The authors found similar results to Bradfer-Lawrence et al. (2021), where 4.7 Mha of land would be available throughout the UK, 1.2 Mha of which was in England (Burke et al., 2021).

Finally, whilst this chapter assessed the efficacy of afforestation for the removal and storage of carbon it did not look at the more holistic impact of planting new woodland on the wider ecosystem. Woodland delivers numerous co-benefits alongside carbon sequestration and storage; for example, water and air quality regulation, improvements in soil health through better nutrient cycling and reduced soil erosion, and biodiversity (Staddon et al., 2021; Bateman et al., 2023). Bateman et al. (2023) argue that woodlands can be ‘net zero plus’, meaning that afforestation should be done with more than just climate change mitigation in mind, with active thought on how to improve biodiversity, soils, and cultural or social benefits through afforestation. Recent research modelling the re-introduction of wolves into expanding woodland areas in Scotland found that the reestablishment of the predator-prey cycle between wolves and red deer could lead to a 1 Mt CO_2e per year sequestration rate due to the reduced herbivory of tree saplings (Spracklen et al., 2025). However, afforestation also comes with potential ecosystem risks and some species could end up displaced or fragmented from other populations unless new woodland is created with the right tree in the right place (Bateman et al., 2023).

4.4.4 Conclusions

The purpose of this chapter was to estimate the amount of land area available for emission reduction and the potential GHG abatement or carbon removal from arable practices or woodland at a regional scale, respectively. The findings of which could then be scaled up to national scale to better understand how these practices would impact net zero emission targets. This research has shown that 7.9 Mha land (Table 4.7) was available across England and Wales to reduce emissions from agricultural soils (linked with arable practices) in the baseline period of 2018 to 2020. A 23% reduction in annual average Soil emissions (2018 – 2020) could be achieved with a combination of the two arable strategies at a 10% increase in uptake of practices across England and Wales. Hypothetically, it would take up to five years of this linear 10% increase in uptake to remove the same amount of the annual average Soil emissions, i.e., net zero. However, the technical possibility of this is acknowledged to be limited due to the fact that reducing nitrogen applications too much can lead to yield losses, so it is unlikely that 50% of farms would be able to achieve this. A key implication of this work is that it is still not clear what individual practices and combinations of practices are required to meet net zero goals, but scenario analysis studies such as this one can help to evidence the lower uptake and more realistic net zero opportunities, as well as the more 'blue sky', high uptake hypothetical scenarios that could increase policymaker ambition.

Afforestation of 1% of medium sensitivity land in England would accumulate around 200 kt CO_2e in the year following the one-off planting, accounting for 3% of average annual Soil emissions (2018 – 2020), with most of this occurring the South East. If looking at the woodland sequestration potential over the next 25 years, up to the UK's net zero timeframe of 2050, sequestration would increase nonlinearly due to differences in carbon removal rates over a tree's lifespan, with the last five years of this period having the greatest sequestration rate (18.6 t CO_2e ha^{-1} yr^{-1}; Table 4.6). This means that a one-off planting event in 2025 of 1% land area with new woodland would accumulate 15 Mt CO_2e across England by 2050, and it would take 15 years (i.e., 2040-2045 period) for the woodland area to have accumulated the same amount of carbon as the 2018 to 2020

Soil emissions annual average. The implication of this is that a one-off 1% woodland creation scenario is not ambitious enough to meet net zero goals, and from other research assessing the potential to use afforestation as a method to 'offset' non-CO_2 GHG emissions (Parliamentary Commissioner for the Environment, 2022), this may not even put a dent in the temperature-relative carbon removals required to reach net zero.

Chapter 5 - Strategies for livestock agriculture in England and Wales to contribute to net zero emissions

5.1 Introduction

Greenhouse gas (GHG) emissions from the production of livestock products (e.g., meat, milk, eggs, wool) were around 12% of total anthropogenic GHG emissions in 2015 (FAO, 2022; FAO, 2023c). The majority of livestock GHG emissions are methane (CH_4) from enteric fermentation in ruminant animals, and managed liquid manures (Smith et al., 2021; Arndt et al., 2022; Beauchemin et al., 2022). CH_4 forms under anaerobic conditions by methanogenic bacteria, which are found in abundance in the rumen of cattle and sheep, in mostly wet stored manure systems and in rice paddies (Smith et al., 2021). The agriculture sector was the largest global contributor of CH_4 emissions in 2019 at roughly 142 million tonnes (Mt) (International Energy Agency [IEA], 2024a), with a combination of enteric fermentation and manure management contributing 30% of those emissions (Reisinger et al., 2021), meaning the climate impact of livestock production is also significantly large.

The currently accepted Intergovernmental Panel on Climate Change (IPCC) methodology describes the Global Warming Potential of CH_4 over 100 years (GWP_{100}) to be able to compare all GHGs and report them in carbon dioxide equivalents (CO_2e). As explored in Chapter 2, CH_4 is a Short Lived Climate Pollutant (SLCP) and breaks down in the atmosphere to CO_2 and water vapour after roughly a decade, although has almost a 30-times greater radiative forcing on the climate system than CO_2 over 100-years (Shine, 2009; Allen et al., 2022). This means that if CH_4 emissions were to stabilise or be reduced, then over the next 10 years the warming effect from CH_4 on the atmosphere would also stabilise or decrease, the latter of which causing a slight ‘cooling’ effect as the concentration of atmospheric CH_4 reduces (Allen et al., 2018; Cain et al., 2019; Lynch et al., 2020). However, the time-integrated radiative forcing over 100 years applied to all GHGs is not representative of the SLCP nature of CH_4, and a new methodology has been suggested to more accurately accounts for the emission rate of CH_4 – GWP* (Lynch et al., 2020). This means that where CH_4 reduction practices (e.g., in livestock production systems) are occurring, which would reduce the atmospheric concentration of CH_4, both the emission and subsequent climate impact of the practice is more accurately accounted for.

In order to keep global warming to 1.5°C, global CH_4 emissions from livestock production need to reduce and some scenarios suggest that a 24-47% reduction by 2050 could help to deliver that (Arndt et al., 2022). There are several global pledges and commitments, and national policies with the aim of reducing the environmental impacts of livestock production. The Global Methane Pledge was initiated at COP26 in 2021 and strives to achieve a 30% reduction in global CH_4 emissions (from 2020 levels) by 2030 (Malley et al., 2023). The UK is signed up to the Pledge, however, China, India and Russia are yet to join but contribute 39 million tonnes (Mt) agricultural CH_4 or 28% of global CH_4 emissions from agriculture (IEA, 2022; IEA, 2024b). In the UK, the policies and targets focusing on livestock production are related to reducing ammonia emissions and nitrate pollution, which impact air quality and water quality, respectively, and indirectly lead to nitrous oxide emissions (Defra, 2019b; Defra, 2021b). Aside from the UK's contribution to the Global Methane Pledge, there are no CH_4 reduction-specific policies yet for the agriculture sector in England and Wales, so any contribution towards net zero emissions is voluntary for livestock farmers.

5.1.1 Research gap

There is a growing body of reviews and meta-analyses looking at strategies to reduce methane emissions from livestock, with two commonly studied options being reducing enteric methane from ruminants (Hristov, et al., 2013; Arndt et al., 2021; Beauchemin et al., 2022) and optimising manure management (Chadwick et al., 2011; Petersen et al., 2013; Montes et al., 2013). Looking at the CH_4 emissions in the UK GHG Inventory, enteric fermentation from ruminants constitutes around 85% of the agriculture emissions, and manure management is 15% of the footprint. Looking at each pathway separately, beef and dairy cattle contribute 77% of enteric methane emissions, and dairy cattle and pigs contribute 81% and 14% of manure waste emissions, respectively (Brown et al., 2023).

Therefore, this chapter focusses on conducting scenario analyses of enteric fermentation reduction and manure management practices, which centre around a key GHG, methane. This chapter aims to assess the potential for reducing methane emissions from livestock in England and Wales to align with net zero ambitions. This aligns with research aim 3 outlined in Chapter 1, which was to investigate the regional-scale opportunities for livestock production to contribute to net zero policy.

5.2 Methods

5.2.1 UK GHG Inventory - regional profiles

In the same UK GHG Inventory and supplementary dataset described in Chapter 4, livestock emissions were averaged over three years (2018 – 2020) to provide a baseline for the mitigation scenarios across the English Government Office Regions (GORs) and Wales (BEIS, 2022b). Table 5.1 provides the 3-year average livestock emissions taken from the UK GHG Inventory and local authority supplementary dataset for each GOR and Wales. 'Agriculture Livestock' (referred to as Livestock emissions henceforth) emissions from the UK GHG Inventory include enteric fermentation and manure management emissions, as well as some emissions from excreta deposited on soils during grazing and the spreading of manures (BEIS, 2022).

Table 5.1 Regional annual average (2018 – 2020) agricultural Livestock emissions (kt CO_2e), and the enteric fermentation, manure management and excreta deposition/spreading emissions that constitute Livestock emissions, from the UK GHG Inventory regional and local authority dataset (BEIS, 2022b).

GOR	Livestock emissions	*Enteric fermentation*	*Manure management*	*Excreta deposited/spread*
North East	883	*664*	*72*	*147*
North West	3,057	*1,984*	*290*	*784*
Yorkshire	2,098	*1,202*	*327*	*569*
East Midlands	1,702	*894*	*185*	*623*
West Midlands	2,250	*1,407*	*233*	*610*
East of England	1,441	*371*	*217*	*853*
South East/London	1,388	*751*	*120*	*518*
South West	5,108	*3,169*	*504*	*1,435*
Wales	3,986	*3,414*	*349*	*222*

TOTAL England & Wales	**21,913**	*13,855*	*2,297*	*5,761*

5.2.2 Livestock population and emissions data

Livestock emissions in Table 5.1 were disaggregated into emissions from enteric fermentation, emissions from manure management and emissions from excreta deposited on soils during grazing and the spreading of manures using livestock population (Appendix C) and CH_4 data (Table 5.2) for those emission sources was needed.

Information on the number of individual livestock animals in each county was publicly available from the June Agricultural Survey. This was aggregated to GOR level in England and the whole of Wales for this study for the latest available reporting year of 2021 (see Appendix C; Defra, 2024). The data was fairly granular, reporting the number of animals in distinct classes (e.g., age and sex), although for the purpose of this chapter these were aggregated into beef cattle, dairy cattle and pigs, sheep, and poultry.

Average emissions of methane from enteric fermentation (kg CH_4 head^{-1} year^{-1}) and manure management (kg CH_4 head^{-1} year^{-1}) for each livestock type were taken from the UK GHG Inventory (Brown et al., 2023) and can be found in Table 5.2.. These values were then multiplied by the GWP_{100} value of 27.2 from IPCC AR6 (IPCC, 2021) to get CH_4 in kg CO_2e head^{-1} yr^{-1}. The values in Table 5.2 were multiplied by the population data in Appendix C and summed to get emissions just from enteric fermentation and manure management across livestock species in each region only, which can be found in Table 5.1. This provided a baseline figure for measuring the reduction in enteric fermentation and manure management CH_4 emissions, although reductions were also measured against the whole Livestock emissions figures in Table 5.1.

Table 5.2 Average methane emissions (kg CH_4 head^{-1} year^{-1}) from enteric fermentation and manure management for different livestock categories[a]. Data from the UK GHG Inventory (Brown et al., 2023).

Livestock	Average enteric fermentation (kg CH_4 head^{-1} yr^{-1})	Average manure management (kg CH_4 head^{-1} yr^{-1})
Dairy	68.48	13.57
Beef	53.88	6.74
Sheep	6.14	0.16
Pigs	1.50	4.03
Poultry	0.00	0.05

[a] Enteric methane and manure methane emissions from the UK GHG Inventory were averaged across life stages/animal classes to give a single value per livestock category.

5.2.3 Farming practices for net zero scenario analysis

There is a growing body of evidence assessing mitigation options for livestock CH_4 emissions and this chapter explored two key strategies to reduce enteric fermentation and manure storage emissions as these contribute the most CH_4 in the UK GHG Inventory agriculture sector emissions. The practice chosen for reducing enteric fermentation was the use of a feed additive in dairy and beef cattle. For reducing CH_4 emissions during manure storage, the analysis was for slurry acidification of stored pig and dairy cattle slurries.

The feed additive 3-Nitrooxypropanol (3-NOP) recently received authorisation from the Food Standards Agency to be used commercially in ruminant livestock systems in Great Britain (Food Standards Agency, 2023). The inhibitor has a wide body of peer-reviewed literature estimating CH_4 reductions of approximately 20-30% in beef and dairy cattle (Hristov et al., 2013; Hristov et al., 2015; Vyas et al., 2016; Dijkstra et al., 2018; Eory et al., 2020; Alemu et al., 2021; Reisinger et al., 2021) and it has minimal impacts on productivity (Alemu et al., 2021). Much of the literature on 3-NOP has been experimental and dairy cattle have more commonly been tested than any other ruminant, as the 3-NOP is more easily combined into the Total Mixed Ration (TMR) of indoor dairy herds (Yu et al., 2021; Beauchemin et al., 2022; Costigan et al., 2024). However, there is research underway to establish the efficacy of practices to mitigate CH_4 emissions from ruminants in grassland-based systems, such as changes in grass digestibility (Vargas et al., 2022).

A study on commercial-scale Canadian beef cattle production by Alemu et al. (2021) found a 22% (20-26% range) reduction in enteric fermentation CH_4 production per head, and no significant animal health effects were noted that altered productivity. This is further supported by LCA research of Australian and Canadian beef and dairy farming systems, where 3-NOP inclusion in the diet reduced whole farm GHG emissions more than another feed additive (nitrate) (Alvarez-Hess et al., 2019). A randomised block experiment from Hristov et al. (2015) looking into Holstein dairy cattle found consistent declines in CH_4 yield regardless of the concentration used (low = 40 mg/kg, medium = 60 mg/kg or high = 80 mg/kg), which averaged at a 30% reduction. Other research has found slightly higher reductions in CH_4 yield (39 ± 6%) for dairy cattle, and for beef cattle a reduction of 17% (± 4%; Dijkstra et al., 2018). Eory et al. (2020) conducted a marginal abatement cost curve analysis for agriculture in Scotland, looking at practices that reduce GHG emissions and are most cost-effective. In this paper, they use a CH_4 yield reduction of 20% for beef cattle and 30% for dairy cattle, and explain a typical dose of 2-3 g 3-NOP per animal per day is most effective (see also Haisan et al., 2014; Martinez-Fernandez et al., 2018). As beef and dairy production in Scotland is unlikely to be too dissimilar to that in England and Wales, these values were used in this chapter, and it is assumed that these proportions are relevant to all dairy and beef cattle as there are no publicly available datasets on the average number of indoor versus outdoor systems to be able to disaggregate further.

On reducing CH_4 from slurry storage, acidification was chosen as a mitigation strategy as it has been widely studied and reviewed (Petersen et al., 2012; Fangueiro et al., 2015; Misselbrook et al., 2016; Overmeyer et al., 2023) and is also an effective policy mechanism in Denmark, where up to 20% of all slurry goes through acidification prior to spreading. The lowering of slurry pH to an optimal 5.5 is promoted in Denmark for ammonia reduction, which reduces odour and also lowers indirect nitrous oxide GHG emissions. However, research has found that CH_4 emissions can be reduced as well due to the lowering of methanogenic bacteria activity at a more acidic pH (Pedersen et al., 2022). Furthermore, acidification can be incorporated into current slurry management systems such as tanks and lagoons, whereas other mitigation options such as slurry cooling, separation and anaerobic digestion require greater infrastructure changes (Dalby et al., 2022; Overmeyer et al., 2023). In terms of CH_4 reduction, slurry acidification in cattle slurry consistently finds over half of CH_4 emissions reduced, but most the most commonly reported reductions are between 61 to 87% (Petersen et al., 2012; Misselbrook et al., 2016). Misslebrook et al. (2016) also examined the effects of acidification on pig slurry, but found no significant differences between the untreated and acidified slurries. However, other research has found > 90% reduction in CH_4 from treating pig slurry with sulphuric acid (Petersen et al., 2014; Shin et al., 2019; Vechi et al., 2022). For this chapter, the more conservative CH_4 reduction percentage of 91% reported in Vechi et al. (2022) was chosen for the pig slurry acidification scenario as it comes from more robust and regionally relevant research than other figures found during the literature search.

Table 5.3 describes the CH_4 reduction for each of the practices chosen and the literature source for the value. Emission reductions for all practices were reported in terms of percentage reduction in CH_4 yield (i.e., kg CH_4 head^{-1} yr^{-1}) in the literature, which was used to reduce the CH_4 yields in Table 5.2. The abated CH_4 yields were then multiplied by the population of beef or dairy cattle or pigs (depending on the practice). As this was a scenario analysis, a proportion of the livestock populations taking up the practice in each GOR of England and Wales was modelled. However, as there is no publicly available data on the proportion of livestock housed or fed indoors versus outdoors, an uptake of 10% of individuals was used for the analysis, which is conservative but scalable.

Table 5.3 Strategies to reduce methane emissions from enteric fermentation (dairy and beef) and manure management (pigs).

Strategy	Practice	Methane reduction	Reference
Enteric methane	Use of 3-NOP: Dairy	30% reduction in CH_4 yield	Eory et al., 2020
	Use of 3-NOP: Beef	20% reduction in CH_4 yield	
Manure management	Acidification of pig slurry	91% reduction in CH_4 yield	(Vechi et al., 2022)
	Acidification of dairy cattle slurry	77% average reduction in CH_4 yield	Petersen et al., 2012

5.3 Results

On average, 74% of Livestock emissions in Table 5.1 across GORs in England and Wales were from enteric fermentation (63%) and manure management (11%) with the remainder being emissions from grassland excreta deposition or spreading of livestock manures.

The following sections demonstrate the regional Livestock emissions abatement potential of introducing 3-NOP into beef and dairy diets and acidifying pig and dairy cattle slurry, and how much regional uptake would be required year-on-year to move towards net zero emissions by 2050. Overall results are found in Table 5.4 and Figure 5.1 and Figure 5.2 illustrate the potential reduction in Livestock emissions (as a whole) and the enteric fermentation and manure management portion of Livestock emissions, respectively, through a 10% uptake in 1) 3-NOP in beef and dairy cattle diets, 2) acidification of dairy and pig slurries, and 3) a combination of the two practices.

5.3.1 Reducing enteric methane in dairy and beef cattle

As 3-NOP has only recently been approved for commercial use, an assumed current uptake of 0% was given for all regions. If 10% of all dairy and beef animals in England and Wales were fed 3-NOP as part of their diet, then Livestock emissions (21,913 kt CO_2e in Table 5.1 from enteric fermentation, manure management and the deposition or spreading of excreta) could be reduced by 1.3% in a single year due to the reduced CH_4 emissions per head per year. The South West of England has the greatest abatement potential with a 1.6% reduction in Livestock emissions with the 10% uptake of 3-NOP in dairy and beef cattle. Other Western regions of England and Wales had similarly high reduction potentials. The North West and Wales would have a 1.5% decrease, and the West Midlands would have a 1.4% decrease in Livestock emissions. The East of England would only have a 0.5% reduction in Livestock emissions if 10% dairy cattle had 3-NOP in the diet (Figure 5.1).

The reductions are slightly higher in each region when just accounting for the enteric fermentation and manure management aspects of the Livestock emissions. The 10% uptake of 3-NOP in dairy and beef diets would reduce emissions by 2.2%, 2.1%, 1.9%, 1.8% and 1.8% in the South West, North West, West Midlands, East Midlands and South East, respectively (Figure 5.2), due to the reduced enteric methane.

With a total 1.3% reduction in Livestock emissions from the inclusion of 3-NOP in 10% of dairy and beef cattle diets per year across England and Wales (Table 5.5), in theory, a linear increase of 10% new uptake in those populations each year until 100% of the beef and dairy cattle were being fed 3-NOP would result in a 13% lower Livestock emission total than the baseline period (2018 – 2020) by 2034.

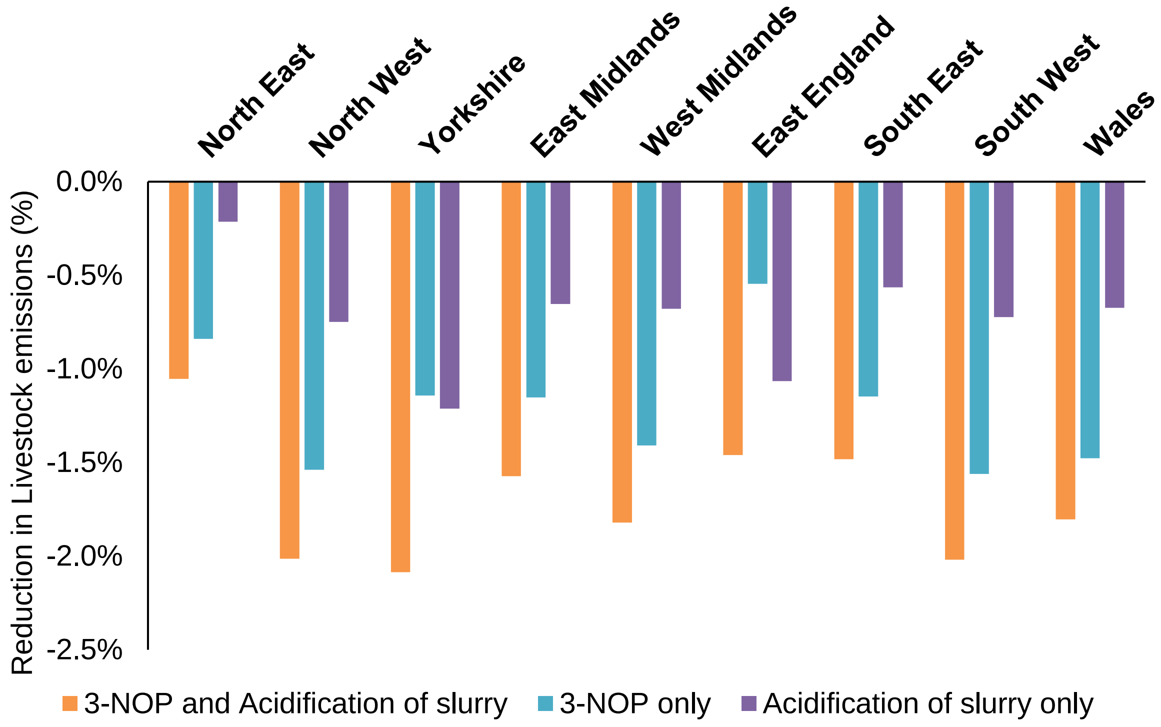

Figure 5.1 Percentage reduction of Livestock emissions (enteric fermentation, manure management and excreta deposition or spreading) for each region from a 10% uptake of 1) 3-NOP in beef and dairy diets and acidification of dairy and pig slurry (orange), 2) 3-NOP only (blue) and 3) Acidification only (purple).

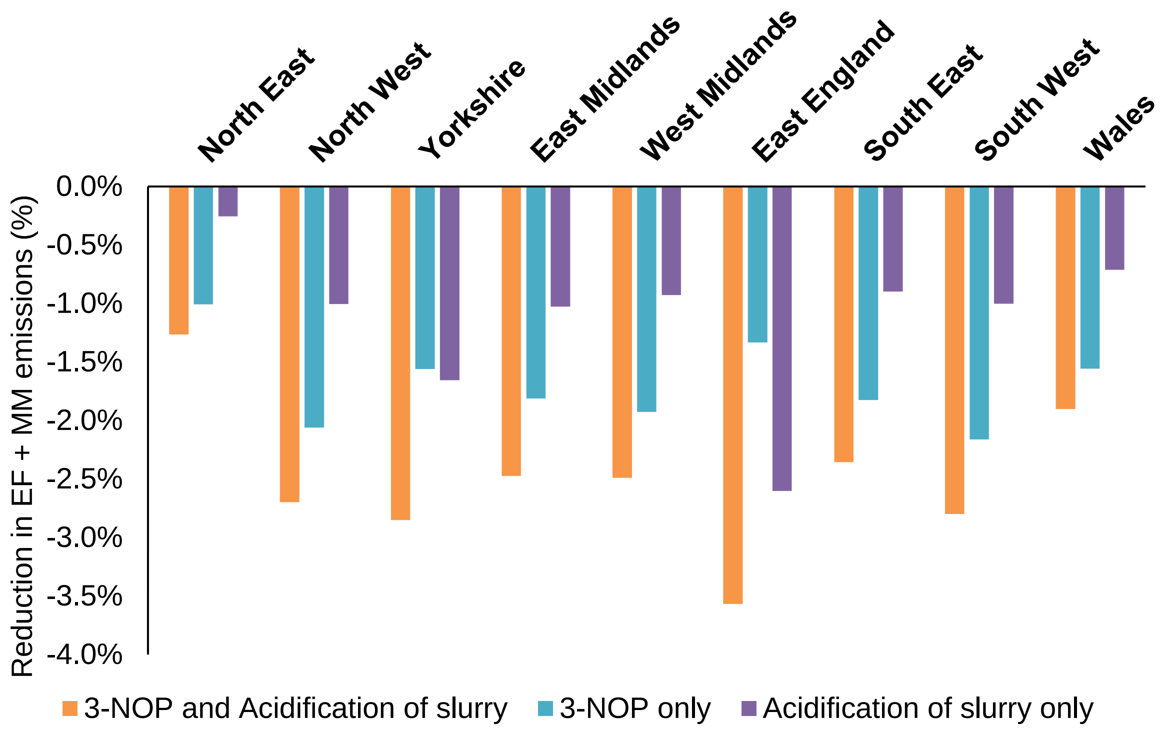

Figure 5.2 Percentage reduction of enteric fermentation (EF) and manure management (MM) portion of Livestock emissions for each region from a 10% uptake of 1) 3-NOP in beef and dairy diets and acidification of dairy and pig slurry (orange), 2) 3-NOP only (blue) and 3) Acidification only (purple).

5.3.2 Reducing methane through acidification of pig and dairy slurry

As there was limited publicly available data on the current uptake of slurry acidification practices in England and Wales, it was assumed current uptake was 0% and the more conservative 10% uptake scenario was used to model emission reductions.

With 10% of dairy cattle and pigs across England and Wales having their slurry acidified, a 0.7% reduction in Livestock emissions would occur in a single year (Table 5.5). The majority of methane reduction from acidification would occur in Yorkshire (1.2%) and the East of England (1.1%). A 0.7% reduction in Livestock emissions would occur in the North West, East and West Midlands, South West and Wales. The lowest reduction would be in the North East at 0.2% (Figure 5.1).

As in the previous section, shifting focus away from Livestock emissions and focussing on the non-excreta deposition and spreading part of the picture shows some increases in reduction potential. The East of England shows the greatest enteric fermentation and manure management emissions reduction of 2.6%, which is followed by Yorkshire at a 1.7% reduction (Figure 5.2).

These results indicate that the 0.7% reduction in Livestock emissions from a 10% uptake of dairy and pig slurry acidification practices across England and Wales is not enough to progress towards net zero GHG emissions by 2050. If the 10% uptake was consistently applied across the next 10 years (i.e., another 10% of dairy and pig slurry each year is acidified), then the total emission reduction by 2034 would be 7% lower than the 2018 – 2020 baseline. Therefore, a third scenario, where the two practices were combined was explored below.

5.3.3 Combining 3-NOP and slurry acidification practices

Clearly both practices in isolation produce modest reductions at 10% uptake of the livestock population and as these practices do not necessarily interact, it is possible to combine them and reap further reductions.

Assuming the 0% current uptake of practices currently, if 10% of the beef and dairy cattle in each region had 3-NOP in their diet, and 10% of dairy cattle and pigs had their slurries acidified in each region, then Livestock emissions across England and Wales could be reduced by 1.8% in a single year (Figure 5.1). This is not a simple addition of the 1.3% from 3-NOP and 0.7% from slurry acidification implementation, as the two practices were modelled separately and produced a new Livestock emissions total each time. The regional new Livestock emission totals for 3-NOP practices only and slurry acidification practices only were summed, and the percentage reduction was calculated to be 1.8%. Looking specifically at the enteric fermentation and manure management portion of Livestock emissions across England and Wales, the combination of practices would deliver a 2.5% reduction (Figure 5.2). See Table 5.5 for a summary of the Livestock emission reductions and reductions in the enteric fermentation and manure management portion of Livestock emissions with the 10% uptake model.

If all beef and dairy animals had 3-NOP in their diet, and all pig and dairy slurry was acidified across England and Wales, then the combined mitigation opportunity would be an 18% reduction in the baseline Livestock emissions value by 2034.

Table 5.4 Changes in Livestock emissions at regional level with the uptake of 3-NOP and slurry acidification practices. Results are broken down into Enteric fermentation ('Enteric') and Manure management ('MM') emissions and then combined to give a Grand Total. Emissions associated with deposited or spread livestock excreta ('Excreta deposit/spread') are reported to show the major representation of enteric and manure methane in Livestock emissions.

Region	Livestock Emissions (t CO_2e)	Livestock Emissions - Enteric/MM (t CO_2e)	Livestock Emissions - Excreta deposit/spread (t CO_2e)	Livestock category	Original values		New values	
					Enteric	MM	Enteric	MM
North East	883,216	735,917	147,300	Dairy*†	39,402	7,808	38,220	7,207
				Beef*	311,739	38,996	305,504	38,996
				Pigs†	5,242	14,084	5,242	12,802
				Sheep	307,609	8,183	307,609	8,183
				Poultry	0	2,853	0	2,853
				TOTAL	**663,992**	**71,924**	**656,575**	**70,041**
				GRAND TOTAL	**735,917**		**726,617**	
North West	3,057,070	2,273,590	783,480	Dairy*†	888,108	175,987	861,465	162,436
				Beef*	600,885	75,166	588,868	75,166
				Pigs†	4,119	11,067	4,119	10,060
				Sheep	490,386	13,045	490,386	13,045
				Poultry	0	14,826	0	14,826
				TOTAL	**1,983,498**	**290,092**	**1,944,837**	**275,534**
				GRAND TOTAL	**2,273,590**		**2,220,371**	
Yorkshire	2,098,077	1,529,394	568,683	Dairy*†	243,131	48,179	235,837	44,469
				Beef*	551,459	68,984	540,430	68,984

Region	Livestock Emissions (t CO_2e)	Livestock Emissions - Enteric/MM (t CO_2e)	Livestock Emissions - Excreta deposit/spread (t CO_2e)	Livestock category	Original values		New values	
					Enteric	MM	Enteric	MM
				Pigs†	65,832	176,868	65,832	160,773
				Sheep	341,539	9,085	341,539	9,085
				Poultry	0	24,317	0	24,317
				TOTAL	**1,201,961**	**327,433**	**1,183,638**	**307,628**
				GRAND TOTAL	**1,529,394**		**1,491,266**	
East Midlands	1,701,650	1,078,293	623,357	Dairy*†	214,511	42,508	208,076	39,234
				Beef*	460,728	57,634	451,513	57,634
				Pigs†	15,823	42,511	15,823	38,643
				Sheep	202,528	5,388	202,528	5,388
				Poultry	0	36,663	0	36,663
				TOTAL	**893,590**	**184,703**	**877,940**	**177,561**
				GRAND TOTAL	**1,078,293**		**1,055,501**	
West Midlands	2,250,100	1,639,851	610,249	Dairy*†	477,026	94,527	462,715	87,249
				Beef*	568,092	71,064	556,730	71,064
				Pigs†	8,077	21,700	8,077	19,726
				Sheep	353,881	9,414	353,881	9,414
				Poultry	0	36,069	0	36,069
				TOTAL	**1,407,076**	**232,775**	**1,381,403**	**223,522**
				GRAND TOTAL	**1,639,851**		**1,604,925**	
East England	1,441,050	587,619	853,431	Dairy*†	45,053	8,928	43,701	8,240
				Beef*	217,318	27,185	212,972	27,185

Region	Livestock Emissions (t CO_2e)	Livestock Emissions - Enteric/MM (t CO_2e)	Livestock Emissions - Excreta deposit/spread (t CO_2e)	Livestock category	Original values		New values	
					Enteric	MM	Enteric	MM
				Pigs†	51,121	137,346	51,121	124,848
				Sheep	57,233	1,522	57,233	1,522
				Poultry	0	41,914	0	41,914
				TOTAL	**370,725**	**216,895**	**365,027**	**203,709**
				GRAND TOTAL	**587,619**		**568,736**	
South East	1,387,900	869,976	517,924	Dairy*†	172,987	34,279	167,798	31,640
				Beef*	377,018	47,162	369,477	47,162
				Pigs†	8,181	21,979	8,181	19,979
				Sheep	192,311	5,116	192,311	5,116
				Poultry	0	10,943	0	10,943
				TOTAL	**750,496**	**119,480**	**737,766**	**114,840**
				GRAND TOTAL	**869,976**		**852,606**	
South West	5,107,800	3,673,339	1,434,461	Dairy*†	1,304,047	258,410	1,264,926	238,512
				Beef*	1,353,382	169,298	1,326,314	169,298
				Pigs†	14,505	38,971	14,505	35,424
				Sheep	497,211	13,227	497,211	13,227
				Poultry	0	24,289	0	24,289
				TOTAL	**3,169,145**	**504,194**	**3,102,956**	**480,750**
				GRAND TOTAL	**3,673,339**		**3,583,706**	
Wales	3,986,080	3,763,650	222,430	Dairy*†	835,641	165,591	810,572	152,840
				Beef*	996,887	124,703	976,949	124,703

Region	Livestock Emissions (t CO_2e)	Livestock Emissions - Enteric/MM (t CO_2e)	Livestock Emissions - Excreta deposit/spread (t CO_2e)	Livestock category	Original values		New values	
					Enteric	MM	Enteric	MM
				Pigs†	1,109	2,979	1,109	2,708
				Sheep	1,580,614	42,047	1,580,614	42,047
				Poultry	0	14,079	0	14,079
				TOTAL	**3,414,251**	**349,399**	**3,369,244**	**336,378**
				GRAND TOTAL	**3,763,650**		**3,705,622**	
England and Wales	**21,912,943**	**16,151,629**	**5,761,314**					

* 3-NOP practices

† Slurry acidification practices

Table 5.5 Outcome of a 10% uptake in 3-NOP use in dairy and beef cattle, dairy and pig slurry acidification or the combination of practices across England and Wales. Data presented are total emissions (t CO_2e), percentage reduction in Livestock emissions (Table 5.4) and percentage reduction in the Enteric fermentation and Manure Management (MM) part of Livestock emissions.

Practice	Emissions (t CO_2e)	Livestock emissions reduction	Livestock emissions reduction – Enteric/MM only
Combination of 3-NOP and slurry acidification	21,513,988	-1.8%	-2.5%
3-NOP only	21,620,919	-1.3%	-1.8%
Slurry acidification only	21,749,335	-0.7%	-1.0%

5.4 Discussion and conclusions

One of the key GHGs that can be reduced through changes in farm practice and can lead to measurable differences in atmospheric climate in the next decade is methane. CH_4 is the predominant GHG in livestock production, which means that any practice that is reducing CH_4 also needs to work around the farming system. This chapter has demonstrated the efficacy of two individual practices and the combination of the two practices on methane emission reduction in England and Wales and has shown that additional practices are needed to reduce CH_4 from livestock production.

5.4.1 Effectiveness of 3-NOP in dairy and beef cattle diets

The results of the scenario analysis demonstrated a 1.3% reduction in Livestock emissions from 10% of dairy and beef cattle being fed 3-NOP in their diets across England and Wales, and so if uptake was 100% then a 13% reduction on the baseline would occur. Whilst it is useful to present what 100% uptake of the practice looks like in terms of reducing CH_4, there are several factors which will have a considerable impact on its feasibility.

Firstly, it is challenging to identify a reasonable level of uptake as 3-NOP has only recently been approved (Food Standards Agency, 2023), which is why 10% was used in this chapter, and it is unknown how many dairy and beef animals this practice is technically relevant to (i.e., indoor or partially indoors systems). A farmer survey conducted by March et al. (2014) on British farmers observed more intensive systems to only represent 8% of the 863 respondents, where lactating dairy cattle were housed for 24 hours per day. On the other hand, a more typical summer grazing system was utilised by 31% of farms without the need for housed feeding and 38% of farmers did have some indoor feeding during summer. However, this research is ten years old and with summer weather becoming warmer and rainfall patterns less predictable with a changing climate in England and Wales (Met Office, 2019), more indoor feeding of concentrate may be required, which improves the opportunity to implement 3-NOP diets.

Secondly, the results reported in this study are reductions in Livestock emissions, and the manure management and enteric fermentation-only portion of Livestock emissions, in comparison to the 2018 to 2020 baseline figures and were calculated using the latest livestock population data. However, reductions in livestock populations over the next 5 to 25 years are likely due to Climate Change Committee (CCC) recommendations to UK Government for a 20% decrease in meat and dairy consumption in the diet by 2030, and a 35% reduction by 2050 (CCC, 2020b). This would naturally lower Livestock emissions anyway, so in combination with the practices assessed in this chapter, could result in more than a 1.3% to 13% reduction with a 10% to 100% uptake of 3-NOP in beef and dairy diets, respectively.

There are also potential challenges of farmers adopting this practice across England and Wales. One is the current lack of incentivisation to farmers to purchase 3-NOP for cattle feed and the current cost data for the UK markets leading 3-NOP additive, Bovaer, is limited (Vera Eory et al., 2020). In a recent call for evidence from Defra on methane inhibitors, 69% of farmers surveyed (n = 139) said that cost would be a key barrier to uptake on farm and indicated that if financial support for inhibitors was made available (e.g., through the Sustainable Farming Incentive) then this may increase uptake (Defra, 2023c). Another more technical challenge of 3-NOP is its efficacy in grazing systems as the inhibitor needs to be consumed regularly (i.e., with every mouthful of feed) to show a notable reduction in CH_4 yield (Costigan et al., 2024). Research indicates that 3-NOP could be used in a bolus to release the additive slowly whilst out at pasture (Vera Eory et al., 2020; Costigan et al., 2024; Muñoz et al., 2024).

Furthermore, there has been recent backlash from the public on the use of cattle feed additives in dairy products. In November 2024, the farmer-owned dairy cooperative business Arla announced it is trialling a 3-NOP product called Bovaer© on several of its associated dairy farms, which resulted in widespread misinformation sharing about the safety of the feed additive to humans in the media. This lead Arla, the Food Standards Agency, and the manufacturers of Bovaer© to release statements assuring the public of the rigorous testing of the additive and its safety in cattle feed and subsequent dairy products (Arla, 2024; dsm-firmenich, 2024; Food Standards Agency, 2024).

Nevertheless, this is the first regional and national-level scenario analysis of 3-NOP uptake using UK GHG Inventory and livestock data for England and Wales and adds further evidence to the unanswered question of how agriculture as a sector can move towards net zero emissions. The findings in this chapter suggest that 3-NOP is an effective practice, which is supported by Eory et al. (2020), who performed a Marginal Abatement Cost Curve for Scotland to assess the efficacy of the methane inhibitor 3-NOP on Scottish agriculture. The authors found incorporation of 3-NOP into the beef and dairy sector to be the most effective GHG mitigation measure, but also presented as costly due to the need to continuously supply 3-NOP in the feed (Eory et al., 2020).

5.4.2 Effectiveness of dairy and pig slurry acidification

As before, this is the first known regional scenario analysis of slurry acidification uptake for methane reduction for England and Wales, so the findings of this chapter present an initial indication of how successful the practice would be. Compared to the use of 3-NOP, the CH_4 mitigation potential of slurry acidification is more limited according to the findings of this chapter, which is likely due to the lower CH_4 yield per head of livestock per year for manure management than enteric fermentation (Table 5.2). In Yorkshire and the East of England, where pig numbers are higher, and the North West and Wales, where dairy numbers are high, greater reductions in CH_4 were found (Figure 5.1; Appendix C).

As mentioned above, slurry acidification is a widely practiced measure in Denmark for pig slurries (Pedersen et al., 2022) and is widely supported as a co-beneficial option for reducing CH_4 emissions, as well as ammonia (NH_3) (Misselbrook et al., 2016; Overmeyer et al., 2023). The key challenge in England and Wales for even a modest 10% uptake of slurry acidification is that there would need to be chemical handling training for farmers and potentially adjustments in some pipework in slurry tank and lagoon systems to handle the acid appropriately (Fangueiro et al., 2015; Regueiro et al., 2016).

Additionally, some research has indicated that acidification of slurry in storage (i.e., in tanks or lagoons) is not appropriate due to the rise in pH after acid treatment has worn off, and instead separation of slurry prior to treatment, or acidifying slurry prior to spreading are more efficient strategies. However, these only reduce NH_3 emissions, rather than the CH_4 produced during storage (Overmeyer et al., 2021).

Furthermore, the acidification practice used in this chapter was agnostic to the type of acid applied, whereas much of the experimental research behind this practice has found differing effects of different acids on NH_3 and GHG levels (Fangueiro et al., 2015). Both the Petersen et al. (2012) and Vechi et al. (2022) paper used sulfuric acid to treat the dairy cattle and pig slurry, respectively, which aids in the comparison of results. However, other research has found alternatives such as aluminium sulfate (Regueiro et al., 2016) and nitric acid (Dalby et al., 2022) to be similarly or more effective than sulfuric acid in treating dairy and pig slurries.

5.4.3 Conclusions

This chapter aimed to assess the potential for reducing methane emissions from livestock in England and Wales to align with net zero ambitions. The findings demonstrate that these practices alone are not enough to reduce Livestock emissions within the range of net zero. If 10% of dairy and beef cattle across England and Wales were fed 3-NOP in their diet, this would reduce baseline Livestock emissions (2018 to 2020) by 1.3% with greatest reductions in western regions of England and Wales. In the scenario where slurry acidification is taken up for 10% of pigs and dairy cattle, this would only result in a 0.7% reduction in the baseline emissions with Yorkshire and the East of

England showing the greatest declines. Combining the two practices will achieve greater reductions (1.8%), and scaling this up to 100% uptake (i.e., all dairy and beef animals had 3-NOP in their diet and all dairy and pig slurry was acidified), the analysis suggests that enteric CH_4 emissions could be reduced by 25% compared to the baseline. Technical challenges regarding uptake potential and the calculations performed were outlined, but the results in this chapter present an initial idea of the potential CH_4 reduction strategies available in England and Wales.

Chapter 6 – General discussion and conclusions

Global temperatures are now over 1°C above pre-industrial levels, and even with current national level policies and future targets, commitments and pledges, temperature rise may still exceed the 1.5°C Paris Agreement limit before the end of the century (Climate Action Tracker, 2023; Boehm et al., 2023). With national legally binding net zero GHG emission targets just 25 years away, and the pressure of agriculture sector reduction targets from the NFU just 15 years away, it is crucial that the agriculture sector in England and Wales escalates actions to reduce its 48 Mt CO_2e per year (Defra, 2023a). Whilst the proportion of emissions from the agriculture sector sits at 11% currently, other economic sectors are decarbonising by phasing out fossil fuel usage and introducing more renewable energy sources (Brown et al., 2023), which will subsequently increase the relative agricultural contribution. Key questions remain of how feasible net zero GHG emission is for the sector, what timeframe net zero might occur in and what practices are most suitable for this transition.

In this thesis, I aimed to further the understanding of whether the agriculture sector could meet net zero GHG emissions by modelling agricultural emissions and mitigation practices at multiple scales, which was done across three data chapters. Chapter 3 focussed on the quantification of GHG emissions using LCA of the main commodities produced on a mixed arable-pig case study farm (the UoL farm), and farm-management relevant mitigation scenarios were presented for the arable and pig enterprises. Chapter 4 then focussed on the mitigation of emissions from cropland soils using spatial analysis to identify land area available for practices relating to the improvement of nitrogen use and introducing legumes into arable rotations. This chapter also investigated an opportunity to afforest 1% of land area in England and how the carbon accumulation in the tree biomass and soils would change up to 2050, discussing implications for balancing GHG emissions with carbon removals. Chapter 5 then focussed on the mitigation of methane-based livestock emissions with a focus on the CH_4 inhibitor 3-NOP for reducing enteric fermentation in beef and dairy cattle, and slurry acidification of dairy and pig manures.

6.1 Quantifying agricultural emissions

The University of Leeds farm is a commercial-scale mixed pig and arable research farm and was used as the case study for the LCA of agricultural products in Chapter 3. In this chapter, I develop a baseline GHG emission inventory for the farm and, using the LCA framework, calculate the GHG emissions (GWP_{100}) of the main combinable crops grown on-farm and pig production to further the scientific evidence around the global warming impacts of agricultural production.

The data presented provides new evidence on the lifecycle emissions of some of the most widely produced and consumed farm products globally, namely wheat, barley, and pigs (Ottosen et al., 2021; IPCC, 2022), as well as oilseed rape and grass silage. For the arable crops, high quality and high granularity primary farm data was used to calculate GHG emissions, providing a more robust LCA. However, for the pig unit data was of mixed quality, including incomplete primary data and secondary data that was geographically relevant, but not necessarily accurate. The LCA analysis picked up on nitrogen fertiliser being a key emission hotspot in arable cropping, and feed and manure management for pig production. These are common findings for these farming systems, so whilst the analysis in Chapter 3 did not establish any novel relationships between the farming systems and GHG emissions, it furthers the evidence base for agricultural LCA and demonstrates key challenges agricultural LCAs face in terms of data quality.

Other product-level LCAs have found similar findings to this study in terms of key sources of GHG emissions. For example, pig feed is consistently the greatest source of GHG emissions, particularly when soya is an ingredient as this is often associated with indirect Land Use Change (LUC) emissions from the country of origin (Stephen, 2012; McAuliffe et al., 2016). Manure management of pig slurries and manure is also a hotspot for GHG emissions and eutrophication in LCA studies (Stephen, 2012; McAuliffe et al., 2016; Ottosen et al., 2021). Williams et al. (2006) is probably one of the most cited LCAs of arable and horticultural products using data from England and Wales. The authors' LCA of feed wheat and barley and oilseed rape found similar results to the study presented in Chapter 3, where fertiliser manufacture and emissions from the application of nitrogen to soils dominated these footprints. However, the actual GHG emission intensity reported in Chapter 3 are lower than figures in Williams et al. (2006), which is partly due to the use of different GWP_{100} metric values and higher average yields on the UoL farm.

A key challenge of the findings in Chapter 3 is around the lack of available data on the pig feed and high quality records of pig weight and populations over the time period of interest. Whilst the data used as a proxy in this analysis highlighted pig feed as a key contributor to the GHG footprint and this aligns with findings in other papers, it does also bring to light an important implication for better data accessibility and quality in any further research with the UoL farm pig unit. This chapter used LCA to quantify the GHG emissions from the farm, and many of the widely available farm carbon calculator tools that farmers are using to carbon audit their farm (e.g., Cool Farm Tool, Agrecalc, Farm Carbon Calculator) are based on similar methodologies and emission factors as used in Chapter 3. However, there is a system-wide discord between tool methodologies, which relates partly due to the data entry required of farmers (e.g., average fertiliser used *versus* specific applications, average livestock numbers over 12 months *versus* accurate accounts of purchases, sales and deaths) and partly due to subtle differences in the choice of methodology between tools, as highlighted in a report by ADAS (2023). Relating this back to the core aim of this chapter to understand whether net zero is feasible for the agriculture sector, these implications suggest that even if GHG emissions could be reduced on each farm, the underlying data and methodology behind this may be inaccurate. The implication of this is that farmers need to be consistently using a single method for carbon auditing to ensure that year-on-year progress in GHG emission reductions is due to actual farm practice change, and not just methodological changes.

6.2 Mitigating agricultural emissions

In this thesis, I have modelled the GHG reductions associated with several on-farm practices to understand how feasible net zero agriculture is. This was done at farm-level in Chapter 3, and at a regional- and national-level in Chapter 4 and Chapter 5.

Using the UoL farm LCA findings in Chapter 3, I modelled the impact of nitrification inhibitor uptake and slurry acidification on the annual average emissions footprint (2015 – 2018) on-farm. I found that in combination, these practices could deliver a 13% reduction in farm emissions in a year, which is a substantial decrease but not enough to be considered net zero (i.e., any GHG emissions remaining are residual and cannot be lowered any further). These practices were appropriate for the farm as they did not interact with each other and aligned with current farm management and the technical potential of the farm (e.g., inhibitors added to the fertiliser as nitrogen application rates were not in excess and acid was added to current slurry management systems to avoid changes in infrastructure). However, the findings of this chapter demonstrate important implications for other farms and the possibility of net zero. For example, farms are not uniform in their size, geography, or current management (to name a few), which means that the introduction of GHG reduction practices will need to be appropriate for each farm without compromising farm productivity (Reay, 2020; Rosa and Gabrielli, 2023). Therefore, it is unlikely that each individual farm can reduce GHG emissions enough to be considered net zero, but if every farm could reduce GHG emissions to the technical limit, as a sector net zero farming may be possible.

The purpose of Chapter 4 was to estimate the amount of land area available for emission reduction, as well as the potential GHG reduction or carbon removal opportunity available from arable practices or woodland at a regional scale, respectively. A regional to national scaled up approach was taken in this chapter to better understand the land area available for GHG reduction practices and use data on current levels of uptake of practices to help inform suitable scenario modelling, which steers away from the idea of a ‘one size fits all’ approach to net zero transitions in agriculture.

The scenario analysis of a one-off 10% increase in the uptake of practices (as measured by land area) to improve the use of nitrogen fertiliser and use of legumes in arable rotations from current levels showed that on the 7.9 Mha land available in England and Wales, annual average Soil emissions (i.e., mostly nitrous oxide emissions from fertiliser applications) could be reduced by 23% if the practices were used in combination. The technical potential of achieving a 10% increase in practice uptake in each region is debatable, and certainly 100% of farmers adopting better nitrogen management practices and using legume cover crops in a rotation is likely impossible. The analysis does not consider the agronomic benefits of nitrogen fertilisers for maximising yields, so reducing their use (as modelled in Chapter 4) has a theoretical limit before crop yields started to decline and food imports were needed to meet demand (Eory et al., 2018). Based off of the findings in this thesis, it is likely that farms will need to use a combination of practices to reduce GHG emissions and a greater uptake than 10% of land area is required to achieve GHG reductions that are aligned with net zero. A key challenge of modelling the nitrogen optimisation and legume practices together in this chapter is that the GHG reduction rates came from different research articles, and thus may not be additive or useful to combine. Therefore, further scenario analyses modelling the reduction in arable-related emissions would need to ensure the practices selected are additive, and that the uptake being modelled does not surpass the theoretical limit to avoid unrealistic expectations of farmers.

Afforestation is a useful strategy for reducing the concentration of CO_2 in the atmosphere, and thus limiting further acceleration of warming. I modelled a one-off 1% afforestation of suitable land area (i.e., not on highly productive ALC grade 1-3 agricultural land or protected landscapes) in Chapter 4. This was shown to sequester an average of 200 kt CO_2e across England in the vegetation and soil in the first five years since planting, which is equal to 3% of annual average emissions between 2018 to 2020. The South East had the greatest amount of land area afforested for the 1% uptake. It would take 15 years (2040-2045) of woodland growth to be sequestering the same amount of CO_2e as the 2018 to 2020 Soil emissions average. The one-off woodland planting of 1% land area (53,865 ha) in England is slightly higher than the target rate proposed by the CCC to UK Government, which was 50,000 ha per year by 2050 (CCC, 2020a; CCC, 2020b), which could indicate that the scenario used in this thesis is roughly on par with Government recommendations, but I did not analyse the 1% as a rate. However, even if the area of afforested land did increase across England to enhance carbon sequestration each year, there is still a question of whether CO_2 removals should be counted against non-CO_2 GHG as offsets. For example, an analysis published by the Parliamentary Commissioner for the Environment of New Zealand showed that it is possible to use afforestation to offset CH_4 emissions from livestock, but as one tonne of CH_4 does not equal the same temperature effect as one tonne of CO_2, unrealistic areas of afforestation would be required to achieve a 100% CH_4 offset and relies on at least a 24-47% reduction in gross CH_4 emissions nationally (Parliamentary Commissioner for the Environment, 2022). Future work could use the spatial analysis produced in this chapter, but utilise calculations performed in the New Zealand work to estimate the area of woodland required by 2050 to achieve an actual net zero effect.

A limitation to some of the conclusions drawn in this chapter was that the scenario analysis only assessed afforestation through the single lens of carbon sequestration for climate change mitigation. Woodland expansion needs to be done carefully and strategically (right tree, right place) to maximise the co-benefit potential, for example considering other ecosystem services like the cultural benefits of woodlands and biodiversity enhancement (Bateman et al., 2023). Further research assessing the spatial implications of afforestation should consider mapping biodiversity, water, urban areas and other available spatial information to ensure woodland is not only being created on agriculturally marginal land, but that it is also providing a range of benefits to the local area.

In Chapter 5, I then used a similar scenario analysis methodology, but instead focussed on reducing livestock production emissions, which are mostly methane. The literature on livestock emission mitigation is extensive (see reviews by Montes et al., 2013; Hristov et al., 2013; Herrero et al., 2016; Arndt et al., 2022), and were used to determine the practices modelled for this chapter. A 10% uptake of the methane inhibiting feed additive 3-NOP for beef and dairy cattle and a 10% uptake of slurry acidification practices for pig and dairy slurries was modelled regionally. Average annual Livestock emissions (2018 to 2020; i.e., enteric fermentation and manure management) would be reduced by only 1.3% with the 3-NOP practices, and 0.7% from slurry acidification. Combined, these practices would reduce baseline Livestock emissions by a total of 1.8%. The reason for the lower mitigation percentages compared to Chapter 4 is because the methane reduction percentage was applied to methane yields (i.e., kg CH_4 $head^{-1}$ $year^{-1}$) for enteric fermentation and manure management in each livestock type (dairy, beef and pigs), rather than a flat reduction percentage. Whilst this research presented interesting GHG reduction data on a newly approved methane inhibitor, the recent media storm surrounding the use of Bovaer (3-NOP) in dairy cattle on several Arla trial farms also raises an important point around public education and future research on this topic (Arla, 2024). In fact, one of the key barriers to meeting net zero goals in the area of methane inhibition could be the public perception and understanding of novel products being used to reduce GHG emissions, which needs to be tackled by educators and policymakers.

Overall, the findings of Chapter 4 and Chapter 5 demonstrate that even a small 10% uptake of practices to reduce GHG emissions can do so at a regional- or national-scale, and even a 1% afforestation uptake can deliver considerable carbon removals. However, as the core aim of this thesis was to better understand how feasible net zero agriculture could be by 2050, the conclusion is that further modelling is required to better understand what combinations of practices are required to reduce GHG emissions and how much uptake should feasibly be to deliver on net zero. Further, whilst afforestation is part of Government strategy (CCC, 2020a), further research – similar to that produced by the New Zealand Government (Parliamentary Commissioner for the Environment, 2022) – is required to fully understand the temperature offset available from carbon removals for offsetting non-CO_2 GHG emissions in England and Wales.

6.3 Final remarks

Agriculture in England and Wales needs to reduce its contribution to climate change by lowering GHG emissions in regionally appropriate and technically feasible ways. This thesis set out to provide new scientific understanding to help answer the critical question of whether agriculture can be net zero and under what timescale. The three data chapters (Chapters 3, 4 and 5) have answered the aims of the thesis through a combination of quantitative LCA, spatial analysis and scenario analysis methodologies using mostly high quality primary farm data and regional-level secondary data. The findings of this research have important implications for how arable and livestock farming practices could be altered to lower GHG emissions, and how afforestation on just 1% of land in England could enhance carbon sequestration and mitigate further climate change, but also how there is a need for better data to be more accurate about net zero feasibility.

This research demonstrates that mitigation actions on-farm can have cascading GHG reduction effects from farm-level up to national-level, even with fairly low uptake levels. However, it will take more than the one or two changes in farm practice modelled in this study and greater levels of uptake across England and Wales to reduce GHG emissions significantly and meet national targets or sectoral ambitions. As observed in Chapter 3, a key limitation to agricultural research is often data access and data quality, which is critical for calculating accurate GHG emission footprints and understanding the technical feasibility of mitigation action. More policy action to support better access to primary farm data in the supply chain will allow both food and drink businesses and policymakers to more confidently incentivise GHG mitigation on-farm and further research will be better able to establish nuances in farm- or regional-level practices required to meet net zero targets.

With the evidence acquired in this research, it is not possible to confidently conclude that net zero is or isn't achievable for agriculture as there are numerous additional analyses required to improve confidence in the results. What is clear, however, is that GHG emissions from agricultural activity are not uniform across the different regions of England or Wales and the processes behind the production of methane and nitrous oxide in particular are too complex to provide a 'silver bullet' solution to reducing them and these GHG emissions will never be zero. Therefore, the final conclusion of this thesis is that net zero may not be possible for each individual farm, especially those that are operating at maximum production efficiency already, those that are at their technical limit to adopting new practices (e.g., reducing the amount of nitrogen applied to crops), or those that may not have marginal land area available for afforestation. However, if every farm could take some action, then potentially as a sector the overall contribution to global warming would become more aligned with net zero.

References

Abalos, D., Jeffery, S., Sanz-Cobena, A., Guardia, G. and Vallejo, A. 2014. Meta-analysis of the effect of urease and nitrification inhibitors on crop productivity and nitrogen use efficiency. *Agriculture, Ecosystems & Environment*. **189**, pp.136–144.

Abbass, K., Qasim, M.Z., Song, H., Murshed, M., Mahmood, H. and Younis, I. 2022. A review of the global climate change impacts, adaptation, and sustainable mitigation measures. *Environmental Science and Pollution Research*. **29**(28), pp.42539–42559.

Abdalla, M., Hastings, A., Cheng, K., Yue, Q., Chadwick, D., Espenberg, M., Truu, J., Rees, R.M. and Smith, P. 2019. A critical review of the impacts of cover crops on nitrogen leaching, net greenhouse gas balance and crop productivity. *Global Change Biology*. **25**(8), pp.2530–2543.

ADAS 2023. *Harmonisation of Carbon Accounting Tools for Agriculture*. Department for Environment, Food & Rural Affairs.

AHDB 2018a. Cereals and Oilseeds markets. [Accessed 16 February 2024]. Available from: https://ahdb.org.uk/cereals-oilseeds-markets.

AHDB 2018b. Hay and straw prices. [Accessed 16 February 2024]. Available from: https://ahdb.org.uk/dairy/hay-and-straw-prices.

AHDB 2021. Nutrient Management Guide (RB209): Section 2 Organic materials. [Accessed 17 October 2022]. Available from: https://projectblue.blob.core.windows.net/media/Default/Imported%20Publication%20Docs/RB209%202022/RB209_Section2_2022_220224_WEB.pdf.pdf.

AHDB 2023a. *Nutrient Management Guide (RB209): Section 4 Arable Crops* [Online]. Coventry. [Accessed 3 July 2024]. Available from: https://projectblue.blob.core.windows.net/media/Default/Imported%20Publication%20Thumbs/RB209/2023/NutManGuideRB209S4_230526_WEB.pdf.

AHDB 2023b. Nutrition guidance for pigs. [Accessed 3 February 2024]. Available from: https://ahdb.org.uk/knowledge-library/nutrition-guidance-for-pigs.

Alemu, A.W., Pekrul, L.K.D., Shreck, A.L., Booker, C.W., McGinn, S.M., Kindermann, M. and Beauchemin, K.A. 2021. 3-Nitrooxypropanol Decreased Enteric Methane Production From Growing Beef Cattle in a Commercial Feedlot: Implications for Sustainable Beef Cattle Production. *Frontiers in Animal Science*. **2**.

Allen, M.R., Friedlingstein, P., Girardin, C.A.J., Jenkins, S., Malhi, Y., Mitchell-Larson, E., Peters, G.P. and Rajamani, L. 2022. Net Zero: Science, Origins, and Implications. *Annual Review of Environment and Resources*. **47**(Volume 47, 2022), pp.849–887.

Allen, M.R., Shine, K.P., Fuglestvedt, J.S., Millar, R.J., Cain, M., Frame, D.J. and Macey, A.H. 2018. A solution to the misrepresentations of CO2-equivalent emissions of short-lived climate pollutants under ambitious mitigation. *npj Climate and Atmospheric Science*. **1**(1), p.16.

Alvarez-Hess, P.S., Little, S.M., Moate, P.J., Jacobs, J.L., Beauchemin, K.A. and Eckard, R.J. 2019. A partial life cycle assessment of the greenhouse gas mitigation potential of feeding 3-nitrooxypropanol and nitrate to cattle. *Agricultural Systems*. **169**, pp.14–23.

Ambrose, H.W., Dalby, F.R., Feilberg, A. and Kofoed, M.V.W. 2023. Additives and methods for the mitigation of methane emission from stored liquid manure. *Biosystems Engineering*. **229**, pp.209–245.

Amelung, W., Bossio, D., de Vries, W., Kögel-Knabner, I., Lehmann, J., Amundson, R., Bol, R., Collins, C., Lal, R., Leifeld, J., Minasny, B., Pan, G., Paustian, K., Rumpel, C., Sanderman, J., van Groenigen, J.W., Mooney, S., van Wesemael, B., Wander, M. and Chabbi, A. 2020. Towards a global-scale soil climate mitigation strategy. *Nature Communications*. **11**(1), p.5427.

Amon, B., Çinar, G., Anderl, M., Dragoni, F., Kleinberger-Pierer, M. and Hörtenhuber, S. 2021. Inventory reporting of livestock emissions: the impact of the IPCC 1996 and 2006 Guidelines. *Environmental Research Letters*. **16**(7), p.075001.

Animal and Plant Health Agency 2024. *Livestock Demographic Data Group: Pig population report – 2023* [Online]. Available from: http://apha.defra.gov.uk/documents/surveillance/diseases/lddg-pig-pop-report-23.pdf.

Antille, D.L., Chamen, W.C., Tullberg, J.N. and Lal, R. 2015. The Potential of Controlled Traffic Farming to Mitigate Greenhouse Gas Emissions and Enhance Carbon Sequestration in Arable Land: A Critical Review. *Transactions of the ASABE*., pp.707–731.

Archer, D., Eby, M., Brovkin, V., Ridgwell, A., Cao, L., Mikolajewicz, U., Caldeira, K., Matsumoto, K., Munhoven, G., Montenegro, A. and Tokos, K. 2009. Atmospheric Lifetime of Fossil Fuel Carbon Dioxide. *Annual Review of Earth and Planetary Sciences*. **37**(1), pp.117–134.

Arias, P.A., Bellouin, N., Jones, R.G., Naik, V., Plattner, G.-K., Rogelj, J., Sillmann, J., Storelvmo, T., Thorne, P.W., Trewin, B., Rao, K.A., Adhikary, B., Allan, R.P., Armour, K., Barimalala, R., Canadell, J.G., Cassou, C., Cherchi, A., Collins, W., Corti, S., Cruz, F., Dentener, F.J., Dereczynski, C., Luca, A.D., Diongue, A., Doblas-Reyes, F.J., Dosio, A., Douville, H., Engelbrecht, F., Fyfe, J.C., Gillett, N.P. and Goldfarb, L. 2021. Technical Summary *In*: *Climate Change 2021: The Physical Science Basis. Contribution of Working Group I to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change*. Cambridge, UK: Cambridge University Press, p.112.

Arla 2024. Arla Bovaer® Statement & FAQs. [Accessed 26 February 2025]. Available from: https://www.arlafoods.co.uk/arla-bovaer-statement/.

Arndt, C., Hristov, A.N., Price, W.J., McClelland, S.C., Pelaez, A.M., Cueva, S.F., Oh, J., Bannink, A., Bayat, A.R., Crompton, L.A., Dijkstra, J., Eugène, M.A., Kebreab, E., Kreuzer, M., McGee, M., Martin, C., Newbold, C.J., Reynolds, C.K., Schwarm, A., Shingfield, K.J., Veneman, J.B., Yáñez-Ruiz, D.R. and Yu, Z. 2021. Strategies to mitigate enteric methane emissions by ruminants - a way to approach the 2.0°C target. *agriRxiv*. **2021**, p.20210085288.

Arndt, C., Hristov, A.N., Price, W.J., McClelland, S.C., Pelaez, A.M., Cueva, S.F., Oh, J., Dijkstra, J., Bannink, A., Bayat, A.R., Crompton, L.A., Eugène, M.A., Enahoro, D., Kebreab, E., Kreuzer, M., McGee, M., Martin, C., Newbold, C.J., Reynolds, C.K., Schwarm, A., Shingfield, K.J., Veneman, J.B., Yáñez-Ruiz, D.R. and Yu, Z. 2022. Full adoption of the most effective strategies to mitigate methane emissions by ruminants can help meet the 1.5 °C target by 2030 but not 2050. *Proceedings of the National Academy of Sciences*. **119**(20), p.e2111294119.

Badagliacca, G., Ruisi, P., Rees, R.M. and Saia, S. 2017. An assessment of factors controlling N2O and CO2 emissions from crop residues using different measurement approaches. *Biology and Fertility of Soils*. **53**(5), pp.547–561.

Balafoutis, A., Beck, B., Fountas, S., Vangeyte, J., Wal, T.V. der, Soto, I., Gómez-Barbero, M., Barnes, A. and Eory, V. 2017. Precision Agriculture Technologies Positively Contributing to GHG Emissions Mitigation, Farm Productivity and Economics. *Sustainability*. **9**(8), p.1339.

Basset-Mens, C., Avadí, A., Bessou, C., Acosta-Alba, I., Biard, Y. and Payen, S. 2022. *Life Cycle Assessment of agri-food systems* [Online]. éditions Quae. [Accessed 16 September 2023]. Available from: https://www.quae-open.com/produit/179/9782759234677/life-cycle-assessment-of-agri-food-systems.

Basset-Mens, C. and van der Werf, H.M.G. 2005. Scenario-based environmental assessment of farming systems: the case of pig production in France. *Agriculture, Ecosystems & Environment*. **105**(1), pp.127–144.

Bateman, I.J., Anderson, K., Argles, A., Belcher, C., Betts, R.A., Binner, A., Brazier, R.E., Cho, F.H.T., Collins, R.M., Day, B.H., Duran-Rojas, C., Eisenbarth, S., Gannon, K., Gatis, N., Groom, B., Hails, R., Harper, A.B., Harwood, A., Hastings, A., Heard, M.S., Hill, T.C., Inman, A., Lee, C.F., Luscombe, D.J., MacKenzie, A.R., Mancini, M.C., Morison, J.I.L., Morris, A., Quine, C.P., Snowdon, P., Tyler, C.R., Vanguelova, E.I., Wilkinson, M., Williamson, D. and Xenakis, G. 2023. A review of planting principles to identify the right place for the right tree for ‘net zero plus’ woodlands: Applying a place-based natural capital framework for sustainable, efficient and equitable (SEE) decisions. *People and Nature*. **5**(2), pp.271–301.

Bava, L., Zucali, M., Sandrucci, A. and Tamburini, A. 2017. Environmental impact of the typical heavy pig production in Italy. *Journal of Cleaner Production*. **140**, pp.685–691.

Bayat, A.R., Ventto, L., Kairenius, P., Stefański, T., Leskinen, H., Tapio, I., Negussie, E., Vilkki, J. and Shingfield, K.J. 2017. Dietary forage to concentrate ratio and sunflower oil supplement alter rumen fermentation, ruminal methane emissions, and nutrient utilization in lactating cows1. *Translational Animal Science*. **1**(3), pp.277–286.

Beauchemin, K.A., Ungerfeld, E.M., Abdalla, A.L., Alvarez, C., Arndt, C., Becquet, P., Benchaar, C., Berndt, A., Mauricio, R.M., McAllister, T.A., Oyhantçabal, W., Salami, S.A., Shalloo, L., Sun, Y., Tricarico, J., Uwizeye, A., Camillis, C.D., Bernoux, M., Robinson, T. and Kebreab, E. 2022. Invited review: Current enteric methane mitigation options. *Journal of Dairy Science*. **105**(12), pp.9297–9326.

Bell, M.J., Hinton, N., Cloy, J.M., Topp, C.F.E., Rees, R.M., Cardenas, L., Scott, T., Webster, C., Ashton, R.W., Whitmore, A.P., Williams, J.R., Balshaw, H., Paine, F., Goulding, K.W.T. and Chadwick, D.R. 2015. Nitrous oxide emissions from fertilised UK arable soils: Fluxes, emission factors and mitigation. *Agriculture, Ecosystems and Environment*. **212**, pp.134–147.

Bellarby, J., Tirado, R., Leip, A., Weiss, F., Lesschen, J.P. and Smith, P. 2013. Livestock greenhouse gas emissions and mitigation potential in Europe. *Global Change Biology*. **19**(1), pp.3–18.

Benchaar, C. and Greathead, H. 2011. Essential oils and opportunities to mitigate enteric methane emissions from ruminants. *Animal Feed Science and Technology*. **166–167**, pp.338–355.

Bernauer, T. 2013. Climate Change Politics. *Annual Review of Political Science*. **16**(1), pp.421–448.

Bernhard, A. 2010. The Nitrogen Cycle: Processes, Players, and Human Impact. *Nature Education*. [Online]. [Accessed 7 March 2022]. Available from: https://www.nature.com/scitable/knowledge/library/the-nitrogen-cycle-processes-players-and-human-15644632/.

Boehm, S., Jeffery, L., Hecke, J., Schumer, C., Jaeger, J., Fyson, C., Levin, K., Nilsson, A., Naimoli, S., Daly, E., Thwaites, J., Lebling, K., Waite, R., Collis, J., Sims, M., Singh, N., Grier, E., Lamb, W., Castellanos, S., Lee, A., Geffray, M.-C., Santo, R., Balehegn, M., Petroni, M. and Masterson, M. 2023. State of Climate Action 2023. *World Resources Institute*.

Brack, D. and King, R. 2020. *Net Zero and Beyond: What Role for Bioenergy with Carbon Capture and Storage?* [Online]. [Accessed 23 March 2020]. Available from: https://www.chathamhouse.org/publication/net-zero-and-beyond-what-role-bioenergy-carbon-capture-and-storage.

Brader, C. 2023. Mission zero: Independent review of net zero.

Bradfer-Lawrence, T., Finch, T., Bradbury, R.B., Buchanan, G.M., Midgley, A. and Field, R.H. 2021. The potential contribution of terrestrial nature-based solutions to a national ‘net zero’ climate target. *Journal of Applied Ecology*. **58**(11), pp.2349–2360.

Brentrup, F., Lammel, J., Stephani, T. and Christensen, B. 2018. Updated carbon footprint values for mineral fertilizer from different world regions *In*: *Global food challenges towards sustainable consumption and production* [Online]. Bangkok, Thailand. Available from: https://www.researchgate.net/publication/329774170_Updated_carbon_footprint_values_for_mineral_fertilizer_from_different_world_regions.

Brown, L., Armstrong Brown, S., Jarvis, S.C., Syed, B., Goulding, K.W.T., Phillips, V.R., Sneath, R.W. and Pain, B.F. 2001. An inventory of nitrous oxide emissions from agriculture in the UK using the IPCC methodology: emission estimate, uncertainty and sensitivity analysis. *Atmospheric Environment*. **35**(8), pp.1439–1449.

Brown, P., Cardenas, L., Del Vento, S., Karagianni, E., MacCarthy, J., Mullen, P., Passant, N. and Richmond, B. 2023. *UK Greenhouse Gas Inventory, 1990 to 2021*.

Büchi, L., Wendling, M., Amossé, C., Necpalova, M. and Charles, R. 2018. Importance of cover crops in alleviating negative effects of reduced soil tillage and promoting soil fertility in a winter wheat cropping system. *Agriculture, Ecosystems & Environment*. **256**, pp.92–104.

Buckingham, S., Topp, C.F.E., Smith, P., Eory, V., Chadwick, D.R., Baxter, C.K., Cloy, J.M., Connolly, S., Cooledge, E.C., Cowan, N.J., Drewer, J., Duffy, C., Fox, N.J., Jebari, A., Jenkins, B., Krol, D.J., Marsden, K.A., Mcauliffe, G.A., Morrison, S.J., O’flaherty, V., Ramsey, R., Richards, K.G., Roehe, R., Smith, J., Smith, K., Takahashi, T., Thorman, R.E., Williams, J., Wiltshire, J. and Rees, R.M. 2023.

Greenhouse Gas and Ammonia Emission Miigation Priorities for UK Policy Targets. *Frontiers of Agricultural Science and Engineering*. **10**(2), pp.268–280.

Burke, T., Rowland, C., Whyatt, J.D., Blackburn, G.A. and Abbatt, J. 2021. Achieving national scale targets for carbon sequestration through afforestation: Geospatial assessment of feasibility and policy implications. *Environmental Science & Policy*. **124**, pp.279–292.

Bustamante, M., Robledo-Abad, C., Harper, R., Mbow, C., Ravindranat, N.H., Sperling, F., Haberl, H., de Siqueira Pinto, A. and Smith, P. 2014. Co-benefits, trade-offs, barriers and policies for greenhouse gas mitigation in the agriculture, forestry and other land use (AFOLU) sector. *Global Change Biology*. **20**(10), pp.3270–3290.

Caffrey, K.R. and Veal, M.W. 2013. Conducting an Agricultural Life Cycle Assessment: Challenges and Perspectives. *The Scientific World Journal*. **2013**, pp.1–13.

Cain, M., Lynch, J., Allen, M.R., Fuglestvedt, J.S., Frame, D.J. and Macey, A.H. 2019. Improved calculation of warming-equivalent emissions for short-lived climate pollutants. *npj Climate and Atmospheric Science*. **2**(1), p.29.

Calzadilla, A., Rehdanz, K., Betts, R., Falloon, P., Wiltshire, A. and Tol, R.S.J. 2013. Climate change impacts on global agriculture. *Climatic Change*. **120**(1), pp.357–374.

Cantarello, E., Newton, A.C. and Hill, R.A. 2011. Potential effects of future land-use change on regional carbon stocks in the UK. *ENVIRONMENTAL SCIENCE & POLICY*. **14**(1), pp.40–52.

Carlson, K.M., Gerber, J.S., Mueller, N.D., Herrero, M., MacDonald, G.K., Brauman, K.A., Havlik, P., O'Connell, C.S., Johnson, J.A., Saatchi, S. and West, P.C. 2017. Greenhouse gas emissions intensity of global croplands. *Nature Climate Change*. **7**(1), pp.63–68.

CCC 2020a. *Land Use: Policies for a Net Zero UK* [Online]. [Accessed 23 January 2020]. Available from: https://www.theccc.org.uk/publication/land-use-policies-for-a-net-zero-uk/.

CCC 2018a. *Land use: Reducing emissions and preparing for climate change* [Online]. Committee on Climate Change. [Accessed 22 September 2022]. Available from: https://www.theccc.org.uk/wp-content/uploads/2018/11/Land-use-Reducing-emissions-and-preparing-for-climate-change-CCC-2018.pdf.

CCC 2019. *Net Zero – The UK's contribution to stopping global warming* [Online]. [Accessed 20 April 2020]. Available from: https://www.theccc.org.uk/publication/net-zero-the-uks-contribution-to-stopping-global-warming/.

CCC 2018b. *Reducing UK emissions: 2018 Progress Report to Parliament* [Online]. CCC. [Accessed 16 September 2022]. Available from: https://www.theccc.org.uk/wp-content/uploads/2018/06/CCC-2018-Progress-Report-to-Parliament.pdf.

CCC 2020b. *Sixth Carbon Budget: Agriculture and Land Use, Land Use Change and Forestry.pdf* [Online]. Committee on Climate Change. [Accessed 5 September 2022]. Available from: https://www.theccc.org.uk/wp-content/uploads/2020/12/Sector-summary-Agriculture-land-use-land-use-change-forestry.pdf.

CCC 2018c. *Technical Annex: The Smart Agriculture Inventory* [Online]. CCC. [Accessed 16 September 2022]. Available from: https://www.theccc.org.uk/wp-content/uploads/2018/08/PR18-Chapter-6-Annex-The-Smart-Agriculture-Inventory.pdf.

CCC 2025. *The Seventh Carbon Budget* [Online]. [Accessed 19 March 2025]. Available from: https://www.theccc.org.uk/wp-content/uploads/2025/02/The-Seventh-Carbon-Budget.pdf.

CEH 2020. UKCEH Land Cover® plus: Crops. [Accessed 30 September 2022]. Available from: https://www.ceh.ac.uk/data/ceh-land-cover-plus-crops-2015.

Ceschia, E., Béziat, P., Dejoux, J.F., Aubinet, M., Bernhofer, Ch., Bodson, B., Buchmann, N., Carrara, A., Cellier, P., Di Tommasi, P., Elbers, J.A., Eugster, W., Grünwald, T., Jacobs, C.M.J., Jans, W.W.P., Jones, M., Kutsch, W., Lanigan, G., Magliulo, E., Marloie, O., Moors, E.J., Moureaux, C., Olioso, A., Osborne, B., Sanz, M.J., Saunders, M., Smith, P., Soegaard, H. and Wattenbach, M. 2010. Management effects on net ecosystem carbon and GHG budgets at European crop sites. *Agriculture, Ecosystems & Environment*. **139**(3), pp.363–383.

Chadwick, D., Sommer, S., Thorman, R., Fangueiro, D., Cardenas, L., Amon, B. and Misselbrook, T. 2011. Manure management: Implications for greenhouse gas emissions. *Animal Feed Science and Technology*. **166–167**, pp.514–531.

Challinor, A.J., Ewert, F., Arnold, S., Simelton, E. and Fraser, E. 2009. Crops and climate change: progress, trends, and challenges in simulating impacts and informing adaptation. *Journal of Experimental Botany*. **60**(10), pp.2775–2789.

Challinor, A.J., Müller, C., Asseng, S., Deva, C., Nicklin, K.J., Wallach, D., Vanuytrecht, E., Whitfield, S., Ramirez-Villegas, J. and Koehler, A.-K. 2018. Improving the use of crop models for risk assessment and climate change adaptation. *Agricultural Systems*. **159**, pp.296–306.

Chambers, A., Lal, R. and Paustian, K. 2016. Soil carbon sequestration potential of US croplands and grasslands: Implementing the 4 per Thousand Initiative. *Journal of Soil and Water Conservation*. **71**(3), pp.68A-74A.

Chan, G., Stavins, R. and Ji, Z. 2018. International Climate Change Policy. *Annual Review of Resource Economics*. **10**(1), pp.335–360.

CHAP 2022. *Benchmarking greenhouse gas emissions for the UK arable and horticultural sector* [Online]. [Accessed 24 August 2024]. Available from: https://chap-solutions.co.uk/wp-content/uploads/2022/08/CHAP_Net_Zero_Report_0822.pdf.

Chen, C., Riley, W.J., Prentice, I.C. and Keenan, T.F. 2022. CO2 fertilization of terrestrial photosynthesis inferred from site to global scales. *Proceedings of the National Academy of Sciences*. **119**(10), p.e2115627119.

Cheng, M., McCarl, B. and Fei, C. 2022. Climate Change and Livestock Production: A Literature Review. *Atmosphere*. **13**(1), p.140.

Ciroth, A., Muller, S., Weidema, B. and Lesage, P. 2016. Empirically based uncertainty factors for the pedigree matrix in ecoinvent. *The International Journal of Life Cycle Assessment*. **21**(9), pp.1338–1348.

Climate Action Tracker 2023. The CAT Thermometer. *Climate Action Tracker*. [Online]. Available from: https://climateactiontracker.org/global/cat-thermometer/.

Colomb, V., Touchemoulin, O., Bockel, L., Chotte, J.-L., Martin, S., Tinlot, M. and Bernoux, M. 2013. Selection of appropriate calculators for landscape-scale greenhouse gas assessment for agriculture and forestry. *Environmental Research Letters*. **8**(1), p.015029.

Costigan, H., Shalloo, L., Egan, M., Kennedy, M., Dwan, C., Walsh, S., Hennessy, D., Walker, N., Zihlmann, R. and Lahart, B. 2024. The impact of twice daily 3-nitroxypropanol supplementation on enteric methane emissions in grazing dairy cows. *Journal of Dairy Science*.

Cowan, N., Carnell, E., Skiba, U., Dragosits, U., Drewer, J. and Levy, P. 2020. Nitrous oxide emission factors of mineral fertilisers in the UK and Ireland: A Bayesian analysis of 20 years of experimental data. *Environment International*. **135**, p.105366.

Cranfield Soil and Agrifood Institute 2010. LandIS Soilscapes Viewer. *Soilscapes*. [Online]. [Accessed 10 July 2020]. Available from: http://www.landis.org.uk/soilscapes/.

Dalby, F.R., Guldberg, L.B., Feilberg, A. and Kofoed, M.V.W. 2022. Reducing greenhouse gas emissions from pig slurry by acidification with organic and inorganic acids. *PLOS ONE*. **17**(5), p.e0267693.

Dawson, J.J.C. and Smith, P. 2007. Carbon losses from soil and its consequences for land-use management. *Science of The Total Environment*. **382**(2), pp.165–190.

Defra 2021a. Agri-climate report 2021. *GOV.UK*. [Online]. [Accessed 6 April 2022]. Available from: https://www.gov.uk/government/statistics/agri-climate-report-2021/agri-climate-report-2021.

Defra 2023a. Agri-climate report 2023. *GOV.UK*. [Online]. [Accessed 9 March 2024]. Available from: https://www.gov.uk/government/statistics/agri-climate-report-2023/agri-climate-report-2023.

Defra 2018. *Agricultural Bill: Analysis of the impacts of removing Direct Payments* [Online]. [Accessed 27 April 2020]. Available from: https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/740669/agri-bill-evidence-slide-pack-direct-payments.pdf.

Defra 2024a. Agricultural facts: Summary. *GOV.UK*. [Online]. [Accessed 12 July 2024]. Available from: https://www.gov.uk/government/statistics/agricultural-facts-england-regional-profiles/agricultural-facts-summary.

Defra 2019a. *Agricultural Statistics and Climate Change* [Online]. [Accessed 24 April 2020]. Available from: https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/835762/agriclimate-9edition-02oct19.pdf.

Defra 2023b. Cereal and oilseed rape production. *GOV.UK*. [Online]. [Accessed 24 August 2024]. Available from: https://www.gov.uk/government/statistics/cereal-and-oilseed-rape-production.

Defra 2019b. *Clean Air Strategy 2019* [Online]. Available from: https://assets.publishing.service.gov.uk/media/5c3b9debe5274a70c19d905c/clean-air-strategy-2019.pdf.

Defra 2019c. *Crops Grown For Bioenergy in the UK: 2018* [Online]. [Accessed 14 April 2020]. Available from: https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/856695/nonfood-statsnotice2018-08jan20.pdf.

Defra 2024b. Farm Practices Survey. [Accessed 21 July 2024]. Available from: https://www.gov.uk/government/collections/farm-practices-survey.

Defra 2024c. Find out what's available in SFI. *GOV.UK*. [Online]. [Accessed 31 August 2024]. Available from: https://www.gov.uk/guidance/find-out-whats-available-in-sfi.

Defra 2022a. Greenhouse gas mitigation practices – Farm Practices Survey England 2022. Available from: https://www.gov.uk/government/statistics/farm-practices-survey-february-2022-greenhouse-gas-mitigation-practices.

Defra 2023c. *Methane suppressing feed products: call for evidence - Summary of responses* [Online]. [Accessed 13 August 2024]. Available from: https://www.gov.uk/government/calls-for-evidence/methane-suppressing-feed-products-call-for-evidence/outcome/summary-of-responses.

Defra 2021b. Nitrate vulnerable zones. *GOV.UK*. [Online]. [Accessed 27 July 2024]. Available from: https://www.gov.uk/government/collections/nitrate-vulnerable-zones.

Defra 2024d. Provisional cereal and oilseed production estimates for England 2024. *GOV.UK*. [Online]. [Accessed 2 February 2025]. Available from: https://www.gov.uk/government/statistics/cereal-and-oilseed-rape-production/provisional-cereal-and-oilseed-production-estimates-for-england-2024.

Defra 2024e. Structure of the agricultural industry in England and the UK at June. [Accessed 20 July 2024]. Available from: https://www.gov.uk/government/statistical-data-sets/structure-of-the-agricultural-industry-in-england-and-the-uk-at-june.

Defra 2022b. *The British Survey of Fertiliser Practice: Fertiliser use on farm crops for crop year 2021* [Online]. London, UK: Department for Environment, Food & Rural Affairs. [Accessed 17 October 2022]. Available from: https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/1094283/fertiliseruse-annualreport2021-28jul22.pdf.

Department for Business, Energy and Industrial Strategy 2022a. *2020 UK Greenhouse Gas Emissions, Final Figures* [Online]. Available from: https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/1051408/2020-final-greenhouse-gas-emissions-statistical-release.pdf.

Department for Business, Energy and Industrial Strategy 2022b. *UK local authority and regional greenhouse gas emissions national statistics, 2005 to 2020* [Online]. London, UK: UK Government. Available from: https://www.gov.uk/government/statistics/uk-local-authority-and-regional-greenhouse-gas-emissions-national-statistics-2005-to-2020.

DESNZ 2024. *2022 UK Greenhouse Gas Emissions, Final Figures*.

DESNZ 2023. Greenhouse gas reporting: conversion factors 2023. *GOV.UK*. [Online]. [Accessed 24 September 2023]. Available from: https://www.gov.uk/government/publications/greenhouse-gas-reporting-conversion-factors-2023.

Dicks, L.V., Rose, D.C., Ang, F., Aston, S., Birch, A.N.E., Boatman, N., Bowles, E.L., Chadwick, D., Dinsdale, A., Durham, S., Elliott, J., Firbank, L., Humphreys, S., Jarvis, P., Jones, D., Kindred, D., Knight, S.M., Lee, M.R.F., Leifert, C., Lobley, M., Matthews, K., Midmer, A., Moore, M., Morris, C., Mortimer, S., Murray, T.C., Norman, K., Ramsden, S., Roberts, D., Smith, L.G., Soffe, R., Stoate, C., Taylor, B., Tinker, D., Topliff, M., Wallace, J., Williams, P., Wilson, P., Winter, M. and Sutherland, W.J. 2019. What agricultural practices are most likely to deliver "sustainable intensification" in the UK? *Food and Energy Security*. **8**(1), p.e00148.

Dijkstra, J., Bannink, A., France, J., Kebreab, E. and Gastelen, S. van 2018. Short communication: Antimethanogenic effects of 3-nitrooxypropanol depend on supplementation dose, dietary fiber content, and cattle type. *Journal of Dairy Science*. **101**(10), pp.9041–9047.

Donkersley, P., Carver, L. and Wentworth, J. 2021. *Sustainable land management: managing land better for environmental benefits*. Westminster, UK: Parliamentary Office of Science and Technology.

Dourmad, J.Y., Ryschawy, J., Trousson, T., Bonneau, M., Gonzàlez, J., Houwers, H.W.J., Hviid, M., Zimmer, C., Nguyen, T.L.T. and Morgensen, L. 2014. Evaluating environmental impacts of contrasting pig farming systems with life cycle assessment. *animal*. **8**(12), pp.2027–2037.

dsm-firmenich 2024. Misinformation about Bovaer®. [Accessed 26 February 2025]. Available from: https://our-company.dsm-firmenich.com/en/our-company/news/updates-and-statements/2025/misinformation-about-bovaer.html.

Duhan, J.S., Kumar, R., Kumar, N., Kaur, P., Nehra, K. and Duhan, S. 2017. Nanotechnology: The new perspective in precision agriculture. *Biotechnology Reports*. **15**, pp.11–23.

Dyke, J., Watson, R. and Knorr, W. 2021. Climate Scientists: concept of net zero is a dangerous trap. *The Conversation*. [Online]. Available from: https://theconversation.com/climate-scientists-concept-of-net-zero-is-a-dangerous-trap-157368.

Eory, V, Maire, J., MacLeod, M., Sykes, A., Barnes, A., Rees, R.M., Topp, C.F.E. and Wall, E. 2020. *Non-CO2 abatement in the UK agricultural sector by 2050* [Online]. SRUC. [Accessed 7 September 2022]. Available from: https://www.theccc.org.uk/wp-content/uploads/2020/12/Non-CO2-abatement-in-the-UK-agricultural-sector-by-2050-Scottish-Rural-College.pdf.

Eory, V., Pellerin, S., Carmona Garcia, G., Lehtonen, H., Licite, I., Mattila, H., Lund-Sørensen, T., Muldowney, J., Popluga, D., Strandmark, L. and Schulte, R. 2018. Marginal abatement cost curves for agricultural climate policy: State-of-the art, lessons learnt and future potential. *Journal of Cleaner Production*. **182**, pp.705–716.

Eory, Vera, Topp, K., Rees, B., Leinonen, I. and Maire, J. 2020. Marginal abatement cost curve for Scottish agriculture.

European Commission 2019. 2050 long-term strategy - European Commission. [Accessed 24 August 2024]. Available from: https://climate.ec.europa.eu/eu-action/climate-strategies-targets/2050-long-term-strategy_en.

European Environment Agency 2019. Atmospheric greenhouse gas concentrations — European Environment Agency. [Accessed 5 January 2023]. Available from: https://www.eea.europa.eu/data-and-maps/indicators/atmospheric-greenhouse-gas-concentrations-6/assessment.

European Environment Agency 2021. Protecting the ozone layer while also preventing climate change — European Environment Agency. [Accessed 5 January 2023]. Available from: https://www.eea.europa.eu/themes/climate/ozone-depleting-substances-and-climate-change.

Fajardy, M., Köberle, A., Dowell, N.M. and Fantuzzi, A. 2019. *BECCS deployment: a reality check* [Online]. [Accessed 14 April 2020]. Available from: https://www.researchgate.net/profile/Andrea_Fantuzzi/publication/330774659_BECCS-deployment-_a-reality-check/links/5c539d81a6fdccd6b5d87347/BECCS-deployment-a-reality-check.pdf.

Falloon, P., Falloon, P., Powlson, D. and Smith, P. 2004. Managing field margins for biodiversity and carbon sequestration: a Great Britain case study. *Soil Use and Management*. **20**(2), pp.240–247.

Fangueiro, D., Hjorth, M. and Gioelli, F. 2015. Acidification of animal slurry– a review. *Journal of Environmental Management*. **149**, pp.46–56.

FAO 2021. *Emissions from agriculture and forest land. Global, regional and country trends 1990-2019*. Rome: FAO.

FAO 2022. GLEAM v3.0 dashboard. [Accessed 10 October 2022]. Available from: https://foodandagricultureorganization.shinyapps.io/GLEAMV3_Public/.

FAO 2023a. Global agricultural land use by major crop type. [Accessed 31 August 2024]. Available from: https://ourworldindata.org/grapher/global-agricultural-land-use-by-major-crop-type.

FAO 2017a. *Livestock solutions for climate change* [Online]. Rome: FAO. Available from: https://www.fao.org/3/I8098EN/i8098en.pdf.

FAO 2023b. *Methane emissions in livestock and rice systems* [Online]. FAO. [Accessed 7 January 2025]. Available from: http://www.fao.org/documents/card/en/c/cc7607en.

FAO 2023c. *Pathways towards lower emissions* [Online]. FAO. [Accessed 13 July 2024]. Available from: http://www.fao.org/documents/card/en/c/cc9029en.

FAO (ed.). 2017b. *The future of food and agriculture: trends and challenges* [Online]. Rome: Food and Agriculture Organization of the United Nations. Available from: http://www.fao.org/publications/fofa/en/.

Fellmann, T., Domínguez, I.P., Witzke, P., Weiss, F., Hristov, J., Barreiro-Hurle, J., Leip, A. and Himics, M. 2021. Greenhouse gas mitigation technologies in agriculture:

Regional circumstances and interactions determine cost-effectiveness. *Journal of Cleaner Production*. **317**, p.128406.

Feng, X.Y., Dijkstra, J., Bannink, A., van Gastelen, S., France, J. and Kebreab, E. 2020. Antimethanogenic effects of nitrate supplementation in cattle: A meta-analysis. *Journal of Dairy Science*. **103**(12), pp.11375–11385.

Flack, J., Lukac, M. and Todman, L. 2022. Woodland planting on UK pasture land is not economically feasible, yet is more profitable than some traditional farming practices. *Central European Forestry Journal*. **68**(2), pp.61–71.

FOOD AND AGRICULTURE ORGANIZATION OF THE UNITED NATIONS 2019. *World food and agriculture - statistical pocketbook 2019*. Place of publication not identified: FOOD & AGRICULTURE ORG.

Food Standards Agency 2024. Bovaer cow feed additive explained. [Accessed 26 February 2025]. Available from: https://food.blog.gov.uk/2024/12/05/bovaer-cow-feed-additive-explained/.

Food Standards Agency 2023. *RP1059 Outcome of assessment of 3- Nitrooxypropanol "3-NOP" - Summary* [Online]. [Accessed 7 August 2024]. Available from: https://www.food.gov.uk/print/pdf/node/15911.

Forest Research 2024. Provisional Woodland Statistics 2024. [Accessed 25 August 2024]. Available from: https://www.forestresearch.gov.uk/tools-and-resources/statistics/statistics-by-topic/woodland-statistics/.

Forestry Commission 2024a. A guide to Forestry Commission's sensitivity maps for woodland creation. *GOV.UK*. [Online]. [Accessed 13 June 2024]. Available from: https://www.gov.uk/guidance/a-guide-to-forestry-commissions-sensitivity-maps-for-woodland-creation.

Forestry Commission 2024b. *Creating new woodland: The England Woodland Creation Offer* [Online]. [Accessed 29 August 2024]. Available from: https://assets.publishing.service.gov.uk/media/66977ccd0808eaf43b50d0c0/FC_Woodland_Creation_A5_April_2024_HP_V2_WEB.pdf.

Forestry Commission 2024c. England Woodland Creation Full Sensitivity Map v 4.0. Available from: https://data-forestry.opendata.arcgis.com/datasets/ffc2d647a1934f75914a9a45bc74c86d_0/explore?location=52.723884%2C-2.489483%2C6.19.

Forrestal, P.J., Somers, C., Plunkett, M., Wall, D. and O'Dwyer, T. 2019. Protected urea: what is it, does it work, and is it cost effective? Available from: https://www.teagasc.ie/media/website/crops/soil-and-soil-fertility/Q--A-Protected-Urea-April-2019.pdf.

Forster, P., Storelvmo, T., Armour, K., Collins, W., Dufresne, J.L., Frame, D., Lunt, D.J., Mauritsen, T., Palmer, M.D., Watanabe, M., Wild, M. and Zhang, H. 2021. *The Earth's Energy Budget, Climate Feedbacks, and Climate Sensitivity* [Online]. Cambridge, UK: Cambridge University Press. Available from: https://www.ipcc.ch/report/ar6/wg1/downloads/report/IPCC_AR6_WGI_Chapter_07.pdf.

Fowler, D., Steadman, C.E., Stevenson, D., Coyle, M., Rees, R.M., Skiba, U.M., Sutton, M.A., Cape, J.N., Dore, A.J., Vieno, M., Simpson, D., Zaehle, S., Stocker, B.D., Rinaldi, M., Facchini, M.C., Flechard, C.R., Nemitz, E., Twigg, M., Erisman, J.W., Butterbach-Bahl, K. and Galloway, J.N. 2015. Effects of global change during the 21st century on the nitrogen cycle. *ATMOSPHERIC CHEMISTRY AND PHYSICS*. **15**(24), pp.13849–13893.

Fraanje, W. and Garnett, T. 2020. *Soy: food, feed, and land use change* [Online]. Oxfordshire: UK: Food Climate Research Network. [Accessed 18 February 2025]. Available from: https://tabledebates.org/building-blocks/soy-food-feed-and-land-use-change.

Freeman, D., Wiltshire, J. and Jenkins, B. 2020. Evidence review of the efficacy of nitrification and urease inhibitors.

Friedlingstein, P., O'Sullivan, M., Jones, M.W., Andrew, R.M., Bakker, D.C.E., Hauck, J., Landschützer, P., Le Quéré, C., Luijkx, I.T., Peters, G.P., Peters, W., Pongratz, J., Schwingshackl, C., Sitch, S., Canadell, J.G., Ciais, P., Jackson, R.B., Alin, S.R., Anthoni, P., Barbero, L., Bates, N.R., Becker, M., Bellouin, N., Decharme, B., Bopp, L., Brasika, I.B.M., Cadule, P., Chamberlain, M.A., Chandra, N., Chau, T.-T.-T., Chevallier, F., Chini, L.P., Cronin, M., Dou, X., Enyo, K., Evans, W., Falk, S., Feely, R.A., Feng, L., Ford, D.J., Gasser, T., Ghattas, J., Gkritzalis, T., Grassi, G., Gregor, L., Gruber, N., Gürses, Ö., Harris, I., Hefner, M., Heinke, J., Houghton, R.A., Hurtt, G.C., Iida, Y., Ilyina, T., Jacobson, A.R., Jain, A., Jarníková, T., Jersild, A., Jiang, F., Jin, Z., Joos, F., Kato, E., Keeling, R.F., Kennedy, D., Klein Goldewijk, K., Knauer, J., Korsbakken, J.I., Körtzinger, A., Lan, X., Lefèvre, N., Li, H., Liu, J., Liu, Z., Ma, L., Marland, G., Mayot, N., McGuire, P.C., McKinley, G.A., Meyer, G., Morgan, E.J., Munro, D.R., Nakaoka, S.-I., Niwa, Y., O'Brien, K.M., Olsen, A., Omar, A.M., Ono, T., Paulsen, M., Pierrot, D., Pocock, K., Poulter, B., Powis, C.M., Rehder, G., Resplandy, L., Robertson, E., Rödenbeck, C., Rosan, T.M., Schwinger, J., Séférian, R., Smallman, T.L., Smith, S.M., Sospedra-Alfonso, R., Sun, Q., Sutton, A.J., Sweeney, C., Takao, S., Tans, P.P., Tian, H., Tilbrook, B., Tsujino, H., Tubiello, F., van der Werf, G.R., van Ooijen, E., Wanninkhof, R., Watanabe, M., Wimart-Rousseau, C., Yang, D., Yang, X., Yuan, W., Yue, X., Zaehle, S., Zeng, J. and Zheng, B. 2023. Global Carbon Budget 2023. *Earth System Science Data*. **15**(12), pp.5301–5369.

Garcia-Launay, F., Dusart, L., Espagnol, S., Laisse-Redoux, S., Gaudré, D., Méda, B. and Wilfart, A. 2018. Multiobjective formulation is an effective method to reduce environmental impacts of livestock feeds. *British Journal of Nutrition*. **120**(11), pp.1298–1309.

Garnett, T. 2009. Livestock-related greenhouse gas emissions: impacts and options for policy makers. *Environmental Science & Policy*. **12**(4), pp.491–503.

Garnett, T. 2011. Where are the best opportunities for reducing greenhouse gas emissions in the food system (including the food chain)? *Food Policy*. **36**, pp.S23–S32.

Gebbers, R. and Adamchuk, V.I. 2010. Precision Agriculture and Food Security. *Science*. **327**(5967), pp.828–831.

Gerber, J.S., Carlson, K.M., Makowski, D., Mueller, N.D., de Cortazar-Atauri, I.G., Havlik, P., Herrero, M., Launay, M., O'Connell, C.S., Smith, P. and West, P.C. 2016. Spatially explicit estimates of N2O emissions from croplands suggest climate mitigation opportunities from improved fertilizer management. *GLOBAL CHANGE BIOLOGY*. **22**(10), pp.3383–3394.

Gerber, P.J., Steinfeld, H., Henderson, B., Mottet, A., Opio, C., Dijkman, J., Falcucci, A. and Tempio, G. 2013. *Tackling climate change through livestock: a global assessment of emissions and mitigation opportunities*. Rome: Food and Agriculture Organization of the United Nations.

Glithero, N.J., Wilson, P. and Ramsden, S.J. 2013. Straw use and availability for second generation biofuels in England. *Biomass and Bioenergy*. **55**, pp.311–321.

Godfray, H.C.J., Beddington, J.R., Crute, I.R., Haddad, L., Lawrence, D., Muir, J.F., Pretty, J., Robinson, S., Thomas, S.M. and Toulmin, C. 2010. Food Security: The Challenge of Feeding 9 Billion People. *Science*. **327**(5967), pp.812–818.

Gooday, R.D., Anthony, S.G., Durrant, C., Harris, D., Lee, D., Metcalfe, P., Newell Price, J.P. and Turner, A. 2015. *Farmscoper extension* [Online]. Cambridge, UK: ADAS. Available from: https://ictagrifood.eu/node/36227.

Goodwin, C.E.D., Bütikofer, L., Hatfield, J.H., Evans, P.M., Bullock, J.M., Storkey, J., Mead, A., Richter, G.M., Henrys, P.A., Pywell, R.F. and Redhead, J.W. 2022. Multi-tier archetypes to characterise British landscapes, farmland and farming practices. *Environmental Research Letters*. **17**(9), p.095002.

Graham, L., Gaulton, R., Gerard, F. and Staley, J.T. 2018. The influence of hedgerow structural condition on wildlife habitat provision in farmed landscapes. *Biological Conservation*. **220**, pp.122–131.

Gruninger, R.J., Zhang, X.M., Smith, M.L., Kung, L., Vyas, D., McGinn, S.M., Kindermann, M., Wang, M., Tan, Z.L. and Beauchemin, K.A. 2022. Application of 3-nitrooxypropanol and canola oil to mitigate enteric methane emissions of beef cattle results in distinctly different effects on the rumen microbial community. *Animal Microbiome*. **4**(1), p.35.

Guo, C., Liu, X. and He, X. 2022. A global meta-analysis of crop yield and agricultural greenhouse gas emissions under nitrogen fertilizer application. *Science of The Total Environment*. **831**, p.154982.

Haisan, J., Sun, Y., Guan, L.L., Beauchemin, K.A., Iwaasa, A., Duval, S., Barreda, D.R. and Oba, M. 2014. The effects of feeding 3-nitrooxypropanol on methane emissions and productivity of Holstein cows in mid lactation. *Journal of Dairy Science*. **97**(5), pp.3110–3119.

Hale, T., Smith, S.M., Black, R., Cullen, K., Fay, B., Lang, J. and Mahmood, S. 2022. Assessing the rapidly-emerging landscape of net zero targets. *Climate Policy*. **22**(1), pp.18–29.

Hama-Aziz, Z.Q., Hiscock, K.M. and Cooper, R.J. 2017. Indirect Nitrous Oxide Emission Factors for Agricultural Field Drains and Headwater Streams. *Environmental Science and Technology*. **51**(1), pp.301–307.

Hamza, M.A. and Anderson, W.K. 2005. Soil compaction in cropping systems. *Soil and Tillage Research*. **82**(2), pp.121–145.

Hansen, M.N., Henriksen, K. and Sommer, S.G. 2006. Observations of production and emission of greenhouse gases and ammonia during storage of solids separated from pig slurry: Effects of covering. *Atmospheric Environment*. **40**(22), pp.4172–4181.

Helm, D. 2017. Agriculture after Brexit. *Oxford Review of Economic Policy*. **33**(suppl_1), pp.S124–S133.

Herrero, M., Henderson, B., Havlík, P., Thornton, P.K., Conant, R.T., Smith, P., Wirsenius, S., Hristov, A.N., Gerber, P., Gill, M., Butterbach-Bahl, K., Valin, H., Garnett, T. and Stehfest, E. 2016. Greenhouse gas mitigation potentials in the livestock sector. *Nature Climate Change*. **6**(5), pp.452–461.

Hillel, D. and Rosenzweig, C. (eds.). 2010. *Handbook of climate change and agroecosystems: impacts, adaptation, and mitigation*. London : Singapore ; Hackensack, NJ: Imperial College Press ; Distributed by World Scientific Publishing Co.

Hristov, A.N., Oh, J., Firkins, J.L., Dijkstra, J., Kebreab, E., Waghorn, G., Makkar, H.P.S., Adesogan, A.T., Yang, W., Lee, C., Gerber, P.J., Henderson, B. and Tricarico, J.M. 2013. SPECIAL TOPICS — Mitigation of methane and nitrous oxide emissions from animal operations: I. A review of enteric methane mitigation options1. *Journal of Animal Science*. **91**(11), pp.5045–5069.

Hristov, A.N., Oh, J., Giallongo, F., Frederick, T.W., Harper, M.T., Weeks, H.L., Branco, A.F., Moate, P.J., Deighton, M.H., Williams, S.R.O., Kindermann, M. and Duval, S. 2015. An inhibitor persistently decreased enteric methane emission from dairy cows with no negative effect on milk production. *Proceedings of the National Academy of Sciences*. **112**(34), pp.10663–10668.

Huntingford, C. and Oliver, R.J. 2021. Constraints on estimating the CO2 fertilization effect emerge. *Nature*. **600**(7888), pp.224–225.

Hurley, P., Lyon, J., Hall, J., Little, R., Tsouvalis, J. and Rose, D. 2020. Co-designing the Environmental Land Management Scheme in England: the why, who, and how of engaging 'harder to reach' stakeholders.

IEA 2022. Global Methane Tracker 2022. Available from: https://www.iea.org/reports/global-methane-tracker-2022.

IEA 2024a. Global Methane Tracker 2024. Available from: https://www.iea.org/data-and-statistics/data-tools/methane-tracker.

IEA 2024b. Main sources of methane emissions. Available from: https://www.iea.org/data-and-statistics/charts/main-sources-of-methane-emissions.

Innovation for Agriculture 2021. *Reducing Greenhouse Gas Emissions at Farm Level: The Go-To Guide* [Online]. [Accessed 7 September 2022]. Available from: https://www.wwf.org.uk/sites/default/files/2021-12/I4Agri_FarmerFacingGuide_Print_Sub1.pdf.

IPCC 2006. *2006 IPCC Guidelines for National Greenhouse Gas Inventories* [Online]. Switzerland: IPCC. [Accessed 1 March 2020]. Available from: https://www.ipcc-nggip.iges.or.jp/public/2006gl/index.html.

IPCC 2019a. *2019 Refinement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories* [Online]. Switzerland: IPCC. [Accessed 14 July 2022]. Available from: https://www.ipcc-nggip.iges.or.jp/public/2019rf/pdf/4_Volume4/19R_V4_Cover.pdf.

IPCC 2001. *Climate change 2001: the scientific basis: contribution of Working Group I to the third assessment report of the Intergovernmental Panel on Climate Change* (J. T. Houghton, ed.). Cambridge ; New York: Cambridge University Press.

IPCC (ed.). 2007. *Climate change 2007: The Physical Science Basis*. Cambridge: Cambridge university press.

IPCC 2013. *Climate change 2013: The Physical Science Basis: Working Group I contribution to the Fifth assessment report of the Intergovernmental Panel on Climate Change* (T. F. Stocker, D. Qin, G.-K. Plattner, M. Tignor, S. K. Allen, J. Boschung, A. Nauels, Y. Xia, V. Bex, & P. M. Midgley, eds.). New York: Cambridge University Press.

IPCC 2021. *Climate Change 2021: The Physical Science Basis. Contribution of Working Group I to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change* [Online]. Cambridge, UK and New York, USA: IPCC. Available from: https://report.ipcc.ch/ar6/wg1/IPCC_AR6_WGI_FullReport.pdf.

IPCC 2022. *Climate Change 2022: Impacts, Adaptation and Vulnerability. Contribution of Working Group II to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change* [Online]. Cambridge, UK and New York, USA: IPCC. Available from: https://report.ipcc.ch/ar6/wg2/IPCC_AR6_WGII_FullReport.pdf.

IPCC 2019b. *Summary for Policymakers* [Online]. IPCC. [Accessed 22 September 2022]. Available from: https://www.ipcc.ch/srccl/chapter/summary-for-policymakers/.

IPCC, I.P. on C.C. 2018. *Global Warming of 1.5°C. An IPCC Special Report on the impacts of global warming of 1.5°C above pre-industrial levels and related global greenhouse gas emission pathways, in the context of strengthening the global response to the threat of climate change, sustainable development, and efforts to eradicate poverty* [Online]. [Accessed 23 March 2020]. Available from: https://www.ipcc.ch/site/assets/uploads/sites/2/2019/06/SR15_Full_Report_High_Res.pdf.

Jacquet, J. and Jamieson, D. 2016. Soft but significant power in the Paris Agreement. *Nature Climate Change*. **6**(7), pp.643–646.

Jensen, E.S., Chongtham, I.R., Dhamala, N.R., Rodriguez, C., Carton, N. and Carlsson, G. 2020. Diversifying European agricultural systems by intercropping grain legumes and cereals. *INTERNATIONAL JOURNAL OF AGRICULTURE AND NATURAL RESOURCES*. **47**(3), pp.174–186.

Jolliet, O., Saadé-Sbeih, M., Shaked, S., Jolliet, A. and Crettaz, P. 2016. *Environmental life cycle assessment*. London: Taylor & Francis.

Karl, T.R. and Trenberth, K.E. 2003. Modern Global Climate Change. *Science*. **302**(5651), pp.1719–1723.

Kaye, J.P. and Quemada, M. 2017. Using cover crops to mitigate and adapt to climate change. A review. *Agronomy for Sustainable Development*. **37**(1), p.4.

Kibblewhite, M. g, Ritz, K. and Swift, M. j 2008. Soil health in agricultural systems. *Philosophical Transactions of the Royal Society B: Biological Sciences*. **363**(1492), pp.685–701.

Klik, A. and Eitzinger, J. 2010. Impact of climate change on soil erosion and the efficiency of soil conservation practices in Austria. *The Journal of Agricultural Science*. **148**(5), pp.529–541.

Kool, A., Blonk, H., Ponsioen, T.C., Sukkel, W., Vermeer, H.M., de Vries, J.W. and Hoste, R. 2009. *Carbon footprints of conventional and organic pork : assessments of typical production systems in the Netherlands, Denmark, England and GermanyCarbon footprints of conventional and organic pork : assessments of typical production systems in the Netherlands, Denmark, England and Germany* [Online]. Wageningen Livestock Research. Available from: https://tinyurl.com/777khkjr.

Kuyper, J., Schroeder, H. and Linnér, B.-O. 2018. The Evolution of the UNFCCC. *Annual Review of Environment and Resources*. **43**(Volume 43, 2018), pp.343–368.

Kweku, D., Bismark, O., Maxwell, A., Desmond, K., Danso, K., Oti-Mensah, E., Quachie, A. and Adormaa, B. 2018. Greenhouse Effect: Greenhouse Gases and Their Impact on Global Warming. *Journal of Scientific Research and Reports*. **17**(6), pp.1–9.

Lal, R. 2004. Carbon emission from farm operations. , pp.981–990.

Lal, R. 1997. Degradation and resilience of soils. *Philosophical Transactions of the Royal Society of London. Series B: Biological Sciences*. **352**(1356), pp.997–1010.

Lal, R. 2003. Global potential of soil carbon sequestration to mitigate the greenhouse effect. *Critical reviews in plant sciences*. **22**(2), pp.151–184.

Lal, R. 2010. Managing Soils and Ecosystems for Mitigating Anthropogenic Carbon Emissions and Advancing Global Food Security. *BioScience*. **60**(9), pp.708–721.

Lal, R. 2020. Managing soils for negative feedback to climate change and positive impact on food and nutritional security. *Soil Science and Plant Nutrition*. **66**(1), pp.1–9.

Lal, R. 2011. Sequestering carbon in soils of agro-ecosystems. *Food Policy*. **36**, pp.S33–S39.

Lal, R. 2016. Soil health and carbon management. *Food and Energy Security*. **5**(4), pp.212–222.

Lal, R., Smith, P., Jungkunst, H.F., Mitsch, W.J., Lehmann, J., Nair, P.K.R., McBratney, A.B., de Moraes Sá, J.C., Schneider, J., Zinn, Y.L., Skorupa, A.L.A., Zhang, H.-L., Minasny, B., Srinivasrao, C. and Ravindranath, N.H. 2018. The carbon sequestration potential of terrestrial ecosystems. *Journal of Soil and Water Conservation*. **73**(6), pp.145A-152A.

Lal, R. and Stewart, B.A. 2010. *Food security and soil quality* [Online]. Boca Raton: CRC Press. [Accessed 28 April 2020]. Available from: http://www.crcnetbase.com/doi/book/10.1201/EBK1439800577.

Lam, S.K., Suter, H., Mosier, A.R. and Chen, D. 2017. Using nitrification inhibitors to mitigate agricultural N2O emission: a double-edged sword? *Global Change Biology*. **23**(2), pp.485–489.

Lamb, A., Green, R., Bateman, I., Broadmeadow, M., Bruce, T., Burney, J., Carey, P., Chadwick, D., Crane, E., Field, R., Goulding, K., Griffiths, H., Hastings, A., Kasoar, T., Kindred, D., Phalan, B., Pickett, J., Smith, P., Wall, E., zu Ermgassen, E.K.H.J. and Balmford, A. 2016. The potential for land sparing to offset greenhouse gas emissions from agriculture. *Nature Climate Change*. **6**(5), pp.488–492.

Le Quéré, C., Andrew, R.M., Friedlingstein, P., Sitch, S., Hauck, J., Pongratz, J., Pickers, P.A., Korsbakken, J.I., Peters, G.P., Canadell, J.G., Arneth, A., Arora, V.K., Barbero, L., Bastos, A., Bopp, L., Chevallier, F., Chini, L.P., Ciais, P., Doney, S.C., Gkritzalis, T., Goll, D.S., Harris, I., Haverd, V., Hoffman, F.M., Hoppema, M., Houghton, R.A., Hurtt, G., Ilyina, T., Jain, A.K., Johannessen, T., Jones, C.D., Kato, E., Keeling, R.F., Goldewijk, K.K., Landschützer, P., Lefèvre, N., Lienert, S., Liu, Z., Lombardozzi, D., Metzl, N., Munro, D.R., Nabel, J.E.M.S., Nakaoka, S., Neill, C., Olsen, A., Ono, T., Patra, P., Peregon, A., Peters, W., Peylin, P., Pfeil, B., Pierrot, D., Poulter, B., Rehder, G., Resplandy, L., Robertson, E., Rocher, M., Rödenbeck, C., Schuster, U., Schwinger, J., Séférian, R., Skjelvan, I., Steinhoff, T., Sutton, A., Tans, P.P., Tian, H., Tilbrook, B., Tubiello, F.N., van der Laan-Luijkx, I.T., van der Werf, G.R., Viovy, N., Walker, A.P., Wiltshire, A.J., Wright, R., Zaehle, S. and Zheng, B. 2018. Global Carbon Budget 2018. *Earth System Science Data*. **10**(4), pp.2141–2194.

Ledley, T.S., Sundquist, E.T., Schwartz, S.E., Hall, D.K., Fellows, J.D. and Killeen, T.L. 1999. Climate change and greenhouse gases. *Eos, Transactions American Geophysical Union*. **80**(39), pp.453–458.

Lehtinen, T., Schlatter, N., Baumgarten, A., Bechini, L., Krüger, J., Grignani, C., Zavattaro, L., Costamagna, C. and Spiegel, H. 2014. Effect of crop residue incorporation on soil organic carbon and greenhouse gas emissions in European agricultural soils. *Soil Use and Management*. **30**(4), pp.524–538.

Lesk, C., Rowhani, P. and Ramankutty, N. 2016. Influence of extreme weather disasters on global crop production. *Nature*. **529**(7584), pp.84–87.

Liebman, M. and Dyck, E. 1993. Crop Rotation and Intercropping Strategies for Weed Management. *Ecological Applications*. **3**(1), pp.92–122.

Lloyd, I.L., Grayson, R.P., Galdos, M.V., Morrison, R. and Chapman, P.J. 2024. Nitrous oxide and methane fluxes from plasma-treated pig slurry applied to winter wheat. *Nutrient Cycling in Agroecosystems*. **129**(3), pp.505–520.

Lovett, G.M., Cole, J.J. and Pace, M.L. 2006. Is Net Ecosystem Production Equal to Ecosystem Carbon Accumulation? *Ecosystems*. **9**(1), pp.152–155.

Lowder, S.K., Sánchez, M.V. and Bertini, R. 2019. *Farms, family farms, farmland distribution and farm labour what do we know today?* Rome: Food and Agriculture Organization of the United Nations.

Lynch, J., Cain, M., Frame, D. and Pierrehumbert, R. 2021. Agriculture's Contribution to Climate Change and Role in Mitigation Is Distinct From Predominantly Fossil CO2-Emitting Sectors. *Frontiers in Sustainable Food Systems*. **4**, p.518039.

Lynch, J., Cain, M., Pierrehumbert, R. and Allen, M. 2020. Demonstrating GWP*: a means of reporting warming-equivalent emissions that captures the contrasting impacts of short- and long-lived climate pollutants. *Environmental Research Letters*. **15**(4), p.044023.

Ma, Y., Liu, D.L., Schwenke, G. and Yang, B. 2019. The global warming potential of straw-return can be reduced by application of straw-decomposing microbial inoculants and biochar in rice-wheat production systems. *Environmental Pollution*. **252**, pp.835–845.

Mackenzie, S.G. 2016. *Modelling the environmental impacts of pig farming systems and the potential of nutritional solutions to mitigate them*.[Online] Newcastle University. Available from: https://theses.ncl.ac.uk/jspui/bitstream/10443/3379/1/Mackenzie%2c%20S.G.%202016.pdf.

MacLeod, M., Moran, D., Eory, V., Rees, R.M., Barnes, A., Topp, C.F.E., Ball, B., Hoad, S., Wall, E., McVittie, A., Pajot, G., Matthews, R., Smith, P. and Moxey, A. 2010. Developing greenhouse gas marginal abatement cost curves for agricultural emissions from crops and soils in the UK. *Agricultural Systems*. **103**(4), pp.198–209.

Malhi, S. and Lemke, R. 2007. Tillage, crop residue and N fertilizer effects on crop yield, nutrient uptake, soil quality and nitrous oxide gas emissions in a second 4-yr rotation cycle. *Soil and Tillage Research*. **96**(1–2), pp.269–283.

Malley, C.S., Borgford-Parnell, N., Haeussling, S., Howard, I.C., Lefèvre, E.N. and Kuylenstierna, J.C.I. 2023. A roadmap to achieve the global methane pledge. *Environmental Research: Climate*. **2**(1), p.011003.

Manzanilla-Pech, C.I.V., L∅vendahl, P., Gordo, D.M., Difford, G.F., Pryce, J.E., Schenkel, F., Wegmann, S., Miglior, F., Chud, T.C., Moate, P.J., Williams, S.R.O., Richardson, C.M., Stothard, P. and Lassen, J. 2021. Breeding for reduced methane emission and feed-efficient Holstein cows: An international response. *Journal of Dairy Science*. **104**(8), pp.8983–9001.

March, M.D., Haskell, M.J., Chagunda, M.G.G., Langford, F.M. and Roberts, D.J. 2014. Current trends in British dairy management regimens. *Journal of Dairy Science*. **97**(12), pp.7985–7994.

Martínez-Fernández, G., Abecia, L., Arco, A., Cantalapiedra-Hijar, G., Martín-García, A.I., Molina-Alcaide, E., Kindermann, M., Duval, S. and Yáñez-Ruiz, D.R. 2014. Effects of ethyl-3-nitrooxy propionate and 3-nitrooxypropanol on ruminal fermentation, microbial abundance, and methane emissions in sheep. *Journal of Dairy Science*. **97**(6), pp.3790–3799.

Martinez-Fernandez, G., Duval, S., Kindermann, M., Schirra, H.J., Denman, S.E. and McSweeney, C.S. 2018. 3-NOP vs. Halogenated Compound: Methane Production, Ruminal Fermentation and Microbial Community Response in Forage Fed Cattle. *Frontiers in Microbiology*. **9**.

McAuliffe, G.A., Chapman, D.V. and Sage, C.L. 2016. A thematic review of life cycle assessment (LCA) applied to pig production. *Environmental Impact Assessment Review*. **56**, pp.12–22.

McAuliffe, G.A., Lynch, J., Cain, M., Buckingham, S., Rees, R.M., Collins, A.L., Allen, M., Pierrehumbert, R., Lee, M.R.F. and Takahashi, T. 2023. Are single global warming potential impact assessments adequate for carbon footprints of agri-food systems? *Environmental Research Letters*. **18**(8), p.084014.

McKay, H. 2011. *Short Rotation Forestry: Review of growth and environmental impacts* [Online]. Surrey, UK: Forest Research. Available from: https://cdn.forestresearch.gov.uk/2011/01/frmg002_short_rotation_forestry.pdf.

McNutt, M. 2013. Climate Change Impacts. *Science*. **341**(6145), pp.435–435.

Menegat, S., Ledo, A. and Tirado, R. 2022. Greenhouse gas emissions from global production and use of nitrogen synthetic fertilisers in agriculture. *Scientific Reports*. **12**(1), p.14490.

Met Office 2019. *UK Climate Projections: Headline Findings* [Online]. [Accessed 27 April 2020]. Available from: https://www.metoffice.gov.uk/binaries/content/assets/metofficegovuk/pdf/research/ukcp/ukcp-headline-findings-v2.pdf.

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., O'Rourke, S., Richer-de-Forges, A.C., Odeh, I., Padarian, J., Paustian, K., Pan, G., Poggio, L., Savin, I., Stolbovoy, V., Stockmann, U., Sulaeman, Y., Tsui, C.-C., Vågen, T.-G., van Wesemael, B. and Winowiecki, L. 2017. Soil carbon 4 per mille. *Geoderma*. **292**, pp.59–86.

Misselbrook, T., Hunt, J., Perazzolo, F. and Provolo, G. 2016. Greenhouse Gas and Ammonia Emissions from Slurry Storage: Impacts of Temperature and Potential Mitigation through Covering (Pig Slurry) or Acidification (Cattle Slurry). *Journal of Environmental Quality*. **45**(5), pp.1520–1530.

Misselbrook, T.H., Cardenas, L.M., Camp, V., Thorman, R.E., Williams, J.R., Rollett, A.J. and Chambers, B.J. 2014. An assessment of nitrification inhibitors to reduce nitrous oxide emissions from UK agriculture. *Environmental Research Letters*. **9**(11), p.115006.

Montes, F., Meinen, R., Dell, C., Rotz, A., Hristov, A.N., Oh, J., Waghorn, G., Gerber, P.J., Henderson, B., Makkar, H.P.S. and Dijkstra, J. 2013. SPECIAL TOPICS — Mitigation of methane and nitrous oxide emissions from animal operations: II. A review of manure management mitigation options1. *Journal of Animal Science*. **91**(11), pp.5070–5094.

Müller, R. 2021. The impact of the rise in atmospheric nitrous oxide on stratospheric ozone. *Ambio*. **50**(1), pp.35–39.

Muñoz, C., Muñoz, I.A., Rodríguez, R., Urrutia, N.L. and Ungerfeld, E.M. 2024. Effect of combining the methanogenesis inhibitor 3-nitrooxypropanol and cottonseeds on methane emissions, feed intake, and milk production of grazing dairy cows. *animal*. **18**(7), p.101203.

Myhre, G., Shindell, D., Bréon, F.M., Collins, W., Fuglestvedt, J., Huang, J., Koch, D., Lamarque, J.F., Lee, D., Mendoza, B. and et al. 2013. Anthropogenic and Natural Radiative Forcing. *In*: *Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change* [Online]. Cambridge: Cambridge University Press, pp.659–740. [Accessed 7 July 2020]. Available from: https://www.ipcc.ch/site/assets/uploads/2018/02/WG1AR5_Chapter08_FINAL.pdf.

NAEI 2020. Emission factors database. Available from: https://naei.beis.gov.uk/data/ef-all.

National Farmers Union 2019. *Achieving net zero: Farming's 2040 goal* [Online]. Warwickshire, UK: National Farmers Union. [Accessed 6 April 2022]. Available from: https://www.nfuonline.com/archive?treeid=138313.

National Infrastructure Commission 2021. *Engineered greenhouse gas removals* [Online]. Available from: https://nic.org.uk/app/uploads/NIC-July-2021-Engineered-Greenhouse-Gas-Removals-UPDATED.pdf.

NFU, N.F.U. 2019. *Achieving Net Zero: Farming's 2040 Goal* [Online]. [Accessed 16 September 2019]. Available from: https://www.nfuonline.com/nfu-online/business/regulation/achieving-net-zero-farmings-2040-goal/.

Notarnicola, B., Sala, S., Anton, A., McLaren, S.J., Saouter, E. and Sonesson, U. 2017. The role of life cycle assessment in supporting sustainable agri-food systems: A review of the challenges. *Journal of Cleaner Production*. **140**, pp.399–409.

Oelkers, E.H. and Cole, D.R. 2008. Carbon Dioxide Sequestration A Solution to a Global Problem. *Elements*. **4**(5), pp.305–310.

Oertel, C., Matschullat, J., Zurba, K., Zimmermann, F. and Erasmi, S. 2016. Greenhouse gas emissions from soils—A review. *Geochemistry*. **76**(3), pp.327–352.

Oldfield, E., Eagle, A.J., Rubin, R.L., Rudek, J., Sanderman, J. and Gordon, D.R. 2021. *Agricultural soil carbon credits: Making sense of protocols for carbon sequestration and net greenhouse gas removals.* [Online]. New York: Environmental Defense Fund. [Accessed 11 January 2025]. Available from: https://www.edf.org/sites/default/files/content/agricultural-soil-carbon-credits-protocol-synthesis.pdf.

Oliver, M.A., Bishop, T.F.A. and Marchant, B.P. (eds.). 2013. *Precision agriculture for sustainability and environmental protection*. Abingdon, Oxon: Routledge.

Ortiz-Bobea, A., Ault, T.R., Carrillo, C.M., Chambers, R.G. and Lobell, D.B. 2021. Anthropogenic climate change has slowed global agricultural productivity growth. *Nature Climate Change*. **11**(4), pp.306–312.

Ottosen, M., Mackenzie, S.G., Filipe, J.A., Misiura, M.M. and Kyriazakis, I. 2021. Changes in the environmental impacts of pig production systems in Great Britain over the last 18 years. *Agricultural Systems*. **189**, p.103063.

Overmeyer, V., Kube, A., Clemens, J., Büscher, W. and Trimborn, M. 2021. One-Time Acidification of Slurry: What Is the Most Effective Acid and Treatment Strategy? *Agronomy*. **11**(7), p.1319.

Overmeyer, V., Trimborn, M., Clemens, J., Hölscher, R. and Büscher, W. 2023. Acidification of slurry to reduce ammonia and methane emissions: Deployment of a retrofittable system in fattening pig barns. *Journal of Environmental Management*. **331**, p.117263.

Parliamentary Commissioner for the Environment 2022. *How much forestry would be needed to offset warming from agricultural methane?* [Online]. New Zealand: New Zealand Parliament. [Accessed 25 January 2025]. Available from: https://pce.parliament.nz/media/03bpa3sn/how-much-forestry-would-be-needed-to-offset-warming-from-agricultural-methane.pdf.

Paustian, K., Babcock, B., Hatfield, J.L., Lal, R., McCarl, B.A., McLaughlin, S., Mosier, A. rvin, Rice, C., Roberton, G.P., Rosenberg, N.J. and Rosenzweig, C. 2001. Agricultural mitigation of greenhouse gases: Science and policy options.

Pedersen, J., Feilberg, A. and Nyord, T. 2022. Effect of storage and field acidification on emissions of NH3, NMVOC, and odour from field applied slurry in winter conditions. *Journal of Environmental Management*. **310**, p.114756.

Pelletier, N., Lammers, P., Stender, D. and Pirog, R. 2010. Life cycle assessment of high- and low-profitability commodity and deep-bedded niche swine production systems in the Upper Midwestern United States. *Agricultural Systems*. **103**(9), pp.599–608.

Peters, G.P., Andrew, R.M., Canadell, J.G., Fuss, S., Jackson, R.B., Korsbakken, J.I., Le Quéré, C. and Nakicenovic, N. 2017. Key indicators to track current progress and future ambition of the Paris Agreement. *Nature Climate Change*. **7**(2), pp.118–122.

Petersen, S.O., Andersen, A.J. and Eriksen, J. 2012. Effects of Cattle Slurry Acidification on Ammonia and Methane Evolution during Storage. *Journal of Environmental Quality*. **41**(1), pp.88–94.

Petersen, S.O., Blanchard, M., Chadwick, D., Del Prado, A., Edouard, N., Mosquera, J. and Sommer, S.G. 2013. Manure management for greenhouse gas mitigation. *Animal*. **7**, pp.266–282.

Petersen, S.O., Højberg, O., Poulsen, M., Schwab, C. and Eriksen, J. 2014. Methanogenic community changes, and emissions of methane and other gases, during storage of acidified and untreated pig slurry. *Journal of Applied Microbiology*. **117**(1), pp.160–172.

Pexas, G., Mackenzie, S.G., Wallace, M. and Kyriazakis, I. 2020. Environmental impacts of housing conditions and manure management in European pig production systems through a life cycle perspective: A case study in Denmark. *Journal of Cleaner Production*. **253**, p.120005.

Poore, J. and Nemecek, T. 2018. Reducing food's environmental impacts through producers and consumers. *Science*.

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**(6), pp.2563–2584.

Powlson, D.S., Whitmore, A.P. and Goulding, K.W. 2011. Soil carbon sequestration to mitigate climate change: a critical re-examination to identify the true and the false. *European Journal of Soil Science*. **62**(1), pp.42–55.

Qiao, C., Liu, L., Hu, S., Compton, J.E., Greaver, T.L. and Li, Q. 2015. How inhibiting nitrification affects nitrogen cycle and reduces environmental impacts of anthropogenic nitrogen input. *Global Change Biology*. **21**(3), pp.1249–1257.

Rapson, T.D. and Dacres, H. 2014. Analytical techniques for measuring nitrous oxide. *TrAC Trends in Analytical Chemistry*. **54**, pp.65–74.

Ravishankara, A.R., Daniel, J.S. and Portmann, R.W. 2009. Nitrous Oxide (N2O): The Dominant Ozone-Depleting Substance Emitted in the 21st Century. *Science*. **326**(5949), pp.123–125.

Reay, D.S. 2020. Land Use and Agriculture: Pitfalls and Precautions on the Road to Net Zero. *Frontiers in Climate*. **2**.

Rees, R.M., Baddeley, J.A., Bhogal, A., Ball, B.C., Chadwick, D.R., Macleod, M., Lilly, A., Pappa, V.A., Thorman, R.E. and Watson, C.A. 2013. Nitrous oxide mitigation in UK agriculture. *Soil Science and Plant Nutrition*. **59**(1), pp.3–15.

Regueiro, I., Coutinho, J. and Fangueiro, D. 2016. Alternatives to sulfuric acid for slurry acidification: impact on slurry composition and ammonia emissions during storage. *Journal of Cleaner Production*. **131**, pp.296–307.

Reisinger, A., Clark, H., Cowie, A.L., Emmet-Booth, J., Gonzalez Fischer, C., Herrero, M., Howden, M. and Leahy, S. 2021. How necessary and feasible are reductions of methane emissions from livestock to support stringent temperature goals? *Philosophical Transactions of the Royal Society A: Mathematical, Physical and Engineering Sciences*. **379**(2210), p.20200452.

Ren, B., Guo, Y., Liu, P., Zhao, B. and Zhang, J. 2021. Effects of Urea-Ammonium Nitrate Solution on Yield, N2O Emission, and Nitrogen Efficiency of Summer Maize Under Integration of Water and Fertilizer. *Frontiers in Plant Science*. **12**.

Richardson, K., Steffen, W., Lucht, W., Bendtsen, J., Cornell, S.E., Donges, J.F., Drüke, M., Fetzer, I., Bala, G., Von Bloh, W., Feulner, G., Fiedler, S., Gerten, D., Gleeson, T., Hofmann, M., Huiskamp, W., Kummu, M., Mohan, C., Nogués-Bravo, D., Petri, S., Porkka, M., Rahmstorf, S., Schaphoff, S., Thonicke, K., Tobian, A., Virkki, V., Wang-Erlandsson, L., Weber, L. and Rockström, J. 2023. Earth beyond six of nine planetary boundaries. *Science Advances*. **9**(37), p.eadh2458.

Ritchie, H. and Roser, M. 2019. *Land Use* [Online]. Our World In Data. [Accessed 7 March 2022]. Available from: https://ourworldindata.org/land-use.

Rockström, J., Edenhofer, O., Gaertner, J. and DeClerck, F. 2020. Planet-proofing the global food system. *Nature Food*. **1**(1), pp.3–5.

Roe, S., Streck, C., Obersteiner, M., Frank, S., Griscom, B., Drouet, L., Fricko, O., Gusti, M., Harris, N., Hasegawa, T., Hausfather, Z., Havlík, P., House, J., Nabuurs, G.-J., Popp, A., Sánchez, M.J.S., Sanderman, J., Smith, P., Stehfest, E. and Lawrence, D. 2019. Contribution of the land sector to a 1.5 °C world. *Nature Climate Change*. **9**(11), pp.817–828.

Roelfsema, M., van Soest, H.L., Harmsen, M., van Vuuren, D.P., Bertram, C., den Elzen, M., Höhne, N., Iacobuta, G., Krey, V., Kriegler, E., Luderer, G., Riahi, K., Ueckerdt, F., Després, J., Drouet, L., Emmerling, J., Frank, S., Fricko, O., Gidden, M., Humpenöder, F., Huppmann, D., Fujimori, S., Fragkiadakis, K., Gi, K., Keramidas, K., Köberle, A.C., Aleluia Reis, L., Rochedo, P., Schaeffer, R., Oshiro, K., Vrontisi, Z., Chen, W., Iyer, G.C., Edmonds, J., Kannavou, M., Jiang, K., Mathur, R., Safonov, G. and Vishwanathan, S.S. 2020. Taking stock of national climate policies to evaluate implementation of the Paris Agreement. *Nature Communications*. **11**(1), p.2096.

Röös, E. and Nylinder, J. 2013. Uncertainties and variations in the carbon footprint of livestock products. *Rapport (Institutionen för energi och teknik, SLU)*. (063).

Roque, B.M., Venegas, M., Kinley, R.D., Nys, R. de, Duarte, T.L., Yang, X. and Kebreab, E. 2021. Red seaweed (Asparagopsis taxiformis) supplementation reduces enteric methane by over 80 percent in beef steers. *PLOS ONE*. **16**(3), p.e0247820.

Rosa, L. and Gabrielli, P. 2023. Achieving net-zero emissions in agriculture: a review. *Environmental Research Letters*. **18**(6), p.063002.

Rowe, S.J., Hickey, S.M., Jonker, A., Hess, M.K., Janssen, P., Johnson, T., Bryson, B., Knowler, K., Pinares-Patiño, C.S., Bain, W., Elmes, S., Young, E., Wing, J., Waller, E., Pickering, N. and McEwan, J.C. 2019. Selection for divergent methane yield in New Zealand sheep - A ten year perspective *In*: *Proceedings of the 23rd Conference of the Association for the Advancement of Animal Breeding and Genetics (AAABG), Armidale, New South Wales, Australia, 27th October-1st November 2019* [Online]. NSW, Australia, pp.306–309. Available from: https://www.researchgate.net/publication/337570664_Selection_for_divergent_methane_yield_in_New_Zealand_sheep_-_A_ten_year_perspective.

Roy, P., Nei, D., Orikasa, T., Xu, Q., Okadome, H., Nakamura, N. and Shiina, T. 2009. A review of life cycle assessment (LCA) on some food products. *Journal of Food Engineering*. **90**(1), pp.1–10.

Rumpel, C., Lehmann, J. and Chabbi, A. 2018. '4 per 1,000' initiative will boost soil carbon for climate and food security. *Nature*. **553**(7686), pp.27–27.

Ruser, R. and Schulz, R. 2015. The effect of nitrification inhibitors on the nitrous oxide (N2O) release from agricultural soils—a review. *Journal of Plant Nutrition and Soil Science*. **178**(2), pp.171–188.

Searchinger, T., Waite, R., Hanson, C., Ranganathan, J., Dumas, P., Matthews, E. and Klirs, C. 2019. *Creating a Sustainable Food Future: A menu of solutions to feed nearly 10 billion people by 2050* [Online]. World Resources Institute. [Accessed 20 July 2020]. Available from: https://agritrop.cirad.fr/593176/1/WRR_Food_Full_Report_0.pdf.

Shin, S.-R., Im, S., Mostafa, A., Lee, M.-K., Yun, Y.-M., Oh, S.-E. and Kim, D.-H. 2019. Effects of pig slurry acidification on methane emissions during storage and subsequent biogas production. *Water Research*. **152**, pp.234–240.

Shine, K.P. 2009. The global warming potential—the need for an interdisciplinary retrial. *Climatic Change*. **96**(4), pp.467–472.

Shukla, P.R., Skea, J., Slade, R., van Diemen, R., Haughey, E., Malley, J., Pathak, M. and Portugal Pereira, J. 2019. *Technical Summary, 2019. In: Climate Change and Land: an IPCC special report on climate change, desertification, land degradation, sustainable land management, food security, and greenhouse gas fluxes in terrestrial ecosystems.* [Online]. IPCC. [Accessed 20 July 2020]. Available from: https://www.ipcc.ch/site/assets/uploads/sites/4/2020/07/03_Technical-Summary-TS_V2.pdf.

Smith, K.A. and Conen, F. 2004. Impacts of land management on fluxes of trace greenhouse gases. *Soil Use and Management*. **20**(2), pp.255–263.

Smith, K.A., Dobbie, K.E., Thorman, R., Watson, C.J., Chadwick, D.R., Yamulki, S. and Ball, B.C. 2012. The effect of N fertilizer forms on nitrous oxide emissions from

UK arable land and grassland. *Nutrient Cycling in Agroecosystems*. **93**(2), pp.127–149.

Smith, P. 2012. Agricultural greenhouse gas mitigation potential globally, in E urope and in the UK: what have we learnt in the last 20 years? *Global Change Biology*. **18**(1), pp.35–43.

Smith, P. 2004. Carbon sequestration in croplands: the potential in Europe and the global context. *European Journal of Agronomy*. **20**(3), pp.229–236.

Smith, P. 2016. Soil carbon sequestration and biochar as negative emission technologies. *Global Change Biology*. **22**(3), pp.1315–1324.

Smith, P., Bhogal, A., Edgington, P., Black, H., Lilly, A., Barraclough, D., Worrall, F., Hillier, J. and Merrington, G. 2010. Consequences of feasible future agricultural land-use change on soil organic carbon stocks and greenhouse gas emissions in Great Britain: Land-use change and GB soil carbon stocks. *Soil Use and Management*. **26**(4), pp.381–398.

Smith, P., Bustamante, M., Ahammad, H., Clark, H., Dong, H., Elsiddig, E., Haberl, H., Harper, R., House, J. and Jafari, M. 2014. Agriculture, Forestry and Other Land Use (AFOLU) *In*: *Climate Change 2014: Mitigation of Climate Change. Contribution of Working Group III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*. Cambridge: Cambridge University Press, pp.811–922.

Smith, P., Goulding, K., Smith, K., Powlson, D., Smith, J., Falloon, P. and Coleman, K. 2001. Enhancing the carbon sink in European agricultural soils: including trace gas fluxes in estimates of carbon mitigation potential. *NUTRIENT CYCLING IN AGROECOSYSTEMS*. **60**(1–3), pp.237–252.

Smith, P., Martino, D., Cai, Z., Gwary, D., Janzen, H., Kumar, P., McCarl, B., Ogle, S., O'Mara, F. and Rice, C. 2007. Policy and technological constraints to implementation of greenhouse gas mitigation options in agriculture. *Agriculture, Ecosystems & Environment*. **118**(1–4), pp.6–28.

Smith, P., Martino, D., Cai, Z., Gwary, D., Janzen, H., Kumar, P., McCarl, B., Ogle, S., O'Mara, F., Rice, C., Scholes, B., Sirotenko, O., Howden, M., McAllister, T., Pan, G., Romanenkov, V., Schneider, U., Towprayoon, S., Wattenbach, M. and Smith, J. 2008. Greenhouse gas mitigation in agriculture. *Philosophical Transactions of the Royal Society B: Biological Sciences*. **363**(1492), pp.789–813.

Smith, P., Reay, D. and Smith, J. 2021. Agricultural methane emissions and the potential formitigation. *Philosophical Transactions of the Royal Society A: Mathematical, Physical and Engineering Sciences*. **379**(2210), p.20200451.

Snyder, C., Davidson, E., Smith, P. and Venterea, R. 2014. Agriculture: sustainable crop and animal production to help mitigate nitrous oxide emissions. *Current Opinion in Environmental Sustainability*. **9–10**, pp.46–54.

Snyder, C.S., Bruulsema, T.W., Jensen, T.L. and Fixen, P.E. 2009. Review of greenhouse gas emissions from crop production systems and fertilizer management effects. *Agriculture, Ecosystems & Environment*. **133**(3–4), pp.247–266.

Soto, I., Barnes, A., Balafoutis, A., Beck, B., Sanchez, B., Vangeyte, J., Fountas, S., Van der Wal, T., Eory, V. and Gómez-Barbero, M. 2019. *The contribution of precision* *agriculture technologies to farm productivity and the mitigation of greenhouse gas emissions in the EU* [Online]. Luxembourg: Publications Office of the European Union. [Accessed 12 January 2023]. Available from: https://data.europa.eu/doi/10.2760/016263.

Soussana, J.-F., Lutfalla, S., Ehrhardt, F., Rosenstock, T., Lamanna, C., Havlík, P., Richards, M., Wollenberg, E. (Lini), Chotte, J.-L., Torquebiau, E., Ciais, P., Smith, P. and Lal, R. 2019. Matching policy and science: Rationale for the '4 per 1000 - soils for food security and climate' initiative. *Soil and Tillage Research*. **188**, pp.3–15.

Spracklen, D.V., Chapman, P.J., Fletcher, T., Lane, J.V., Nilsen, E.B., Perks, M., Schofield, L. and Scott, C.E. 2025. Wolf reintroduction to Scotland could support substantial native woodland expansion and associated carbon sequestration. *Ecological Solutions and Evidence*. **6**(1), p.e70016.

Staddon, P., Urquhart, J., Mills, J., Goodenough, A., Powell, J., Vigani, M., Simmonds, P. and Rowe, E. 2021. *Encouraging woodland creation, regeneration and tree planting on agricultural land: A literature review*. UK: Natural England.

Stavi, I. and Lal, R. 2013. Agriculture and greenhouse gases, a common tragedy. A review. *Agronomy for Sustainable Development*. **33**(2), pp.275–289.

Stephen, K.L. 2012. *Life Cycle Assessment of UK Pig Production Systems: The impact of dietary protein source*.[Online] The University of Edinburgh. Available from: https://era.ed.ac.uk/bitstream/handle/1842/6222/Stephen2012.pdf?sequence=2&isAllowed=y.

Storr, T., Simmons, R.W. and Hannam, J.A. 2019. A UK survey of the use and management of cover crops. *Annals of Applied Biology*., aab.12488.

Styles, D., Gibbons, J., Williams, A.P., Dauber, J., Stichnothe, H., Urban, B., Chadwick, D.R. and Jones, D.L. 2015. Consequential life cycle assessment of biogas, biofuel and biomass energy options within an arable crop rotation. *GLOBAL CHANGE BIOLOGY BIOENERGY*. **7**(6), pp.1305–1320.

Swales, D. 2022. Will Defra's new Sustainable Farming Incentive ease farmers' transition away from subsidies? *Livestock*. **27**(4), pp.172–173.

Sykes, A.J., Macleod, M., Eory, V., Rees, R.M., Payen, F., Myrgiotis, V., Williams, M., Sohi, S., Hillier, J., Moran, D., Manning, D.A.C., Goglio, P., Seghetta, M., Williams, A., Harris, J., Dondini, M., Walton, J., House, J. and Smith, P. 2020. Characterising the biophysical, economic and social impacts of soil carbon sequestration as a greenhouse gas removal technology. *Global Change Biology*. **26**(3), pp.1085–1108.

Sylvester-Bradley, R. and Kindred, D.R. 2009. Analysing nitrogen responses of cereals to prioritize routes to the improvement of nitrogen use efficiency. *Journal of Experimental Botany*. **60**(7), pp.1939–1951.

Sylvester-Bradley, R., Thorman, R., Kindred, D.R., Wynn, S., Smith, K., Rees, R., Topp, C.F.E., Pappa, V., Mortimer, N., Misselbrook, T.H., Gilhespy, S., Cardenas, L.M., Chauhan, M., Bennett, G., Malkin, S. and Munro, D. 2015. *Minimising nitrous oxide intensities of arable crop products (MIN-NO)* [Online]. AHDB. [Accessed 30 August 2024]. Available from: https://projectblue.blob.core.windows.net/media/Default/Research%20Papers/Cereals%20and%20Oilseed/pr548.pdf.

Thornton, P., Nelson, G., Mayberry, D. and Herrero, M. 2021. Increases in extreme heat stress in domesticated livestock species during the twenty-first century. *Global Change Biology*. **27**(22), pp.5762–5772.

Thorpe, A. 2009. Enteric fermentation and ruminant eructation: the role (and control?) of methane in the climate change debate. *Climatic Change*. **93**(3), pp.407–431.

Tilman, D., Fargione, J., Wolff, B., D'Antonio, C., Dobson, A., Howarth, R., Schindler, D., Schlesinger, W.H., Simberloff, D. and Swackhamer, D. 2001. Forecasting Agriculturally Driven Global Environmental Change. *Science*. **292**(5515), pp.281–284.

Tribouillois, H., Cohan, J.-P. and Justes, E. 2016. Cover crop mixtures including legume produce ecosystem services of nitrate capture and green manuring: assessment combining experimentation and modelling. *Plant and Soil*. **401**(1), pp.347–364.

Tyllianakis, E., Martin-Ortega, J., Ziv, G., Chapman, P.J., Holden, J., Cardwell, M. and Fyfe, D. 2023. A window into land managers' preferences for new forms of agri-environmental schemes: Evidence from a post-Brexit analysis. *Land Use Policy*. **129**, p.106627.

UK Government 2021. *Net Zero Strategy: Build Back Greener* [Online]. United Kingdom: Department for Business, Energy & Industrial Strategy. Available from: https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/1033990/net-zero-strategy-beis.pdf.

UK Government 2019. UK becomes first major economy to pass net zero emissions law. *GOV.UK*. [Online]. [Accessed 6 April 2022]. Available from: https://www.gov.uk/government/news/uk-becomes-first-major-economy-to-pass-net-zero-emissions-law.

United Nations Environment Programme 2021. Homepage | Global Methane Pledge. [Accessed 24 August 2024]. Available from: https://www.globalmethanepledge.org/#about.

United Nations Framework Convention on Climate Change 1997. Kyoto Protocol to the United Nations Framework Convention on Climate Change. [Accessed 23 August 2024]. Available from: https://digitallibrary.un.org/record/250111/files/FCCC_CP_1997_L.7_Add.1-EN.pdf?ln=en.

United Nations Framework Convention on Climate Change 2015. The Paris Agreement | UNFCCC. [Accessed 23 August 2024]. Available from: https://unfccc.int/process-and-meetings/the-paris-agreement.

Vargas, J., Ungerfeld, E., Muñoz, C. and DiLorenzo, N. 2022. Feeding Strategies to Mitigate Enteric Methane Emission from Ruminants in Grassland Systems. *Animals*. **12**(9), p.1132.

Vechi, N.T., Jensen, N.S. and Scheutz, C. 2022. Methane emissions from five Danish pig farms: Mitigation strategies and inventory estimated emissions. *Journal of Environmental Management*. **317**, p.115319.

Vermeulen, S.J., Campbell, B.M. and Ingram, J.S.I. 2012. Climate Change and Food Systems. *Annual Review of Environment and Resources*. **37**(1), pp.195–222.

Verschuuren, J. 2016. The Paris Agreement on Climate Change: Agriculture and Food Security. *European Journal of Risk Regulation*. **7**(1), pp.54–57.

Vinci, G., Ruggieri, R., Ruggeri, M. and Zaki, M.G. 2022. Application of Life Cycle Assessment (LCA) to cereal production: an overview. *IOP Conference Series: Earth and Environmental Science*. **1077**(1), p.012004.

Voosen, P. 2021. Why is atmospheric methane surging? (Hint: It's not fracking). [Accessed 5 January 2023]. Available from: https://www.science.org/content/article/why-atmospheric-methane-surging-hint-its-not-fracking.

Vyas, D., McGinn, S.M., Duval, S.M., Kindermann, M. and Beauchemin, K.A. 2016. Effects of sustained reduction of enteric methane emissions with dietary supplementation of 3-nitrooxypropanol on growth performance of growing and finishing beef cattle1. *Journal of Animal Science*. **94**(5), pp.2024–2034.

Waste & Resources Action Programme 2024. *Scope 3 GHG Measurement & Reporting Protocols: Sector Guidance for Food & Drink Businesses* [Online]. WRAP. [Accessed 8 February 2025]. Available from: https://www.wrap.ngo/sites/default/files/2024-06/scope3-GHG-measurement-reporting-protocols-v2.pdf.

Webb, N.P., Marshall, N.A., Stringer, L.C., Reed, M.S., Chappell, A. and Herrick, J.E. 2017. Land degradation and climate change: building climate resilience in agriculture. *Frontiers in Ecology and the Environment*. **15**(8), pp.450–459.

Weiss, F. and Leip, A. 2012. Greenhouse gas emissions from the EU livestock sector: A life cycle assessment carried out with the CAPRI model. *Agriculture, Ecosystems & Environment*. **149**, pp.124–134.

Wernet, G., Bauer, C., Steubing, B., Reinhard, J., Moreno-Ruiz, E. and Weidema, B. 2016. The ecoinvent database version 3 (part I): overview and methodology. *The International Journal of Life Cycle Assessment*. **21**(9), pp.1218–1230.

West, V. 2011. Soil Carbon and the Woodland Carbon Code. [Accessed 15 August 2024]. Available from: https://woodlandcarboncode.org.uk/images/PDFs/SoilCarbonandtheWoodlandCarbonCode_FINAL_14July2011.pdf.

Westaway, S., Grange, I., Smith, J. and Smith, L.G. 2023. Meeting tree planting targets on the UK's path to net-zero: A review of lessons learnt from 100 years of land use policies. *Land Use Policy*. **125**, p.106502.

Williams, A.G., Audsley, E. and Sandars, D.L. 2006. *Determining the environmental burdens and resource use in the production of agricultural and horticultural commodities* [Online]. Bedford: Cranfield University. [Accessed 11 May 2021]. Available from: https://www.researchgate.net/profile/Eric_Audsley/publication/265084052_Determining_the_environmental_burdens_and_resource_use_in_the_production_of_agricultural_and_horticultural_commodities_Defra_project_report_IS0205/links/544e841c0cf29473161be279.pdf.

Winkler, K., Fuchs, R., Rounsevell, M. and Herold, M. 2021. Global land use changes are four times greater than previously estimated. *Nature Communications*. **12**(1), p.2501.

Woodland Carbon Code 2022. Woodland Carbon Code: Requirements for voluntary carbon sequestration projects. [Accessed 9 March 2023]. Available from: https://www.woodlandcarboncode.org.uk/images/PDFs/Woodland_Carbon_Code_V2.2_April_2022.pdf.

World Meteorological Organization (ed.). 1979. *Proceedings of the World Climate Conference: a conference of experts on climate and mankind, Geneva, 12-23 February 1979*. Geneva: Secretariat of the World Meteorological Organization.

World Resources Institute and World Business Council for Sustainable Development 2014. *GHG Protocol Agricultural Guidance* [Online]. Washington, DC; Geneva, Switzerland: World Resources Institute ; World Business Council for Sustainable Development. [Accessed 12 December 2020]. Available from: https://ghgprotocol.org/sites/default/files/standards/GHG%20Protocol%20Agricultural%20Guidance%20%28April%2026%29_0.pdf.

World Resources Institute and World Business Council for Sustainable Development 2011. *Greenhouse gas protocol: product life cycle accounting and reporting standard.* [Online]. Washington, DC; Geneva, Switzerland: World Resources Institute ; World Business Council for Sustainable Development. Available from: https://ghgprotocol.org/sites/default/files/standards/Product-Life-Cycle-Accounting-Reporting-Standard-EReader_041613_0.pdf.

Wu, Y., Zhao, F., Liu, S., Wang, L., Qiu, L., Alexandrov, G. and Jothiprakash, V. 2018. Bioenergy production and environmental impacts. *Geoscience Letters*. **5**(1), p.14.

Yu, G., Beauchemin, K.A. and Dong, R. 2021. A Review of 3-Nitrooxypropanol for Enteric Methane Mitigation from Ruminant Livestock. *Animals*. **11**(12), p.3540.

Zhao, C., Liu, B., Piao, S., Wang, X., Lobell, D.B., Huang, Y., Huang, M., Yao, Y., Bassu, S., Ciais, P., Durand, J.-L., Elliott, J., Ewert, F., Janssens, I.A., Li, T., Lin, E., Liu, Q., Martre, P., Müller, C., Peng, S., Peñuelas, J., Ruane, A.C., Wallach, D., Wang, T., Wu, D., Liu, Z., Zhu, Y., Zhu, Z. and Asseng, S. 2017. Temperature increase reduces global yields of major crops in four independent estimates. *Proceedings of the National Academy of Sciences*. **114**(35), pp.9326–9331.

Appendix A

Estimating GHG emissions and modelling mitigation options for an integrated pig-arable farm

Table A.1 University of Leeds farm arable data for yields (t ha^{-1}), area (ha) and production (t) per crop per year.

Year	Crop	Yield	Area	Production
		t ha^{-1}	ha	t
2015	WW	10.7	128	1,366
	WB	9.9	29	289
	OSR	4.7	44	206
	GRASS	6.8	28	192
2016	WW	8.3	98	809
	WB	7.3	50	367
	OSR	3.0	29	88
	GRASS	6.3	36	230
	SB	6.8	19	132
2017	WW	10	75	748
	WB	8.3	57	471
	OSR	4.2	36	150
	GRASS	4.0	39	150
2018	WW	9.7	112	1,079
	WB	8.5	16	139
	OSR	4.0	60	237
	GRASS	5.7	33	190

Table A.2 Total GHG emissions (kg CO_2e) associated with arable production on the University of Leeds farm between 2015 and 2018. N_2O = nitrous oxide, CO_2 = carbon dioxide. WW = winter wheat, WB = winter barley, SB = spring barley, OSR = oilseed rape, GRASS = grass silage.

Year	Crop	Energy	Soil emissions									Embedded emissions			
		Diesel consumption	Direct N_2O	Direct N_2O - synthetic	Direct N_2O - organic	Direct N_2O - residues	Indirect N_2O	Indirect N_2O - synthetic	Indirect N_2O - organic	Indirect N_2O - residues	CO_2 urea emissions	Fertiliser manufacture	Pesticide manufacture	Seed	Tractor
2015	WW	22,189	98,423	67,796	4,771	25,857	35,079	27,400	3,025	4,654	0	112,964	4,005	18,190	7,142
	WB	6,192	27,973	12,051	8,432	7,490	12,153	4,890	5,915	1,348	0	27,460	4,453	1,047	7,142
	OSR	9,139	43,990	20,886	9,858	13,246	25,000	8,456	14,160	2,384	0	32,974	1,379	160	7,142
	GRASS	3,461	13,377	11,715	1,661	0	5,726	4,673	1,053	0	0	18,776	642	0	7,142
2016	WW	17,405	72,615	44,928	14,589	13,098	30,664	22,998	5,308	2,358	11	74,850	2,230	10,990	5,714
	WB	9,550	34,463	15,312	9,635	9,516	15,754	7,932	6,110	1,713	4	26,774	677	4,750	5,714
	OSR	5,331	20,255	11,254	3,322	5,680	11,883	6,089	4,771	1,022	4	15,843	679	1,419	5,714
	GRASS	1,685	11,974	4,285	7,689	0	9,065	4,190	4,876	0	4	4,693	0	0	5,714
	SB	2,881	10,135	6,739	0	3,396	4,280	3,668	0	611	2	11,264	79	2,115	5,714
2017	WW	14,880	66,210	34,925	17,700	13,584	32,124	18,455	11,224	2,445	10	55,062	1,626	8,494	7,142
	WB	6,153	40,305	28,097	0	12,207	16,837	14,640	0	2,197	8	55,314	754	3,805	7,142
	OSR	3,452	29,902	16,053	4,334	9,515	16,894	8,956	6,225	1,713	6	26,140	1,350	127	7,142
	GRASS	1,446	18,586	14,834	3,752	0	10,899	8,520	2,379	0	5	19,318	0	0	7,142
2018	WW	10,168	56,009	29,133	14,534	12,342	27,856	16,419	9,216	2,222	0	84,576	1,313	9,507	5,571
	WB	3,080	8,606	2,405	3,386	2,815	4,296	1,430	2,360	507	0	7,826	288	1,438	7,142
	OSR	6,296	44,621	25,504	6,132	12,985	24,561	14,410	7,813	2,337	0	48,319	2,305	80	7,142
	GRASS	421	1,335	360	975	0	1,637	1,018	618	0	0	3,728	0	0	7,142

Table A.3 GHG emission intensity (kg CO_2e t^{-1}) associated with arable production on the University of Leeds farm between 2015 and 2018. N_2O = nitrous oxide, CO_2 = carbon dioxide. WW = winter wheat, WB = winter barley, SB = spring barley, OSR = oilseed rape, GRASS = grass silage.

Year	Crop	Energy	Soil emissions									Embedded emissions			
		Diesel consumption	Direct N_2O	Direct N_2O - synthetic	Direct N_2O - organic	Direct N_2O - residues	Indirect N_2O	Indirect N_2O - synthetic	Indirect N_2O - organic	Indirect N_2O - residues	CO_2 urea emissions	Fertiliser manufacture	Pesticide manufacture	Seed	Tractor
2015	WW	16	72	50	3	19	26	20	2	3	0	83	3	13	5
	WB	21	96	42	29	26	42	17	20	5	0	95	15	4	25
	OSR	44	213	101	48	64	121	41	69	12	0	160	7	1	35
	GRASS	18	69	60	9	0	30	24	5	0	0	97	3	0	37
2016	WW	22	90	56	18	16	38	28	7	3	0	93	3	14	7
	WB	26	94	42	26	26	43	22	17	5	0	73	2	13	16
	OSR	65	246	137	40	69	144	74	58	12	0	192	8	17	69
	GRASS	7	52	19	33	0	39	18	21	0	0	20	0	0	25
	SB	22	77	51	0	26	33	28	0	5	0	86	1	16	43
2017	WW	20	89	47	24	18	43	25	15	3	0	74	2	11	10
	WB	13	85	60	0	26	36	31	0	5	0	117	2	8	15
	OSR	23	201	108	29	64	114	60	42	12	0	176	9	1	48
	GRASS	9	119	95	24	0	70	55	15	0	0	124	0	0	46
2018	WW	9	52	27	13	11	26	15	9	2	0	97	2	11	6
	WB	22	62	17	24	20	31	10	17	4	0	72	3	13	66
	OSR	27	189	108	26	55	104	61	33	10	0	239	11	0	35
	GRASS	2	7	2	5	0	9	5	3	0	0	37	0	0	71

Table A.5 GHG emission per hectare (kg CO_2e ha^{-1}) associated with arable production on the University of Leeds farm between 2015 and 2018. N_2O = nitrous oxide, CO_2 = carbon dioxide. WW = winter wheat, WB = winter barley, SB = spring barley, OSR = oilseed rape, GRASS = grass silage.

Year	Crop	Energy	Soil emissions									Embedded emissions			
		Diesel consumption	Direct N_2O	Direct N_2O - synthetic	Direct N_2O - organic	Direct N_2O - residues	Indirect N_2O	Indirect N_2O - synthetic	Indirect N_2O - organic	Indirect N_2O - residues	CO_2 urea emissions	Fertiliser manufacture	Pesticide manufacture	Seed	Tractor
2015	WW	174	771	531	37	203	275	215	24	36	0	885	31	142	56
	WB	212	958	413	289	256	416	167	202	46	0	940	152	36	245
	OSR	209	1,006	478	225	303	572	193	324	55	0	754	32	4	163
	GRASS	123	474	415	59	0	203	166	37	0	0	665	23	0	253
2016	WW	179	745	461	150	134	314	236	54	24	0	768	23	113	59
	WB	190	686	305	192	189	313	158	122	34	0	533	13	94	114
	OSR	182	693	385	114	194	407	208	163	35	0	542	23	49	196
	GRASS	46	328	118	211	0	249	115	134	0	0	129	0	0	157
	SB	148	522	347	0	175	220	189	0	31	0	580	4	109	294
2017	WW	199	885	467	237	182	430	247	150	33	0	736	22	114	95
	WB	108	710	495	0	215	297	258	0	39	0	974	13	67	126
	OSR	97	839	450	122	267	474	251	175	48	0	733	38	4	200
	GRASS	37	474	379	96	0	278	217	61	0	0	493	0	0	182
2018	WW	91	502	261	130	111	250	147	83	20	0	758	12	85	50
	WB	189	528	148	208	173	264	88	145	31	0	480	18	88	438
	OSR	106	748	428	103	218	412	242	131	39	0	810	39	1	120
	GRASS	13	40	11	29	0	49	31	19	0	0	112	0	0	215

Table A.6 Average (2016 – 2018) total GHG emissions (kg CO_2e) and GHG intensity (kg CO_2e kg $liveweight^{-1}$) for the pig unit on the UoL farm, broken down by the four main emission sources.

Herd/class	Average population size (number of head)	Liveweight (kg)	Total GHG emissions (kg CO_2e)				GHG intensity (kg CO_2e kg liveweight^{-1})			
			Enteric fermentation	Manure management	Feed	Bedding	Enteric fermentation	Manure management	Feed	Bedding
Indoor herd										
In pig sows and gilts	76	190	3,120	24,953	25,204	635	0.22	1.73	1.75	0.04
Suckling sows	27	190	1,096	7,905	1,631	223	0.21	1.54	0.32	0.04
Piglets	284	8.5	11,594	8,095	0	0	4.80	3.35	0.00	0.00
Nursery (up to 40kg)	308	30	12,555	30,940	5,562	673	1.36	3.35	0.60	0.07
Bacon pigs	615	87	25,085	179,272	173,169	2,150	0.47	3.35	3.24	0.04
Maiden gilts	11	150	453	2,862	6,623	92	0.27	1.73	4.01	0.06
Boars	2	100	84	355	1,955	5	0.42	1.77	9.78	0.03
Outdoor herd										
Sows	207	200	8,432	36,980	80,667	1,717	0.20	0.89	1.95	0.04
Piglets	310	8.5	12,648	2,701	0	0	4.80	1.02	0.00	0.00
Maiden gilts	31	150	1,265	4,766	18,482	258	0.27	1.02	3.97	0.06
Boars	3	100	126	277	2,933	8	0.42	0.92	9.78	0.03

Appendix B

Strategies for arable agriculture and woodland creation in England & Wales to contribute to net zero emissions

Table B.1 GHG reduction (%) compared to the baseline Soil emissions calculated for 2018 – 2020 from the Medium GHG abatement potential of 2.06 t CO_2e ha^{-1} yr^{-1} for legumes/non-legumes/mixed or 0.98 t CO_2e ha^{-1} yr^{-1} for agronomy – cover crops. Scenario analysis of a 10-100% uptake of two practices across England and Wales shown with conditional formatting ranging from red (lower % reduction) to green (higher reduction potential). The first practice is introducing a legume, non-legume and mixed legume/non-legume plants into arable rotations (grey highlight) and the second practice is introducing cover crops (including legumes) into arable rotations (white shading).

Region	Practice	Medium									
		10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
NE	Legume/non-legume/mixed	-10.4%	-20.8%	-31.3%	-41.7%	-52.1%	-62.5%	-72.9%	-83.4%	-93.8%	-104.2%
NE	Agronomy - Cover crops	-5.0%	-9.9%	-14.9%	-19.8%	-24.8%	-29.7%	-34.7%	-39.7%	-44.6%	-49.6%
NW	Legume/non-legume/mixed	-3.5%	-7.0%	-10.5%	-13.9%	-17.4%	-20.9%	-24.4%	-27.9%	-31.4%	-34.8%
NW	Agronomy - Cover crops	-1.7%	-3.3%	-5.0%	-6.6%	-8.3%	-9.9%	-11.6%	-13.3%	-14.9%	-16.6%
Y	Legume/non-legume/mixed	-13.6%	-27.2%	-40.7%	-54.3%	-67.9%	-81.5%	-95.1%	-108.6%	-122.2%	-135.8%
Y	Agronomy - Cover crops	-6.5%	-12.9%	-19.4%	-25.8%	-32.3%	-38.8%	-45.2%	-51.7%	-58.1%	-64.6%
EM	Legume/non-legume/mixed	-39.3%	-78.6%	-117.9%	-157.2%	-196.6%	-235.9%	-275.2%	-314.5%	-353.8%	-393.1%
EM	Agronomy - Cover crops	-18.7%	-37.4%	-56.1%	-74.8%	-93.5%	-112.2%	-130.9%	-149.6%	-168.3%	-187.0%
WM	Legume/non-legume/mixed	-13.1%	-26.1%	-39.2%	-52.2%	-65.3%	-78.3%	-91.4%	-104.5%	-117.5%	-130.6%
WM	Agronomy - Cover crops	-6.2%	-12.4%	-18.6%	-24.8%	-31.1%	-37.3%	-43.5%	-49.7%	-55.9%	-62.1%
E	Legume/non-legume/mixed	-21.9%	-43.9%	-65.8%	-87.8%	-109.7%	-131.7%	-153.6%	-175.5%	-197.5%	-219.4%
E	Agronomy - Cover crops	-10.4%	-20.9%	-31.3%	-41.8%	-52.2%	-62.6%	-73.1%	-83.5%	-93.9%	-104.4%
SE	Legume/non-legume/mixed	-39.7%	-79.3%	-119.0%	-158.6%	-198.3%	-237.9%	-277.6%	-317.3%	-356.9%	-396.6%
SE	Agronomy - Cover crops	-18.9%	-37.7%	-56.6%	-75.5%	-94.3%	-113.2%	-132.1%	-150.9%	-169.8%	-188.7%
SW	Legume/non-legume/mixed	-11.1%	-22.3%	-33.4%	-44.6%	-55.7%	-66.9%	-78.0%	-89.2%	-100.3%	-111.5%
SW	Agronomy - Cover crops	-5.3%	-10.6%	-15.9%	-21.2%	-26.5%	-31.8%	-37.1%	-42.4%	-47.7%	-53.0%
Wales	Legume/non-legume/mixed	-3.4%	-6.7%	-10.1%	-13.5%	-16.9%	-20.2%	-23.6%	-27.0%	-30.4%	-33.7%
Wales	Agronomy - Cover crops	-1.6%	-3.2%	-4.8%	-6.4%	-8.0%	-9.6%	-11.2%	-12.8%	-14.4%	-16.0%

Table B.2 GHG reduction (%) compared to the baseline Soil emissions calculated for 2018 – 2020 from the Low GHG abatement potential of -0.04 t CO_2e ha^{-1} yr^{-1} for legumes/non-legumes/mixed or 0.51 t CO_2e ha^{-1} yr^{-1} for agronomy – cover crops. Scenario analysis of a 10-100% uptake of two practices across England and Wales shown with conditional formatting ranging from red (lower % reduction) to green (higher reduction potential). The first practice is introducing a legume, non-legume and mixed legume/non-legume plants into arable rotations (grey highlight) and the second practice is introducing cover crops (including legumes) into arable rotations (white shading).

Region	Practice	Low									
		10%	**20%**	**30%**	**40%**	**50%**	**60%**	**70%**	**80%**	**90%**	**100%**
NE	Legume/non-legume/mixed	0.2%	0.4%	0.6%	0.8%	1.0%	1.2%	1.4%	1.6%	1.8%	2.0%
NE	Agronomy - Cover crops	-2.6%	-5.2%	-7.7%	-10.3%	-12.9%	-15.5%	-18.1%	-20.6%	-23.2%	-25.8%
NW	Legume/non-legume/mixed	0.1%	0.1%	0.2%	0.3%	0.3%	0.4%	0.5%	0.5%	0.6%	0.7%
NW	Agronomy - Cover crops	-0.9%	-1.7%	-2.6%	-3.5%	-4.3%	-5.2%	-6.0%	-6.9%	-7.8%	-8.6%
Y	Legume/non-legume/mixed	0.3%	0.5%	0.8%	1.1%	1.3%	1.6%	1.8%	2.1%	2.4%	2.6%
Y	Agronomy - Cover crops	-3.4%	-6.7%	-10.1%	-13.4%	-16.8%	-20.2%	-23.5%	-26.9%	-30.3%	-33.6%
EM	Legume/non-legume/mixed	0.8%	1.5%	2.3%	3.1%	3.8%	4.6%	5.3%	6.1%	6.9%	7.6%
EM	Agronomy - Cover crops	-9.7%	-19.5%	-29.2%	-38.9%	-48.7%	-58.4%	-68.1%	-77.9%	-87.6%	-97.3%
WM	Legume/non-legume/mixed	0.3%	0.5%	0.8%	1.0%	1.3%	1.5%	1.8%	2.0%	2.3%	2.5%
WM	Agronomy - Cover crops	-3.2%	-6.5%	-9.7%	-12.9%	-16.2%	-19.4%	-22.6%	-25.9%	-29.1%	-32.3%
E	Legume/non-legume/mixed	0.4%	0.9%	1.3%	1.7%	2.1%	2.6%	3.0%	3.4%	3.8%	4.3%
E	Agronomy - Cover crops	-5.4%	-10.9%	-16.3%	-21.7%	-27.2%	-32.6%	-38.0%	-43.5%	-48.9%	-54.3%
SE	Legume/non-legume/mixed	0.8%	1.5%	2.3%	3.1%	3.9%	4.6%	5.4%	6.2%	6.9%	7.7%
SE	Agronomy - Cover crops	-9.8%	-19.6%	-29.5%	-39.3%	-49.1%	-58.9%	-68.7%	-78.5%	-88.4%	-98.2%
SW	Legume/non-legume/mixed	0.2%	0.4%	0.6%	0.9%	1.1%	1.3%	1.5%	1.7%	1.9%	2.2%
SW	Agronomy - Cover crops	-2.8%	-5.5%	-8.3%	-11.0%	-13.8%	-16.6%	-19.3%	-22.1%	-24.8%	-27.6%
Wales	Legume/non-legume/mixed	0.1%	0.1%	0.2%	0.3%	0.3%	0.4%	0.5%	0.5%	0.6%	0.7%
Wales	Agronomy - Cover crops	-0.8%	-1.7%	-2.5%	-3.3%	-4.2%	-5.0%	-5.8%	-6.7%	-7.5%	-8.4%

Table B.3 GHG reduction (%) compared to the baseline Soil emissions calculated for 2018 – 2020 from the High GHG abatement potential of 4.16 t CO_2e ha^{-1} yr^{-1} for legumes/non-legumes/mixed or 1.45 t CO_2e ha^{-1} yr^{-1} for agronomy – cover crops. Scenario analysis of a 10-100% uptake of two practices across England and Wales shown with conditional formatting ranging from red (lower % reduction) to green (higher reduction potential). The first practice is introducing a legume, non-legume and mixed legume/non-legume plants into arable rotations (grey highlight) and the second practice is introducing cover crops (including legumes) into arable rotations (white shading).

Region	Practice	High									
		10%	**20%**	**30%**	**40%**	**50%**	**60%**	**70%**	**80%**	**90%**	**100%**
NE	Legume/non-legume/mixed	-21.0%	-42.1%	-63.1%	-84.2%	-105.2%	-126.3%	-147.3%	-168.3%	-189.4%	-210.4%
NE	Agronomy - Cover crops	-7.3%	-14.7%	-22.0%	-29.3%	-36.7%	-44.0%	-51.3%	-58.7%	-66.0%	-73.3%
NW	Legume/non-legume/mixed	-7.0%	-14.1%	-21.1%	-28.1%	-35.2%	-42.2%	-49.3%	-56.3%	-63.3%	-70.4%
NW	Agronomy - Cover crops	-2.5%	-4.9%	-7.4%	-9.8%	-12.3%	-14.7%	-17.2%	-19.6%	-22.1%	-24.5%
Y	Legume/non-legume/mixed	-27.4%	-54.8%	-82.3%	-109.7%	-137.1%	-164.5%	-192.0%	-219.4%	-246.8%	-274.2%
Y	Agronomy - Cover crops	-9.6%	-19.1%	-28.7%	-38.2%	-47.8%	-57.4%	-66.9%	-76.5%	-86.0%	-95.6%
EM	Legume/non-legume/mixed	-79.4%	-158.8%	-238.2%	-317.5%	-396.9%	-476.3%	-555.7%	-635.1%	-714.5%	-793.8%
EM	Agronomy - Cover crops	-27.7%	-55.3%	-83.0%	-110.7%	-138.4%	-166.0%	-193.7%	-221.4%	-249.0%	-276.7%
WM	Legume/non-legume/mixed	-26.4%	-52.7%	-79.1%	-105.5%	-131.8%	-158.2%	-184.6%	-210.9%	-237.3%	-263.7%
WM	Agronomy - Cover crops	-9.2%	-18.4%	-27.6%	-36.8%	-46.0%	-55.1%	-64.3%	-73.5%	-82.7%	-91.9%
E	Legume/non-legume/mixed	-44.3%	-88.6%	-132.9%	-177.2%	-221.5%	-265.9%	-310.2%	-354.5%	-398.8%	-443.1%
E	Agronomy - Cover crops	-15.4%	-30.9%	-46.3%	-61.8%	-77.2%	-92.7%	-108.1%	-123.6%	-139.0%	-154.4%
SE	Legume/non-legume/mixed	-80.1%	-160.2%	-240.3%	-320.3%	-400.4%	-480.5%	-560.6%	-640.7%	-720.8%	-800.9%
SE	Agronomy - Cover crops	-27.9%	-55.8%	-83.7%	-111.7%	-139.6%	-167.5%	-195.4%	-223.3%	-251.2%	-279.1%
SW	Legume/non-legume/mixed	-22.5%	-45.0%	-67.5%	-90.0%	-112.5%	-135.1%	-157.6%	-180.1%	-202.6%	-225.1%
SW	Agronomy - Cover crops	-7.8%	-15.7%	-23.5%	-31.4%	-39.2%	-47.1%	-54.9%	-62.8%	-70.6%	-78.5%
Wales	Legume/non-legume/mixed	-6.8%	-13.6%	-20.4%	-27.3%	-34.1%	-40.9%	-47.7%	-54.5%	-61.3%	-68.1%
Wales	Agronomy - Cover crops	-2.4%	-4.7%	-7.1%	-9.5%	-11.9%	-14.2%	-16.6%	-19.0%	-21.4%	-23.7%

Table B.4 GHG reduction (%) compared to the baseline Soil emissions calculated for 2018 – 2020 from the GHG abatement potential of 0.5 t CO_2e ha^{-1} yr^{-1} for reduced nitrogen (N) fertiliser or 0.4 t CO_2e ha^{-1} yr^{-1} for avoiding excess N fertiliser. Scenario analysis of a 10-100% uptake of two practices across England and Wales shown with conditional formatting ranging from red (lower % reduction) to green (higher reduction potential). The first practice is reducing the rate of fertiliser application (grey highlight) and the second practice is reducing application of N in excess amounts (white shading).

Region	Practice	Uptake									
		10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
NE	Reduce N fertiliser	-4.9%	-9.7%	-14.6%	-19.5%	-24.4%	-29.2%	-34.1%	-39.0%	-43.9%	-48.7%
NE	Avoid excess N	-3.9%	-7.8%	-11.7%	-15.6%	-19.5%	-23.4%	-27.3%	-31.2%	-35.1%	-39.0%
NW	Reduce N fertiliser	-1.9%	-3.9%	-5.8%	-7.8%	-9.7%	-11.7%	-13.6%	-15.6%	-17.5%	-19.4%
NW	Avoid excess N	-1.6%	-3.1%	-4.7%	-6.2%	-7.8%	-9.3%	-10.9%	-12.4%	-14.0%	-15.6%
Y	Reduce N fertiliser	-4.2%	-8.5%	-12.7%	-16.9%	-21.1%	-25.4%	-29.6%	-33.8%	-38.1%	-42.3%
Y	Avoid excess N	-3.4%	-6.8%	-10.1%	-13.5%	-16.9%	-20.3%	-23.7%	-27.1%	-30.4%	-33.8%
EM	Reduce N fertiliser	-9.8%	-19.5%	-29.3%	-39.1%	-48.9%	-58.6%	-68.4%	-78.2%	-87.9%	-97.7%
EM	Avoid excess N	-7.8%	-15.6%	-23.4%	-31.3%	-39.1%	-46.9%	-54.7%	-62.5%	-70.3%	-78.2%
WM	Reduce N fertiliser	-4.1%	-8.2%	-12.3%	-16.5%	-20.6%	-24.7%	-28.8%	-32.9%	-37.0%	-41.1%
WM	Avoid excess N	-3.3%	-6.6%	-9.9%	-13.2%	-16.5%	-19.8%	-23.0%	-26.3%	-29.6%	-32.9%
E	Reduce N fertiliser	-5.4%	-10.8%	-16.3%	-21.7%	-27.1%	-32.5%	-37.9%	-43.4%	-48.8%	-54.2%
E	Avoid excess N	-4.3%	-8.7%	-13.0%	-17.3%	-21.7%	-26.0%	-30.4%	-34.7%	-39.0%	-43.4%
SE	Reduce N fertiliser	-10.5%	-21.0%	-31.4%	-41.9%	-52.4%	-62.9%	-73.4%	-83.8%	-94.3%	-104.8%
SE	Avoid excess N	-8.4%	-16.8%	-25.2%	-33.5%	-41.9%	-50.3%	-58.7%	-67.1%	-75.5%	-83.8%
SW	Reduce N fertiliser	-3.7%	-7.4%	-11.2%	-14.9%	-18.6%	-22.3%	-26.1%	-29.8%	-33.5%	-37.2%
SW	Avoid excess N	-3.0%	-6.0%	-8.9%	-11.9%	-14.9%	-17.9%	-20.9%	-23.8%	-26.8%	-29.8%
Wales	Reduce N fertiliser	-1.7%	-3.5%	-5.2%	-6.9%	-8.7%	-10.4%	-12.1%	-13.9%	-15.6%	-17.3%
Wales	Avoid excess N	-1.4%	-2.8%	-4.2%	-5.5%	-6.9%	-8.3%	-9.7%	-11.1%	-12.5%	-13.9%

Appendix C

Strategies for livestock agriculture in England and Wales to contribute to net zero emissions

Table C.1 2021 population data for livestock in each GOR of England and Wales. Data from the June Agriculture Survey (Defra, 2024e).

Region	Livestock category	Population in 2021
North East	Dairy	21,154
	Beef	212,713
	Sheep	1,841,882
	Pigs	128,484
	Poultry	2,098,024
North West	Dairy	476,796
	Beef	410,011
	Sheep	2,936,303
	Pigs	100,960
	Poultry	10,901,661
Yorkshire & The Humber	Dairy	130,529
	Beef	376,285
	Sheep	2,045,048
	Pigs	1,613,521
	Poultry	17,880,385
East Midlands	Dairy	115,164
	Beef	314,375
	Sheep	1,212,682
	Pigs	387,821
	Poultry	26,957,963
West Midlands	Dairy	256,100
	Beef	387,634
	Sheep	2,118,949
	Pigs	197,968

	Poultry	26,521,496
East of England	Dairy	24,187
	Beef	148,286
	Sheep	342,695
	Pigs	1,252,974
	Poultry	30,818,830
South East	Dairy	92,871
	Beef	257,256
	Sheep	1,151,506
	Pigs	200,511
	Poultry	8,046,500
South West	Dairy	92,871
	Beef	257,256
	Sheep	2,977,168
	Pigs	355,519
	Poultry	17,859,196
Wales	Dairy	448,629
	Beef	680,220
	Sheep	9,464,299
	Pigs	27,181
	Poultry	10,352,244