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Improving the Carbon Footprinting of Lamb Production

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Improving the Carbon Footprinting of Lamb Production

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Declaration

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

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

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Executive Summary

Lamb production is a key agricultural system within the UK and Wales in particular. While nationally significant, the sector contributes to climate change due to it being responsible for the release of various greenhouse gases (GHGs) namely enteric methane (CH_4), excretal nitrous oxide (N_2O), and carbon dioxide (CO_2). It is therefore important to quantify emissions and consider how they could be reduced in the context of Net Zero climate change targets.

As lamb production generally occurs across a variety of altitudes, including lowland, upland and hill pasture grazing, the animals are subject to varying environmental conditions and dietary options. These factors have been linked to affecting enteric CH_4 and excretal N_2O emissions. However, there is a lack of assessment of emissions changes across the altitudes present in lamb production systems. Additionally, evaluation of how those changes affect carbon footprints (CFs) of lamb in comparison to default methodologies is limited.

This thesis aimed to fill this gap and assess altitudinal emissions changes within lamb production systems. Experimental work indicated that emissions differed across altitudes, with enteric CH_4 output and methane conversion factor (Ym) decreasing with increasing altitude. The same result was determined with excretal N_2O, with the N_2O-N emission factor (EF_{3prp}) being lower in upland pasture than in lowland pasture. A CF tool was then developed that could disaggregate altitudinal variability in emission factors, forage characteristics and farm activity data. This determined that use of default methodologies vs. altitude specific resulted in a change in footprint results. When comparing a non-altitudinally disaggregated footprint with a disaggregated footprint, although both followed Intergovernmental Panel on Climate Change (IPCC) default emission factors (EFs), the CF increased. However, when site-specific EFs and forage characteristics were introduced, the footprint value decreased. This was attributed mostly to the site-specific forage characteristics as they were determined as having the most significant impact on the footprint.

Finally, an assessment of mitigation strategies was performed. It was determined that the GHG reduction effect of different strategies varied dependent on default vs. site-specific modelling, although improving productivity by increasing the number of lambs

per ewe was consistently effective in terms of reducing emissions intensity. Increasing the number of animals grazing upland and hill sites was deemed to increase the footprint when following default methodologies, whereas the site-specific footprint saw a decrease. This highlights that default methodologies may not be full appropriate for assessing extensive grazing. It was also shown that some strategies decreased the CF while increasing the farm annual GHGs, highlighting that the desired outcome should be considered before implementing a strategy in the context of reaching Net Zero. This work also indicated that improvements to productivity on farm could result in reduced requirements for grazing land and therefore the opportunity for woodland creation. Improving the upland grazing to increase grass growth resulted in farm area being spared for woodland creation, resulting in 50% and 74% of annual farm GHGs being offset when following IPCC and site-specific footprints respectively.

Overall, this work highlighted that significant sources of uncertainty are still present within CF of lamb production. Disaggregation of EFs, forage characteristics and animal information produce different results to the use of default inputs. There is therefore progress required both in terms of emissions quantification at the different pastures to improve modelling inputs, but also to improve modelling techniques that better capture variability across systems. The work also highlights that it is possible to pair mitigation options across the different altitudes to spare land on farm and offset emissions via woodland creation. However, further work is required on the practicality of this at a farm level, and to improve robustness of the values used to estimate woodland carbon sequestration in a Welsh context.

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Abbreviations

C = Carbon

CCC = Climate Change Committee

CH_4 = Methane

CHUM = Centre for Hill and Upland Management

CO_2 = Carbon dioxide

CO_2eq = CO_2 equivalent

CP% = Crude protein %

CF = Carbon footprinting

CV% = Coefficient of variation

CW = Carcass weight

DE% = Digestible energy %

DMI = Dry matter intake

EF = Emission factor

EF_{3prp} = Excretal N_2O-N emission factor

FAO = Food and Agriculture Organisation

FU = Functional unit

GEI = Gross energy intake

GF = GreenFeed

GHGs = Greenhouse gases

GWP = Global warming potential

IPCC = Intergovernmental Panel on Climate Change

ISO = International Organization for Standardization

LCA = Life cycle assessment

LW = Liveweight

MACCs = Marginal abatement cost curves

MCF = Manure conversion factor

N = Nitrogen

N_2O = Nitrous oxide

NAD = Non-altitudinally disaggregated

NZ = New Zealand

PPE = Personal Protective Equipment

RC = Respiration chamber

REG = ratio of net energy available for growth in a diet to digestible energy consumed

REM = ratio of net energy available in diet for maintenance to digestible energy consumed

SF_6 = sulphur hexafluoride

UK = United Kingdom

UNFCCC = United Nations Convention on Climate Change

Ym = Methane conversion factor

Chapter 1 Research rationale, thesis aims and objectives, and chapter outlines

1.1 Research rationale

Climate change poses many threats to humanity, and is now being considered as an existential risk (Richards et al., 2021). The scientific evidence that the climate crisis is occurring and has been driven by human behaviour due to release of excess greenhouse gases (GHGs) is now undeniable (Van Linden et al., 2015). A key sector that contributes to climate change is agriculture, and livestock production in particular is receiving increasing attention for its contribution (Grossi et al., 2019). Provision of food security is essential for the wellbeing of global communities, yet agriculture is dependent on a healthy environment to be productive. Evidence is already suggesting that gains in agricultural productivity have slowed directly due to climate change, with a baseline model suggesting that productivity has slowed by 21% since 1961 (Ortiz-Bobea et al., 2021). It is therefore imperative that mitigation options are considered and our understanding of variations of GHGs across livestock systems is developed.

Lamb production is a key agricultural industry within the UK and Wales in particular. A significant proportion of UK sheep are raised in Wales, with recent figures showing approximately 9.5 million sheep currently in Wales alone and 33 million across the UK as a whole (DEFRA, 2022). At a UK level, the sector provides an important contribution to the economy with total UK lamb product exports totalling 70,000 tonnes and being worth £438 million in 2021 (HMRC, 2022). In Wales, products derived from sheep accounted for 17.1% of Welsh agricultural production in 2021, as a share of the gross agricultural output (Welsh Government, 2022). Livestock production is a key agricultural land use, with 86% of Welsh agricultural land being used for livestock grazing and 25% of all farms being sheep or cattle grazed on Less Favoured Area (LFA) land (Welsh Parliament, 2022). In addition, the management of livestock grazing is a significant rural source of employment (Chatterton et al., 2015). It is also associated with other ecosystem services, including cultural services related to the impact of grazing on rural landscapes and the positive effect on biodiversity that can arise from well-managed low-input grazing (Teague and Kreuter, 2020).

While lamb production remains a key UK industry, it is responsible for the release of GHGs, principally methane (CH_4) and nitrous oxide (N_2O) (O'Brien and Shalloo, 2021). It is known that GHGs vary both spatially and temporally across livestock grazed pastures, yet the magnitude of this variation is less well understood (Charteris et al., 2021; Thompson and Rowntree, 2020). Given the hardy nature of sheep, they are often grazed across altitudes (lowland, upland, and hill) in a variety of climatic and environmental conditions (Morris, 2017). The lack of understanding around emissions variations is therefore of particular importance to the sector. Research has shown variations in both CH_4 and N_2O emissions across altitudes, with N_2O generally being lower in upland and hill sites compared to lowland pastures and CH_4 varying significantly with available forage (Fraser et al., 2015; Mancia et al., 2022). However, literature documenting these differences for lamb production systems is sparse and the effect on life cycle assessment (LCA) and carbon footprinting (CF) remains unquantified.

LCA and CF are methodologies that have been used since the 1990s to quantify the environmental impacts of products through the entirety of their life cycle (Haas et al., 2000). LCA and CF has been completed for lamb production systems but less so than cattle, with a review of the current standard of LCA methodology within the sector being published only recently (Bhatt and Abbassi, 2021). Many of the recent assessments of Welsh lamb have followed Tier 1 or limited Tier 2 Intergovernmental Panel on Climate Change (IPCC) methodologies (Hyland et al., 2016; Jones et al., 2014; Taylor et al., 2010). In addition, the IPCC guidelines for quantifying emissions from livestock were updated in 2019, therefore these studies are outdated (IPCC, 2019, 2006). In addition, limitations exist within agriculture greenhouse gas inventories for fully accounting for the temporal and spatial variability of lamb production systems, particularly in terms of EF_{3prp} (Mancia et al., 2022). For example, the UK agriculture greenhouse gas inventory currently uses a scaled value of the UK cattle EF_{3prp} and is not sheep-specific (Brown et al., 2022). Developing methodologies appropriate for lamb production is therefore imperative, both in terms of calculations within modelling and improving model input parameters.

Given the increasing pressure on the agriculture sector to reduce emissions towards the Net Zero target in 2050, it is necessary to consider the impact of lamb production on the environment. Additionally, discussions are ongoing on future land uses in Wales, particularly in the uplands where sheep are largely grazed (Hardaker et al., 2020). Further developing knowledge related to emissions burdens across the altitudes involved in production, as well as emissions reductions options that are appropriate for grazing in contrasting altitudes is needed to aid in promoting sustainability and ensuring longevity of the sector.

1.2 Aims and objectives

The aim of this thesis is to assess GHG emissions from grazing pastures in different altitudes in a typical lamb production system and utilise these data within an altitudinally disaggregated CF tool, to assess uncertainties in underpinning data and effectiveness of mitigation strategies once altitudinal factors are accounted for.

The thesis objectives are as follows:

1. To critically evaluate current CF and LCA progress in relation to lamb production systems (Chapter 2)

2. To measure and assess any changes in enteric CH_4 (Ym) and excretal N_2O (EF_{3prp}) from both urine and faeces across the altitudes (lowland, upland, and hill) (Chapter 3)

3. To develop an altitudinally disaggregated model for lamb production systems that accounts for variation in key inputs across altitudes and compare with default methodologies (Chapter 4)

4. To assess the sensitivity of the CF results to different input parameters, highlight key areas of data unavailability, and assess the effect of different global warming potentials (GWP) (Chapter 5)

5. To investigate the efficacy of select mitigation strategies and assess the potential for introducing afforestation if land can be spared elsewhere on-farm. Additionally, to compare mitigation efficacy when using default methodologies versus site-specific modelling (Chapter 6)

6. To discuss limitations within this research and recommend areas for future study (Chapter 7)

1.3 Chapter outlines

Chapter 2 is a critical review that outlines the global lamb production industry and its contribution to climate change via release of GHGs. It further outlines the specific systems present in the UK and Wales. It then synthesises the current state of CF and LCA methodologies in relation to lamb production systems and critically evaluates key areas of uncertainty within these that are present for the sector. Mitigation strategies specific to lamb production are also described and discussed.

Chapter 3 describes a study that utilised an automated, mobile enteric CH_4 emission measurement system and manual static chamber methodology to assess enteric CH_4 and excretal N_2O that arise from sheep at different altitudes (lowland, upland, and hill). The aims were to determine any differences in emissions across altitudes and calculate a Ym (methane conversion factor) and EF_{3prp} (N_2O emission factor (EF)) value specific to each site.

Chapter 4 describes the development of an annual farm-level CF tool that has selected model inputs and EFs disaggregated by altitude. A comparison between default IPCC Tier 2 methodologies and the use of site-specific altitudinal data is made.

Chapter 5 is a sensitivity analysis of the tool to model inputs. The relative importance of selected input parameters to final CF results is tested, allowing identification of key areas where data availability needs to be addressed. The effect of uncertainty within measured site-specific emission factors (as described in Chapter 3) is also assessed. The data inputs are then ranked to show order of importance for further research. Finally, the effect of the IPCC Assessment Report global warming potentials (AR4, AR5 and AR6) on the footprint results are quantified.

Chapter 6 assess different mitigation scenarios that could be applied across altitudes in lamb production systems. The level of carbon (C) reduction using the default IPCC methodologies and site-specific inputs are compared to assess if strategy ranking changes with chosen methodology. In some cases, the chosen mitigation options result in land being spared on farm - therefore the potential for providing farm-offsets by introducing afforestation alongside the chosen mitigation options is also assessed.

Chapter 7 discusses where this thesis has areas of limitation that could be improved in future work. It also discusses the wider implications of the research and where future research should focus.

1.4 Thesis flowchart

A flowchart to represent the thesis structure and linkages is shown in Figure 1.1.

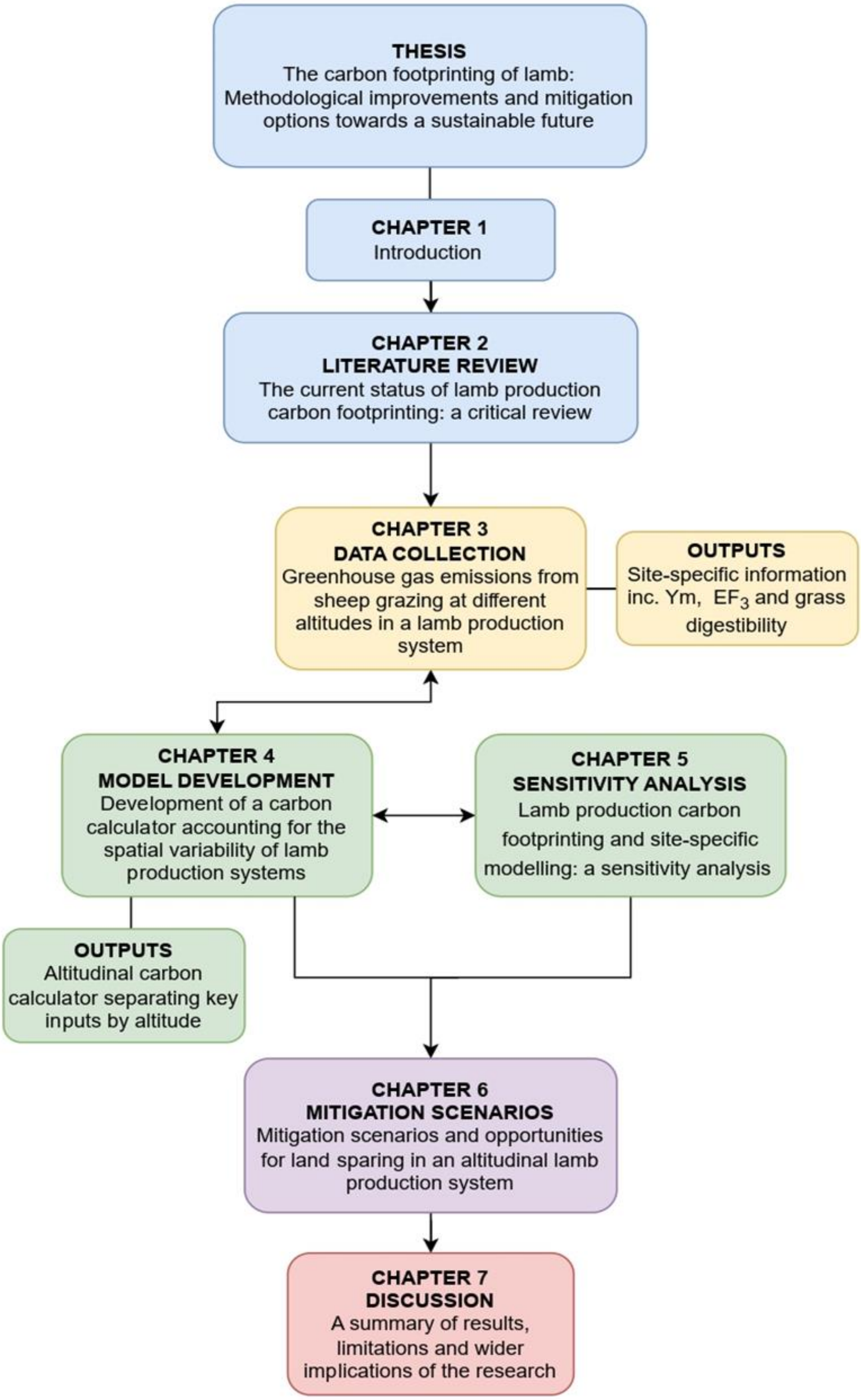

Figure 1.1 Thesis flowchart

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Chapter 2 The current status of lamb production carbon footprinting: a critical review

2.1 Background

Agriculture has experienced substantial growth globally over the last century, with overall production more than tripling between 1960 and 2011 (Alston and Pardey, 2014). More recently, crop production increased by approximately 53% and meat production by 44% between 2000 and 2019 (FAO, 2021). Predicted population growth alongside changes in dietary habits are expected to further increase demand for agricultural products. Agricultural productivity is therefore predicted to need to increase by 50% (of 2012 levels) by 2050 (Calicioglu et al., 2019). In addition, increasing global incomes and changes in diet further compound this demand. Livestock products specifically have likewise seen substantial growth; global *per capita* consumption was 25 kg in 1961 compared to 48 kg in 2013 (FAO, 2018). Overall demand is expected to rise by at least 72% but potentially by as much as 145% by 2050 (Godfray et al., 2018; Wu et al., 2014). A further implication for livestock sectors is the link between the increase in *per capita* wealth resulting in a shift towards greater consumption of livestock products. In low-income countries, animal protein intake was previously determined as 6 g day^{-1}, in comparison to high-income countries where 30 g day^{-1} was consumed (Sans and Combris, 2015). Lower income groups are expected to have increased population growth rate and demand for livestock products, presenting another inherent system inequality and problem that will need to be solved (Duro et al., 2020).

The expansion of the agricultural sector has resulted in growing concerns over the significant negative impacts that production of this scale is having on the environment, particularly climate change (Rotz, 2020). Livestock production contributes to climate change yet conversely depends on a productive environment. This conflict means that concerns over the ability to produce enough sustainable livestock products are intensifying (Harrison et al., 2021). Our understanding related to the breadth of impacts that climate change may have on livestock is not yet developed, but it is expected that there will be effects on feed and water availability, as well as animal health and the processing and transport of goods (Godde et al., 2021). Combined with the increasing population and demand for livestock products, the worldwide situation raises concerns for the sustainability of livestock systems and achieving global food security in the future (O'Mara, 2011).

A significant environmental impact associated with livestock systems is the production of greenhouse gases (GHGs), which are a key cause of climate change and occur across the entirety of the production process (Garnett, 2013). In addition, forestry loss and biodiversity decline occur due to agricultural land expansion and land use change for grazing and production of animal feed (Hanley et al., 2012). Impacts to water and soil also arise as farm management practices affect nutrient balances across the farm (Leip et al., 2015). Leaching of nutrients from agricultural systems (e.g., from fertilisation and animal wastes) to waterways further exacerbates eutrophication problems. Ammonia (NH_3) emissions from animal wastes result in acidification, contributing to poor air quality and habitat degradation (Hou et al., 2015). Soil quality decreases with acidification and soil erosion is now a major complication in agricultural production. NH_3 is commonly referred to as an indirect GHG where its release to the atmosphere affects cloud and aerosol formation, exacerbating warming effects in the atmosphere. It is estimated that ~14% of indirect N_2O arises from NH_3 and NO_x gases (Denmead et al., 2008). Additionally, the use of different resources is of concern (e.g. water or energy), either directly from on-farm activities or other related processes such as transportation and storage (Tullo et al., 2019). GHGs from livestock production in particular have been receiving increasing attention worldwide and livestock are now deemed to be a key anthropogenic source (Garnier et al., 2019; Jose et al., 2016; O'Brien and Shalloo, 2021; O'Mara, 2011). There are three major GHGs emitted as a result of production: methane (CH_4), carbon dioxide (CO_2) and nitrous oxide (N_2O). CH_4 is estimated to comprise 50% of livestock emissions, CO_2 26% and N_2O 24% (FAO, 2019). An additional concern for climate change is the effect it can have on worsening these other environmental impacts associated with livestock systems, understanding of which has not been well developed.

Given the high contribution of the livestock sector to global agriculture and its impact on socio-economic factors, it is imperative that solutions are found in improving sustainability of livestock production. As there is an expected increase in demand for animal products, consideration of how livestock systems can be a solution to the problem as opposed to a hindrance is important. Ruminant animals can consume feedstuffs unsuitable for human consumption due to them having a rumen, allowing digestion of coarse forage. Their manure can be used for crop fertilisation and their capability to survive on marginal land allows more productive land to be conserved for arable production. Breeding and producing animals that are appropriately adapted for their region of production can aid in increasing productivity, improving animal welfare and decreasing disease risk (Eisler et al., 2014). A variety of solutions are being suggested to mitigate livestock GHG outputs. Key animal strategies highlighted in the recent UK Marginal Abatement Cost Curves (MACC) include improving animal health, improving animal breeding, alterations to manure management practices and utilising dietary additives to reduce enteric CH_4 (Eory et al., 2020). Improving production efficiencies could allow a decrease in livestock numbers by decreasing the number of animals necessary to yield the required amount of product (Gill et al., 2010). There remains an inequality in reducing livestock impacts, where options have been evaluated for developed countries yet developing countries may experience less adoption of recommend practices due to inherent differences in the systems present (Arango et al., 2020). For example, use of dietary supplements to decrease enteric CH_4 may be too costly to smallholders and adequate support in effectively using the supplement may not be as readily available as in developed countries. In general, further research is required regarding mitigation options, how they interlink and what options may be most appropriate for different systems (Gerber et al., 2013).

Extensification of livestock systems whereby fertiliser applications and stocking rates are reduced, is considered as one process by which sustainability and promotion of wider environmental benefits can be achieved (Ripoll-Bosch et al., 2013). Sheep are one ruminant species highlighted in the context of this, particularly for UK based systems. Sheep in the UK are often grazed on upland and hill marginal grasslands as well as on higher quality lowland fields, meaning animals can be moved frequently. The role of variable environmental aspects of these different altitudes of grazing, for example increased soil acidity in the uplands, on GHGs and the overall carbon footprint (CF) of the meat produced is not clear. Sheep can also contribute to conservation grazing as controlled numbers can result in varied plant structures and promote carbon (C) sequestration, although evidence suggests this may be a slow process (Barthram et al., 2002; Marriott et al., 2009). Sheep are suitable for grazing on marginal land, as they are generally capable of surviving on low quality forage and on steep slopes with harsher climates (Dwyer, 2008). Full analysis of the environmental implications and trade-offs of extensification compared to intensification is not yet well documented. It is difficult to directly compare intensive and extensive systems In terms of emissions intensity (GHG emissions per unit output), intensive systems have higher productivity therefore lower intensity (and CF) (Eldesouky et al., 2018). However, extensive grazing is generally associated with lower stocking rates and animal inputs therefore lower net GHG emissions. They may also be more able to offset emissions due to larger land area and potential for tree cover, but C sequestration is not commonly assessed in CF studies. Consideration of emissions intensity and net GHGs is therefore crucial in accurately assessing systems.

A key tool implemented when attempting to understand the environmental impacts of a system is life cycle assessment (LCA) (Rebitzer et al., 2004). Correct and robust use of

LCA can aid in informing supply chains and assessing where improvements can be made (Edwards-Jones et al., 2009). The LCA process quantifies the environmental burdens related to a product during all stages of its life cycle, from its manufacture to shipping through to its use and final discarding. These impacts are assessed regarding a functional unit, for example within lamb footprinting systems this may be 1 kg of lamb meat. LCA has been applied across a variety of sheep production systems for both meat and milk production (Ibidhi et al., 2017a; Jones et al., 2014a; Plaza et al., 2021; Taylor et al., 2010; Wiedemann et al., 2016). Though, application of LCA to sheep systems is less developed compared with application to cattle systems, and there is considerable scope for improvement in how specificities of sheep systems are represented.

Uncertainties arise within LCA modelling, for example from lack of empirical data and high variability between systems, leading to variations in results particularly for agricultural footprints (Niero et al., 2015). Natural variation in the measured GHGs (CH_4, N_2O and CO_2) may not be fully captured by emission factors (EFs), creating uncertainties in the model itself alongside difficulties defining a system boundary and accounting for multifunctionality of a system (Chen and Corson, 2014; Ripoll-Bosch et al., 2013). Variation is also known to occur between production systems due to contrasting environmental conditions (Jones et al, 2014). As mentioned, sheep systems are highly variable involving frequent movement of animals to areas of different environmental conditions throughout the year. This variability combined with the lack of understanding of system benefits makes application of LCA to the sheep sector particularly challenging.

2.2 Overview of lamb production systems

2.2.1 Global overview of production

Globally, over 1.2 billion sheep have been recorded (FAO, 2019). They were one of the first animal groups to be domesticated and have since been bred extensively worldwide as a source of nutrition and for wool (Ramankutty et al., 2018). The total number of sheep globally is expected to continue to increase 60% by 2050 (Marino et al., 2016). Sheep meat production is spread widely across the world with China leading production, followed by Australia, New Zealand, and the United Kingdom (UK) respectively with 32.9 million sheep currently being farmed within the UK (as of June 2021) (DEFRA, 2022a). Global lamb meat consumption comprises ~5% of total meat consumed. Patterns of sheep meat consumption vary significantly between countries with overall consumption decreasing in the majority of countries, with the UK seeing a 32.8% decrease (between 2000 and 2019). Even so, consumption is increasing across Asia and overall global consumption has increased 6.7% since 2000 (Whitton et al., 2021).

There exist three main production systems within the lamb production industry globally, including extensive systems for wool and meat production, intensive systems, and a traditional pastoral system (Benoit et al., 2019). Production systems occur across a range of grazing areas with differing climatic variables, with animals bred to be adapted to these variable conditions. Systems in the UK are often stratified, referring to a method of production that originates in the 20th century. It refers to animals being grazed across lowland, upland, and hill areas to increase production, with the hardier hill breeds bred on the mountain and then moved to the uplands. Surplus animals from this production tier are then moved to the lowlands to be further crossbred with larger lowland sires (Mansfield, 2015). This allows production traits, including hardiness and meat quality, from all levels of production to be hybridised in one offspring. In some cases, animals will remain in one tier of production for all of their productive lives (Rodriguez-Ledesma et al., 2011).

The global exports of sheep meat are dominated by Australia and New Zealand who account for 68% of exports (Colby, 2016). The UK follows as the 3rd biggest exporter sharing 9% of the total. It is estimated trade from sheep products is worth US$5.9 billion globally.

2.2.2 The UK perspective

Sheep farming has taken place in the UK for thousands of years. As a result, it has played a key role in developing the character of the UK landscape today. There are sheep across the entirety of the UK, but a significant population of sheep are found in the uplands in part due to their hardy nature and ability to survive in the harsher conditions. The uplands are a diverse range of habitats (generally at least 250m above sea level) that are highly important natural areas contributing to the rural economy and farming, UK water supply, biodiversity, C sequestration and tourism. Large areas of the uplands are protected as conservation areas or national parks (DEFRA, 2011). Sheep numbers have fluctuated across the UK since the early 2000s but remained broadly stable. A small drop was seen at the UK level in 2010, with a similar but less significant drop seen in Wales in the same year. The most recent figures (2022) show that there are currently approximately 9.9 million sheep in Wales with 33.1 million across the UK as a whole (Figure 2.1) (Welsh Government, 2022a).

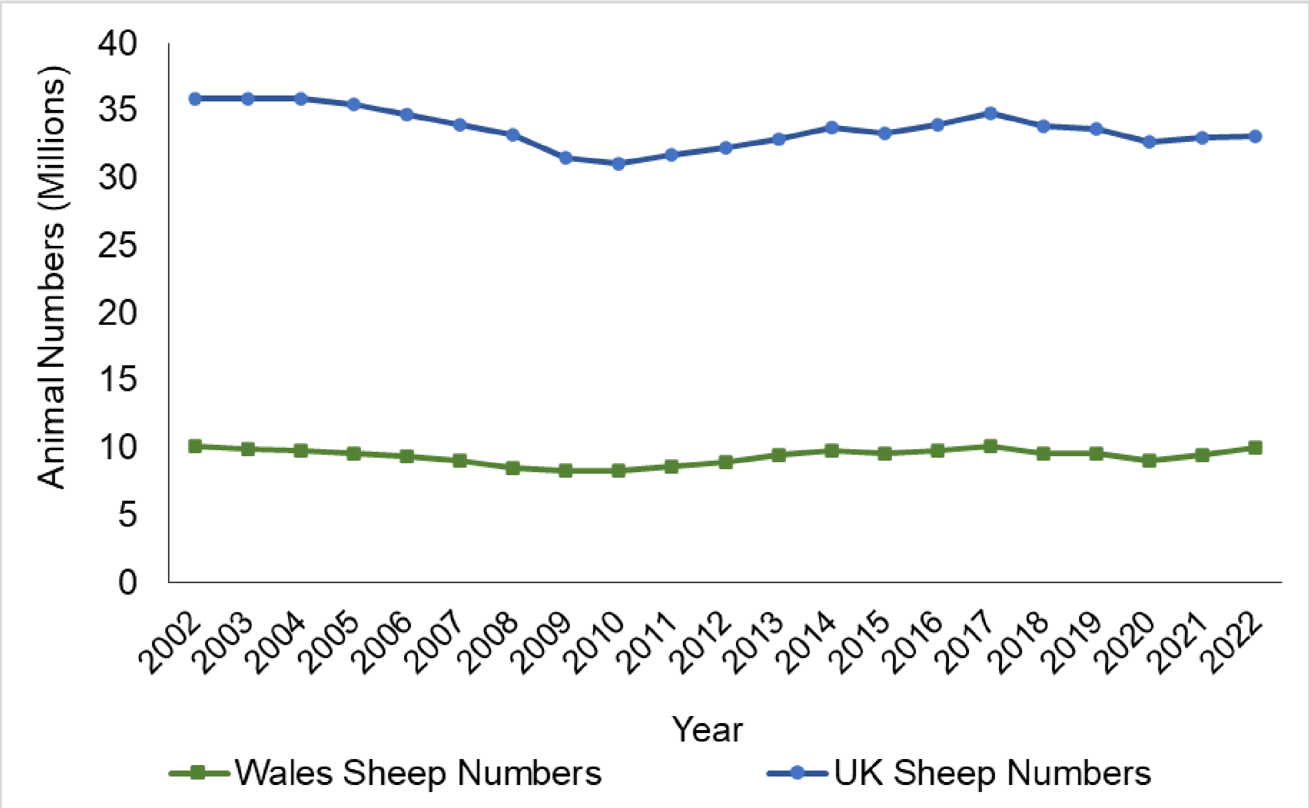

Figure 2.1 Total sheep numbers since 2002 in Wales and the UK (Million) (Welsh Government, 2022a).

Just under 9 million of Welsh sheep are currently being farmed in less favoured area (LFA) land, split between disadvantaged and severely disadvantaged land. LFA describes land with low agricultural productivity (Welsh Government, 2018). Overall, it has been estimated that 43% of the UK breeding ewe population are hill type breeds (Zhao et al, 2017). Animals being farmed primarily in extensive areas highlights the diversity of production systems that may be found and the wide-reaching implications that the systems can have as a whole.

From an economic perspective, total sheep meat produced within the UK was 299,900 tonnes, equating to a national income value of £2,474million in 2018 (AHDB, 2018). Historical data of sheep meat production is presented in Figure 2.2 (DEFRA, 2023).

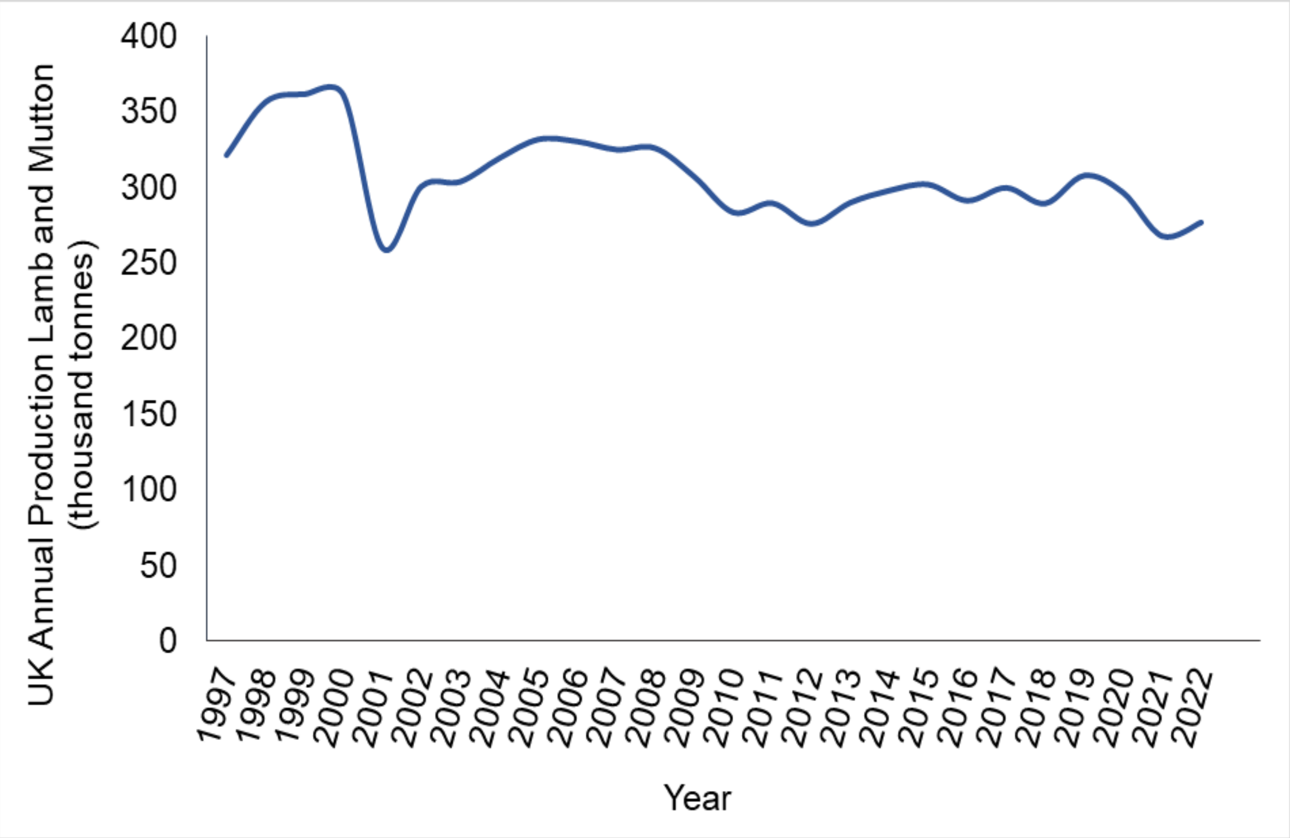

Figure 2.2 UK annual lamb and mutton production (thousand tonnes) (DEFRA, 2023). Data are presented at a UK level only, as Wales level data was not available.

With regards to employment, the most recent figures found in the UK (2015) revealed that lamb production was associated with 34,000 farming positions, with 111,415 additional jobs linked to the sector although not specifically on-farm. This employment was estimated to have a worth of £291.4million (NFU, 2018).

With regards to policy, agriculture is currently undergoing a period of change in the UK and Wales following the UK leaving the European Union in 2016. The Agriculture (Wales) Bill is being presented to the Senedd this year and is underpinned by sustainable land management (Welsh Government, 2022b). It includes aims to produce food sustainably, mitigate agricultural impacts on the climate, increase ecosystem resilience and to protect the wider cultural aspects of Welsh agriculture (including countryside access and the Welsh language). The Sustainable Farming Scheme is a new scheme that is set to replace previous agricultural subsidies in Wales from 2025 (Welsh Government, 2022c). It is still under development but proposes having a variety of actions that farmers can partake in to meet the requirements for subsidy. Actions relevant to sheep farming include grazing and sward management, undertaking carbon assessments and having 10% of the farm area under afforestation.

2.3 Greenhouse gas emissions

2.3.1 Introduction to greenhouse gas emissions

Anthropogenic emission of GHGs and the resulting climate change pose global consequences in environmental, economic, and social terms (Pachauri et al., 2015). Livestock production is deemed as a key global contributor to GHG emissions (FAO, 2018). A study by the Food and Agriculture Organisation (FAO) calculating total lifecycle emissions states that the sector represents 14.5% of global anthropogenic GHG emissions, with 6.5% of that total deriving from small ruminants specifically (Gerber et al., 2013). Other studies based on Intergovernmental Panel on Climate Change (IPCC) direct emissions methodology (therefore do not include wider lifecycle emissions such as transport and production of inputs) quantify livestock derived emissions as being 9% of total anthropogenic GHGs (Caro et al., 2014). More recent work claims that the initial FAO figure of 14.5% should be revaluated to at least 16.5%, representing an increased share of global GHGs being assigned to livestock. This increase is due to revised accounting of the GHG contribution of land use change from feed production and use of grazing land (Twine, 2021).

The main GHGs produced from livestock production include CH_4, N_2O and CO_2. CH_4 is generated primarily through enteric fermentation with a smaller proportion from manure management (Kumari et al., 2020). N_2O arises from manure management, deposition of animal excreta to pasture and fertiliser use (during land management and animal feed production) and CO_2 arises from energy use on farm as well as emissions from the production of animal feeds and land use change (Edwards-Jones et al., 2009; Saggar et al., 2015). Recent UK agricultural statistics state that ~10% of emissions are related to the agriculture sector, with CH_4 accounting for 55% and N_2O 32% of agricultural emissions (BEIS, 2021). In the 2020 inventory, UK enteric fermentation emissions totalled 23450kt and manure management emissions 6771kt across the agricultural sector (Garland et al., 2022). Reduction in emissions is intrinsic to sustainability of agriculture and achieving food security. The UK has achieved overall emissions reduction in recent years, particularly in the power sector. However, reduction of emissions in the agriculture sector has lagged therefore focus on this area is necessary to achieve climate change targets (CCC, 2018).

2.3.2 Methane emissions

2.3.2.1 Introduction to methane

CH_4 is a major GHG produced in livestock systems with 1 kg of the gas, in relation to 1 kg of CO_2, having an estimated global warming potential (GWP) 27 times greater (over a 100 year time frame) (IPCC, 2021). The main source of CH_4 in agricultural systems is from enteric fermentation in ruminants. Ruminants are multi-stomached animals with a stomach chamber named the rumen, where enteric fermentation takes place. This is a digestive process whereby approximately 200 species of microbe break down and ferment the plant material, producing CH_4 as a by-product (Haque, 2018). This allows the animal to effectively gain sustenance from plant materials that are difficult to digest. Most CH_4 produced by the animal is released due to eructation with no more than 3% of overall CH_4 produced in the hindgut and released as flatulence (Hristov et al., 2015). Another source of CH_4 from livestock systems is manure management, relating to both the urine and dung produced. When manure is stored and decomposes anaerobically, it produces CH_4. The amount of CH_4 produced depends on the amount of manure overall and how it is stored. If large amounts are stored as a liquid in for example a pit or tank, larger amounts of CH_4 are produced due to the increased anaerobic conditions (Broucek, 2018). However in the context of UK sheep farming, the majority of sheep are kept outdoors so CH_4 from manure management is generally of low consequence (van Amstel and Swart, 1994).

2.3.2.2 Methane measurement technologies

CH_4 measurement from ruminants can be conducted in various ways with the use of respiration chambers (RC) (Sun et al., 2015) and the sulphur hexafluoride tracer (SF_6) (Dong et al., 2019) being commonplace. The SF_6 tracer involves insertion of a small permeation tube into the rumen of the selected animal and the SF_6 and CH_4 concentrations are quantified at the mouth and nose (Williams et al., 2011). Use of the tracer allows determination of gas concentrations despite dilution of CH_4 post eructation (Johnson et al., 1994). Use of RC involves the animal being kept in a closed chamber for a set period of time allowing total quantification of gas from the digestive tract (eructation and flatus) (Pinares-Patiño et al., 2011).

Another methodology, and the one used throughout this study, is the GreenFeed (GF) system developed by C-Lock Inc., which is a static device that measures the CH_4 from an animal's breath (Zimmerman and Zimmerman, 2016). When the systems proximity sensor detects an animal within the head chamber, it offers a reward in the form of a small drop of feed. This encourages the animal to visit the system frequently, resulting in multiple short-term CH_4 measurements that are used to derive an overall emission rate. The system measures gas concentrations by use of a nondispersive near infra-red analyser (NDIR), which is calibrated using certified zero and span gases (McGinn et al, 2021). The timing of feed drops and number of drops available per animal per day can be altered to enable a custom 24 hour overview of emissions from the animal (Hammond et al., 2015).

Each methodology has its strengths and weaknesses that must be accounted for in an experimental protocol. RC is a highly precise method however does come with high costs and is unsuitable for stimulation of natural animal behaviours during grazing (Goopy et al., 2016). Forage selection, movement and eating patterns are all inhibited by RC, which is likely to affect overall emissions quantification in comparison to grazing animals (Huhtanen et al., 2019). The SF6 approach is more representative of natural behaviour as it allows diet selection within the animals but the insertion of rumen boluses and regular animal control that is involved makes it labour intensive. The halter used for CH_4 measurement can also impact grazing behaviour and background CH_4 levels can cause inaccurate estimation of emissions (Hammond et al., 2015). By contrast, GF system allows evaluation of grazing systems and is relatively low in labour inputs. It does not involve excessive animal handling as use of the unit is voluntary, although the voluntary aspect introduces a key weakness as there is a possibility that animals will not visit the system throughout the experimental period (Della Rosa et al., 2021). This may result in trials having higher animal numbers and longer time periods. Additionally, variations in diurnal CH_4 may not be fully captured if the animal chooses not to visit at different times of the day (Hristov et al., 2016). GF, like SF6, also does not account for CH_4 released as flatus. A summary of these strengths and weaknesses are presented in Table 2.1.

Table 2.1 Summary of strengths and weaknesses of different methane measurement techniques.

Methodology	Strengths	Weaknesses
Respiration chamber	• Associated with high precision • Captures CH_4 from eructation and flatulence • Can capture CH_4 over full diurnal timeframes	• Costly • Inhibits natural animal behaviour • Cannot measure under grazing conditions
Sulphur hexafluoride tracer (SF_6)	• More representative of natural animal behaviour • Relatively inexpensive • Can measure large number of animals	• Labour intensive • CH_4 measurement halter can affect grazing behaviour • Does not account for CH_4 from flatulence • Background CH_4 may affect emissions measurements
GreenFeed	• Can be used indoor and outdoor under grazing conditions • Has been assessed as comparable to other methods • Not labour intensive and involves little animal handling	• Relies on animals visiting the unit • Evidence suggests environmental factors may affect validity • Does not account for CH_4 from flatulence • May need more animals or longer time frames to gather enough data

The validity of GF measurements against other methods of measuring CH_4 has been assessed in multiple trials. One study found that GF appeared to be reliable in estimating animal CH_4 output due to the similarity between measurements from a GF unit and estimations derived from RC using a model (Huhtanen et al., 2019). Direct comparisons of GF and RC also gave close similarity in results. A review of 22 experiments that compared

GF with wider CH_4 measurement techniques found GF measured emissions were correlated (medium to high) with RC and SF6 measurements (Zimmerman and Zimmerman, 2016). Another more recent study generated similar conclusions (McGinn et al., 2021). The results from the GF were found to be similar to those measured in a RC for CO_2 and CH_4, with differences being equal to or less than 3%. The small difference was concluded to be due to systematic error. There were concerns surrounding the sensitivity of the NDIR analyser within GF to assess background CH_4 concentrations but according to McGinn et al., 2021, it was deemed unimportant due to the high levels of CH_4 eructed by the cattle when in the machine. However, concern was speculated for sheep units due to the lower levels of CH_4 produced by the animal.

Other work has found larger differences between techniques. A comparison of GF and SF_6 comparatively low correlation between the two methods, with an average difference of 8% (Hristov et al., 2016). Mean CH_4 measurement (g animal^{-1} day^{-1}) was 373 using the GF unit and 405 using the SF_6 tracer. Reasons for the difference were thought to be due to ventilation levels within the barn affecting background CH_4 and SF_6 concentrations. An evaluation comparing 397 publications of CH_4 measurement technologies found high variation in experimental protocol that affected the ability to accurately assess differences between the technologies (Della Rosa et al., 2021). For example, differences in experimental length, reporting of technical processes followed within experiments (such as individual measurement lengths using GF and gas recovery data using RC) affected confidence in final results. Standardisation of protocols as well as the data that should accompany publications (e.g., visit numbers per animal using GF) using different measurement techniques was recommended to increase robustness of measurements.

2.3.3 Nitrous oxide from agriculture

N_2O is another key GHG that is released from livestock systems, and has a GWP approximately 273 times more than CO_2 in the IPCC Sixth Assessment Report (IPCC, 2021). Globally, agriculture remains the dominant source of N_2O followed by industry and the burning of fossil fuels. Within agriculture, use of nitrogen (N) fertilisers and management of animal wastes are the key origins (Reay et al., 2012). How manure is stored (solid storage, slurry or anaerobic lagoon) and the animal housing system present (bedding used and frequency of stall cleaning) both influence N_2O emissions (Broucek, 2018). For grass-based ruminant systems, a significant source of N_2O is through deposition of animal excreta on grassland (Krol et al., 2016). If the nitrogen (N) levels present in animal excreta when applied to grassland are in excess of plant requirements, this excess is lost via various processes and N_2O is one such loss (Selbie et al., 2015). In urine, urea is the dominant N compound and undergoes hydrolysis, producing the ammonium ion (NH_4^+) as well as hydroxyl ions (OH^-) and CO_2. N_2O is produced when NH_4^+ is converted due to microbial activity via nitrification and denitrification (Hallin et al., 2018). In dung, the N is primarily in its organic form therefore mineralisation must take place before there is availability of NH_4^+ for denitrification and nitrification (and resulting N_2O emission) (Chadwick et al., 2018). This occurs over a longer period of time than in urine therefore urine generally has increased emissions due to the N present being more readily available (Liu and Zhou, 2014).

2.4 Life cycle assessment and carbon footprinting

2.4.1 Background

Global concern regarding the impact of different sectors on the environment, in particular in the context of climate change, has resulted in increased efforts to develop approaches that will account for these impacts. Modelling methodologies and GHG accounting tools have therefore been created. Life cycle assessment (LCA) is one such modelling method and has become a common way of assessing the environmental impacts of a product or system. Use of early forms of the method began in the latter half of the 20th century and towards the 1990s, the full LCA method used currently became more widespread (Guinée et al., 2011). International Organization for Standardization (ISO) have now developed two international standards that define the principles and frameworks to follow in an LCA study: ISO 14040 and ISO 14044 both released in 2006 (International Organization for Standardization, 2006a, 2006b). These principles remain the core of LCA studies, but the method has undergone significant development and continues to do so. The principles do not contain in-depth guidance on the processes and EFs to include when modelling livestock systems. Wider guidance aligned to the ISO principles has been developed for livestock, and specifically small ruminant, systems by the FAO (FAO, 2014).

The United Nations Convention on Climate Change (UNFCCC) makes it an international requirement for countries to calculate and declare their annual emissions (or fluxes) across all sectors. Methodologies to calculate emissions are issued by the IPCC and exist on a tiered basis (IPCC, 2019, 2006). There are three tiers available to follow: 1, 2 and 3. The most basic approach is Tier 1, which utilises national data on e.g., animal population (for ruminant CH_4 emissions) and fertiliser N use (for N_2O emissions) with default IPCC EFs for each livestock cohort. Tier 2 utilises data that is specific to the country requiring information regarding the gross energy intake (GEI) and CH_4 conversion factors related to particular animal cohorts and N_2O EFs that reflect national soils and climate. Tier 3 is the most complex methodology and involves more sophisticated country-specific data and process modelling. This may be in the form of a model that utilises specific information such as feed characteristics, disaggregated process specific EFs and any seasonal variations in input data that may be present. CF, an LCA study focussed on the singular environmental impact category of global warming potential (i.e. GHGs) is often underpinned by these IPCC guidelines (Sukhoveeva, 2021).

With 190 parties signing the Paris Agreement and committing to limiting global warming by well below 2°C (Savaresi, 2016), nations must take action to reduce GHG emissions from all sectors. Agriculture and livestock have an important role to play in reaching this target as key contributors to GHG emissions. The UK has set a target of Net Zero emissions by 2050, which requires balancing the sources and sinks of GHG emissions so that they equal net zero (CCC, 2020). The EU has also adopted this target (European Commission, 2021). In this context, the accurate quantification of GHG emissions from livestock is paramount in order to direct effective mitigation strategies. While this quantification is not straightforward, it is essential in reaching sustainable production goals (Sykes et al., 2017). While reporting annual emissions in the country inventory is applicable for monitoring national progress towards C targets, application of C calculators at a farm-scale can be of further help in terms of expressing emissions at an area and product unit level. This can aid in benchmarking farm and product-level performance and selection of mitigation options (Martineau et al., 2019). CF tools have a variety of benefits including many being online and free to use, applicable for a wide range of farms and can effectively be used to assess farm-scale impacts. However, different tools have different approaches to methodology and scope, causing disparity in final results (Sukhoveeva, 2021). C calculators therefore must continue to be improved, particularly for pasture-based sheep systems with high complexity, due to frequent animal movement and grazing of a variety of pastures. Particularly as they have been subject to less scrutiny than beef and dairy systems (Bhatt and Abbassi, 2021).

2.4.2 The current status of carbon footprinting within lamb production

Global analyses of lamb production CFs have given a range of results. A recent meta-analysis assessing data from 119 countries determined a mean value of 20 kg CO_2eq 100g protein^{-1} for lamb and mutton (Poore and Nemecek, 2018). FAO estimates give values of 19.1 kg CO_2eq kg product^{-1} for sheep meat and 6.0 kg CO_2eq kg product^{-1} for sheep milk (FAO, 2022). Sheep system emissions occur predominately during the on-farm phase of production, largely due to CH_4 from enteric fermentation and N_2O emitted from soils due to deposition of animal excreta and/or fertilisers (Geß et al., 2022). A summary of global CF results alongside the contribution of different emissions to the total (where available) are shown in Table 2.2. This summary is not intended to be exhaustive.

Table 2.2 Summary of global carbon footprinting studies for sheep meat (summary not exhaustive).

Study	Country	GHG emissions	GHG emissions split (% of footprint)
(O'Brien et al., 2016)	Ireland	10.4 CO_2eq kg LW^{-1} lowland 8.7 CO_2eq kg LW^{-1} lowland with grassland carbon sequestration 14.2 CO_2eq kg LW^{-1} hill 7.0 CO_2eq kg LW^{-1} hill with grassland carbon sequestration	Enteric CH_4 61 - 68% Manure management CH_4 and N_2O 23 – 25% Fertiliser 8 – 12%
(Yetişgin et al., 2022)	Turkey	20.8 kg CO_2eq kg LW^{-1} transhumance 25.4 kg CO_2eq kg LW^{-1} semi-intensive	Data not provided
(Morgan-Davies et al., 2021)	Scotland, Ireland, Norway, France	10.35 – 30.44 kg CO_2eq kg LW^{-1} dependent on management system and country	Data not provided
(Toro-Mujica et al., 2016)	Chile	11.84 ± 2.06 kg CO_2eq kg LW^{-1} mean value without sequestration 10.7 ± 2.0 kg CO_2eq kg LW^{-1} mean value with sequestration	Enteric CH_4 75% Manure Management N_2O 11%
(Horrillo et al., 2020)	Spain	11.42 – 13.24 kg CO_2eq kg LW^{-1}	Enteric CH_4 78 – 80% Manure and Soil Management N_2O and CH_4 17%
(Peri et al., 2020)	Patagonia	10.64 to 41.32 kg CO_2eq kg lamb $meat^{-1}$	Enteric CH_4 60 – 65% Direct and indirect N_2O 27 – 30% Other energy use 7%
(Mazzetto, et al., 2023)	New Zealand	6.01 kg CO_2eq kg LW^{-1} (mean value, country-level estimate)	CH_4 84% N_2O 10% Other CO_2 8%
(Bhatt and Abbassi, 2022)	Canada	13.2 ± 3.7 kg CO_2eq kg LW^{-1}	Enteric CH_4 39% Manure Management CH_4 and N_2O 10% Feed production 29% Operations 23%

In the UK, historically the farm type has been closely associated with emissions. For example, lamb produced in lowland farms have been associated with lower emissions than those produced by upland and hill farms on a per unit product basis. The most recent peer-reviewed assessment of UK lamb C footprints that is published, gave mean values of 10.9, 12.9 and 17.9 kg CO_2 equivalent (CO_2eq) kg liveweight^{-1} (LW) for lowland, upland, and hill lamb production systems respectively. This included 64 farms across England and Wales (Jones et al, 2014). Similarly, an England-based report commissioned by the Agriculture and Horticulture Development Board (AHDB) gave values of 11.0, 10.9 and 14.4 kg CO_2eq kg LW^{-1} for lowland, upland and hill lamb showing a slight decrease when moving from lowland to upland (EBLEX, 2012). While variation between farm type is consistently seen, variation within farm type was similarly detected in both the above studies (Table 2.3). A range of values were reported with quite large differences between minimum and maximum values, highlighting that on-farm efficiency represents an area of possible significant gain when considering reduction of sheep system CF.

Table 2.3 Breakdown of min, max and mean values of CF per farm classification (kg CO_2eq kg liveweight (LW)$^{-1}$ finished lamb) (EBLEX, 2012; Jones et al., 2014)

Farm Type	Lowland			Upland			Hill		
	Min	**Max**	**Mean**	**Min**	**Max**	**Mean**	**Min**	**Max**	**Mean**
(Jones et al, 2014)	5.4	21.5	10.9	8.3	18.3	12.9	8.8	33.3	17.9
(EBLEX, 2012)	6.4	17.8	11.0	8.3	15.4	10.9	8.4	19.7	14.4

Older studies further highlight this variation, with lamb footprint values of 7 to 51 kg CO_2eq kg LW^{-1} being reported (Taylor et al., 2010). However, it is important to highlight that the EFs used within these studies are based on IPCC 2006 values, which have since been updated in 2019 (IPCC, 2019). It is therefore important to exercise caution when comparing these publications with studies that utilise the updated IPCC 2019 guidelines.

A more recent report by Hybu Cig Cymru (HCC) aimed to provide an updated view of ruminant systems in Wales. Within the report, an updated C calculator was used to calculate CF of lamb. The improvements include revisions to CH_4 and N_2O EFs and UK-specific data. Values of 11.5, 12.5 and 10.1 (kg CO_2eq kg LW^{-1}) not including C sequestration and 9.5, 10.5 and 7.6 (kg CO_2eq kg LW^{-1}) including C sequestration were reported for lowland, upland and hill sheep, respectively (HCC, 2021).

2.5 Sources of uncertainty in carbon footprinting related to lamb production systems

C calculators represent an opportunity to better understand the environmental impacts of agricultural systems, but there still exist inherent limitations that prevent tools from fully capturing system impacts. Use of CF requires assumptions to be made about systems and simplification of input data and methodology. These limitations can be related to CF modelling itself in terms of methodology and input data used but also in reference to information about the system that is being assessed. This can include EFs used and local variations in systems of the study region. Absolute measurement of key GHGs, such as CH_4 and N_2O, comes with significant uncertainty (Hristov et al, 2018). Variation in GHGs can occur at the farm level due to different animal breeds, seasonal changeability in system inputs and weather as well as other local influences. For example, CH_4 EF has been found to vary by season and breed (Islam et al., 2022). These variations are not always fully captured in default EFs and CF results, therefore introducing uncertainty to the results. Agricultural systems in general are particularly challenging to assess due to the multiple inputs and outputs involved at a farm level (Geß et al., 2020).

A variety of different C calculators are currently available for calculating GHGs from agricultural systems produced by a range of organisations from academic to consultancies and not-for-profits. Some examples include AgreCalc (SAC Consulting) (AgreCalc, 2023), EX-ACT (FAO) (FAO, 2023) and EAg-RET (AHDB, ADAS) (Skirvin et al., 2015). In-depth comparisons of specific selected tools have been conducted previously, particularly for application within beef production systems. One review found over 580 tools available for application to Welsh agriculture, although after applying selected exclusion criteria (such as the breadth of focus of the model and calculation scale) 14 remained within the analysis. Following this, further assessment resulted in only 3 tools scoring greater than 50% across different criteria (e.g., robustness and ease of use). These three tools were AgreCalc, CFF Farm Carbon Calculator and Cool Farm Tool (Taft et al., 2018). The same three tools were identified as particularly suitable in further work related to Welsh agriculture: AgRE Calc, CFF Farm Carbon Calculator and Cool Farm Tool (Martineau et al., 2019).

Although determined as suitable, results have been found to vary between the three calculators identified above, in an assessment of beef production (Sykes et al., 2017). Reasons for this have been attributed to differences in methodology, for example the CFF Farm Carbon Calculator appeared to omit certain emissions sources such as emissions from fertiliser production. Differences in how the calculators estimated emissions related to manure management and embedded emissions (e.g., production of inputs) were also highlighted as the source of potential differences. This has been identified by other work where variation in emissions methodology, particularly for N_2O, as well as the lack of uncertainty analysis appeared to underpin variation between calculators (Whittaker et al., 2013). These studies highlight the discrepancy between different tools when considering the same input data. Care must therefore be taken when comparing systems that are assessed by different calculators. It is also possible to identify limitations within these calculators in the context of lamb production systems and highlight potential improvements that could be made.

2.6 Emission factors

2.6.1 Variability in methane from enteric fermentation

CH_4 from enteric fermentation is a core component of any livestock CF. Calculation following IPCC guidelines involves knowledge of the GEI of the animal as well as use of a CH_4 conversion factor (Ym, % of GEI released as CH_4) (IPCC, 2019). The guidelines recommend a general value of 6.7% for adult sheep to be used when calculating enteric CH_4. This value is deemed as most applicable to systems where animals consume 0.6 – 0.8 kg DMI day^{-1}. However, a value of 7.0% is suggested for intakes <0.6 kg DMI day^{-1} and 6.5% for intakes greater than 0.8 kg DMI day^{-1}. Variations in the key factors that influence the Ym value have implications for CH_4 calculation and results. For example, factors such as the digestibility of the diet, feed intake and animal weight are known to have an effect on emissions and therefore calculation of the Ym used within modelling (Liu et al., 2017). In particular, the diet of the animal is known to affect absolute CH_4 output per animal (Shibata and Terada, 2010) This includes total feed intake, the feed type, for example feed species and how the feed was processed and feed digestibility. A review of dairy cattle found positive correlation between DMI and GEI and CH_4 emissions output (Min et al., 2022). This has also been found in sheep with a positive linear association between DMI and CH_4 ($r=0.77$) (Moorby et al., 2015). Forages higher in structural carbohydrates fermented in the rumen longer resulting in increased CH_4 (Sun et al., 2022). Different carbohydrates affect the digesta passage rate and pH of the rumen, further influencing CH_4 emissions. Environmental factors such as temperature are likewise relevant. Exposing sheep to a decrease in temperature reduced the length of time that feed was retained in the rumen, resulting in a decrease in CH_4 output (Barnett et al., 2015). Animal genetics, breeding and size all have a contribution although studies find that forage type appears to have a greater influence than breed type on CH_4 emissions (Fraser et al., 2015; Zhao et al., 2017).

As lamb production systems can (although not always) involve frequent movements between different pasture types, it could be expected that differences may occur in CH_4 output dependent on where the animal is grazed. For example, it was recently found that an increase in consumption of ericaceous species (common on hill areas used for grazing) within deer and sheep populations in Europe, resulted in a decrease in CH_4 emissions per unit of digestible DMI. A Europe-wide modelling exercise whereby it was assumed that the animals had 30% of their diet from ericaceous species resulted in a 68.3kt year^{-1} decrease of CH_4 in comparison to a grass only diet (Pérez-Barbería et al., 2020). However, there is limited evidence directly comparing different forages typically found and consumed by the animals in lamb production systems. Studies conducted with lowland diets found differences between typical lowland forages. For example, forage rape was found to have 22-30% lower emissions than perennial ryegrass (Sun et al., 2015). One comparison of sheep fed lowland grass with those fed hill grass showed that lowland grass resulted in an increased DMI and CH_4 output (g day^{-1}) than when fed hill grass. However, when expressed relevant to DMI, organic matter intake (OMI) and GEI, there was no difference found in CH_4 output between the forages (Zhao et al., 2017). Similarly, lambs fed ryegrass had higher daily CH_4 than those fed permanent pasture (16.1 g day^{-1} compared to 12.9 g day^{-1}) (Fraser et al., 2015). However, hill forages had lower metabolisable energy (ME) content and energy and N utilisation efficiency than lowland forage. CH_4 output was therefore greater from hill grass than lowland grass and pelleted ryegrass when put in relation to kg digestible DMI and digestible organic matter intake (DOMI) (Zhao et al., 2017). This was also shown in lambs, with CH_4 unit DMI^{-1} and GEI^{-1} lost as CH_4 both being greater from permanent pasture (Fraser et al., 2015).

A similar study conducted with cattle showed results comparable to those above. Upland cattle had lower mean daily DMI, CH_4 output and body weight gain (BWG) than lowland cattle but there was no significant difference in the CH_4 produced per unit feed intake on a DM or GEI basis between the two (lowland and upland cattle). However, when presented in respect to BWG, upland cattle had much higher daily CH_4 output per kg weight gain (261 v. 197g CH_4 kg^{-1}), representing the poorer nutrition received from upland grass. As a result, the benefit of the lower absolute emissions from upland cattle may be offset by the slower growth rate and therefore longer finishing time (Richmond et al., 2015). Understanding the factors that affect emissions output are important in reference to improving CF methodologies and inventory reporting. However, measuring CH_4 emissions under grazing conditions, particularly extensive grazing, is problematic as commonly used techniques do not fully capture animal grazing preferences or interactions with the wider environment e.g., herd behaviour and climate effects (Vargas et al., 2022).

As demonstrated, CH_4 emissions are associated with high uncertainty partly due to a lack of data across different systems. Data measured from ruminants under grazing conditions is particularly limited. This causes uncertainty within EFs and therefore GHG inventories and CF models. To further exemplify this, a comparison of default IPCC EFs with factors generated from local data in Sub-Saharan Africa determined that local suggested values were 56% lower than defaults for sheep (Ndao et al., 2019). Clearly, a broader understanding of the CH_4 burden of different systems will allow accuracy within inventory reporting but also in future mitigation strategies and upland management. Additionally, how variations in CH_4 output alongside variations in other GHGs affect the overall GHG burden of a particular system will further aid in future decisions.

2.6.2 Variability in nitrous oxide emissions

Quantification of N_2O emissions faces similar difficulties as CH_4 measurements. N_2O emissions occur both directly and indirectly in livestock systems (Matthews et al., 2010). Direct N_2O occurs due to application of manure or fertiliser to soil and deposition of urine and dung. Indirect N_2O arises due to atmospheric deposition of nitrogen (N), ammonia (NH_3) volatilisation as well as N leaching. Calculation of livestock N_2O emissions involves knowledge of the animal N excretion (and therefore N intake and retention) as well as use of a relevant EF dependent on the source (IPCC, 2019). Excretal N_2O is particularly relevant for lamb production systems given the animals are mostly grazed therefore deposit excreta to pasture. A key EF in these systems is therefore EF_{3prp}, the EF for direct emissions from livestock excreta.

Excretal N_2O emissions are highly variable and difficult to predict, with environmental factors such as rainfall, soil moisture and ambient temperature all having an effect (Krol et al., 2016). In addition, imperfectly drained soils resulted in higher EFs than well drained soils. The soil moisture content is influenced both by the soil texture and level of clay present. Further evidence indicates seasonal changes in EF due to variations in humidity, temperature and wind speed as well as across soil types (Broucek, 2018). The UK inventory now accounts for rainfall level when calculating fertiliser N_2O emissions, recognising the link between rainfall and N_2O emissions (Brown et al., 2022). However, this has not been applied to livestock excretal N_2O emissions within the inventory. Areas with low vegetative cover have been seen to result in higher EFs in comparison to areas with sufficient vegetative cover (Chirinda et al., 2019). Vegetation type, land topography and stocking density further influence emissions (Marsden et al., 2018). Additionally, soil pH was demonstrated as having an effect on inhibiting nitrification and therefore N_2O emissions (Marsden et al., 2019). Different factors can also interact together both temporally and spatially, for example vegetation type will vary across soil and pasture types as well as the season. In summer, plantain gave higher N_2O emissions than perennial ryegrass and white clover but not in autumn or winter. Lucerne emissions were only lower in winter (Luo et al., 2018).

The excreta type (urine or faeces) that is deposited to soil is known to result in different emissions rates, leading to recommendations to disaggregate EFs between urine and faeces (van der Weerden et al., 2020). The split of N excreted in urine in comparison to faeces is dependent on the level of protein intake of the animal and so the split will vary dependent on the animal's diet (Chadwick et al., 2018). An increase of dietary crude protein (CP) from 15.1% to 18.4% resulted in a 12% increase of N within urine but a 4% decrease in N within dung (from dietary N) as well as reduced overall N efficiency (Broderick, 2003). Dietary manipulations can have other effects as high mineral levels resulted in increased urine volume (Dijkstra et al., 2013). This then affects N excretion as higher urine volumes result in less N concentration in the urine. Given the high variability in N_2O emissions and the interactions between environmental and management factors, variation across the lamb production tiers could be significant.

Understanding the N excretion of the animal is intrinsic to accurate prediction of resulting N_2O output, as methodologies to estimate emissions within models are reliant on accuracy of the N excretion estimations. Despite current work, accuracy of calculating N excretion is still low, partly due to the natural complications of understanding ruminant digestion but also due to model methodology and limited datasets that can be used to develop models (Reed et al., 2015). Specifically, understanding feed intake and CP intake are key to improving estimates of N_2O emissions, as data regarding these factors is particularly lacking within N balance calculations (Cederberg et al., 2013). If estimates of feed and CP intake are improved, N excretion calculations will increase in accuracy. Any errors at the animal level can have a compounded effect on the final accuracy of emissions as the error will then be multiplied by the full animal population. Excretal EFs within models can be combined urine and dung but models that disaggregate emissions by excreta type can offer improved accuracy of predictions and therefore reduce model uncertainty. Datasets related to lamb production are particularly limited for this use with few replications in terms of animal numbers within the experiments and number of urinations (Marsden et al., 2020).

Limited literature exists on N_2O emission values and related EFs across the pastures and altitudes present in lamb production systems. The IPCC have recently published a refinement to the 2006 guidelines, that lowered the excretal N_2O-N EF (EF_{3prp}) for sheep from 1% to 0.3% (IPCC, 2019, 2006). A recent UK study suggests that EFs measured from sheep urine deposited in extensively managed pastures may be significantly lower than even the new revised EF. In spring, EFs of 0.02 ± 0.04% and 0.03 ± 0.09% were determined for artificial and real urine respectively. In autumn, EFs of 0.02 ± 0.04% and 0.08 ± 0.04% were determined for artificial and real urine respectively (Marsden et al., 2018). Further work by the same authors hypothesised extensive grasslands typical of lamb production in the UK would have lower N_2O-N than default EFs from both UK country-specific and IPCC perspectives. Lower emissions are attributed to organic soils having lower soil oxygen and lower pH, which may decrease nitrification and therefore N_2O (Marsden et al., 2019). Artificial urine gave EFs of 0.01 ± 0.00% in summer and 0.00 ± 0.00% in autumn. Real urine resulted in EFs of 0.01 ± 0.02% in summer. A further treatment of NO_3^- and glucose solution was applied and the results of this indicated that inhibition of nitrification was a key limiting factor for N_2O in organic soils. Disaggregation of EFs appear to have a significant effect on GHG accounting from an inventory perspective, with a 43% reduction of excretal N_2O-N found from the analysis.

Further evidence from other studies shows similar findings. Areas with increased slope had lower EFs than those on lowland in spring but this effect was not seen in urine applied in autumn (Luo et al., 2013). However, when the spring and autumn values were combined, EFs from urine on low slope (<12°) sites were 0.24% - a significantly greater value than the 0.07% emission factor recorded on medium slopes (12-25°). This is attributed due to low slopes having increased soil fertility and microbial activity compared to medium slopes. Recent work in Ireland shows negligible EFs from sheep urine and dung, with lower EFs in hill pasture than in lowland pasture, corresponding with results from other studies (combined EF -0.01 ± 0.03% in lowland and -0.03 ± 0.03% in hill) (Mancia et al., 2022b). Disaggregating EF_{3prp} by altitude has already been recommended for use in NZ inventory reporting (van der Weerden et al., 2020). From a UK perspective, some disaggregation exists between lowland, upland, and hill sheep but this is limited to animal numbers, average LW and management practices. Cattle EFs from excretal N_2O are UK-specific and disaggregated by excreta type but there are no UK-specific sheep EFs, rather the cattle EF is scaled to sheep (Brown et al, 2022). As has been shown above, EFs have been shown to vary further than the defaults allow capturing within models, with some studies reporting values even lower than the new IPCC default of 0.3%. While improvements have been made, uncertainties still exist regarding N_2O for modelling of lamb production systems. Further study is needed to quantify EFs across production areas so this can be included in inventory and CF tools as well as avoid over-generalisation between production systems. In the case of lamb production, disaggregation could be by altitude, soil type, soil pH, land slope or soil C content.

2.7 System multi-functionality and the functional unit

A functional unit (FU) is the measurement chosen by which the value derived from the function of the system being studied is quantified. For example, in livestock systems the functional unit is often kg animal LW, or kg protein produced. However, given the complexity of agricultural systems, there are often multiple functional units available to use for a system. This is where uncertainty can be introduced as the choice of FU has been found to have a considerable effect on results (Caffrey and Veal, 2013). Consideration of the study objectives is likewise significant, as this can affect which FU is most appropriate. For example, in relation to food products, FU based on the mass of a product is common. However, if the study objective is to determine nutritional value, use of a different FU such as g nutrient (e.g., calcium) may be more appropriate (McAuliffe et al., 2020). Different FU choice can affect the ranking of different products in terms of their GWP impact.

Choice of FU alongside a system or product having multiple functions introduces common problems across LCA studies. ISO 14044 standards suggest ways to counteract issues of multifunctionality within a system and that is to allocate emissions between different products (Pelletier et al., 2015). Effective allocation of emissions to co-products has implications for lamb production systems as they are often multifunctional ventures. Dependent on the region, systems can produce meat, milk, wool, or a combination of all three as well as provide wider socio-economic services (Ripoll-Bosch et al., 2013). In agricultural systems, co-product allocation has often historically been determined via economic value. However, this can cause the burden of emissions to deviate from the physical flows present in the system. For example, a study assessing poultry production whereby the manure was being sold as fuel resulted in 27.7% of the environmental burdens being allocated to the manure (Mackenzie et al., 2017). In the case of the manure having no economic value, there would be no environmental impacts associated with the manure despite the significant mass of manure leaving the system. Economic allocation for sheep system co-products resulted in differences in GWP impact at the product footprint level, with variation being between 7 and 52% (Eady et al., 2012). In comparison to biophysical allocation, the economic allocation caused a shift of the GHG burden to co-products of a high value rather than those using higher resources. Additionally, economic allocation is open to influence of economic fluctuations. There is now a shift towards biophysical allocation methodology, which allocates emissions based on physical associations amid co-products. One study developed and recommended a particular biophysical allocation, which accounts for co-product allocation based on protein requirements. The results are then related to both wool production and LW offering a method that prevents burden shifting in the system and is more stable (in terms of price changes) than a typical economic allocation method (Wiedemann et al., 2015).

To further show how this can result in inaccuracy within sheep systems, an assessment of lamb production in Spain compared three different production systems with varying levels of extensification. Initial results gave values of 19.5-25.9 kg CO_2eq kg of lamb LW^{-1} with the most extensive pasture-based system reporting the highest value compared to the zero-grazed system (Ripoll-Bosch et al., 2013). Multifunctionality within the system was then accounted for by allocating GHGs across goods (meat, wider ecosystem services e.g., biodiversity) using EU subsidy data and other economic indicators. This accounting resulted in the GHG intensity order of the systems being reversed, with the most extensive system having the lowest C footprint (13.9 kg CO_2eq kg LW^{-1}) and zero-grazed having the highest (19.5 kg CO_2eq kg LW^{-1}). These results further support developing effective accounting for multi-functionality, particularly within sheep systems. The contribution of pasture-based farming to providing wider and diverse ecosystems services should also not be ignored (Rodríguez-Ortega et al., 2014).

2.8 Carbon sequestration inclusion and other benefits

Given the high occurrence of grass-based systems within lamb production, particularly in the UK, an important aspect to consider is the impact of C sequestration on net emissions balances. Inclusion of C sequestration in CF has been assessed by various studies (Nayak et al., 2019) However, there is no standardised method for inclusion making it difficult to draw comparisons between studies. One study used four methodologies for inclusion of soil C in an LCA of Spanish sheep milk and compared three groups of farms: intensive using foreign sheep breeds, intensive using local sheep breeds and extensive grazed using local breeds. It found that a normal CF would give a range of 2.0-5.2 kg CO_2eq kg $FPCM^{-1}$ between the systems with extensive systems having a higher footprint (Batalla et al., 2015). However, when soil C sequestration was included the biggest decrease on overall C burdens was seen in extensive systems and the difference in C footprint between the systems was no longer statistically significant. Similar observations have been made in goat systems, as grazing farms with lower productivity had higher GWP impact than intensive farms with higher productivity (Gutiérrez-Peña et al., 2019). However, when including consideration for C sequestration, the lower productivity farm emissions reduced 23-26% resulting in there being no difference in footprint between the three farm types. A recent assessment of CF of lamb production in Wales showed a similar effect. Hill sheep footprints saw a reduction from 10.1 to 7.6 kg CO_2eq kg LW^{-1}, upland lamb footprints saw a decrease from 12.5 to 10.5 kg CO_2eq kg LW^{-1} and lowland lamb footprints from 11.5 to 9.5 kg CO_2eq kg LW^{-1} when sequestration in the soil was included (HCC, 2021).

This illustrates that inclusion of C sequestration can improve the outlook of extensive systems in comparison to more intensive systems. Nature-based GHG removals, such as soil C sequestration and woodland creation, represent opportunities for C offsets in the livestock sector, particularly extensive grazing. Studies have shown that sheep farms with lower stocking rates can be C neutral if sequestration is included (Doran-Browne et al., 2017). Farms with higher stocking rates were able to offset emissions with woodland creation by 48% (if 20% of the land was afforested), although were not C neutral. However, issues exist in accounting for C sequestration in CF modelling. Long-term data regarding the quantity of C stored under different land management practices is still required as well as concerns related to permanency and additionality of nature-based sequestration options (Brandão et al., 2013).

While not necessarily related to more accurately capturing environmental impacts, there remains a lack of consolidated methodology related to capturing wider system benefits within CF and LCA studies. This may skew overall views of systems and hinder devising appropriate sustainability pathways for the future. Lamb production can be associated with other ecological and social benefits. It offers provisioning services in the form of meat, milk and wool and economic benefits by supplying a source of income for otherwise marginalised mountainous areas (Marino et al., 2016). Appropriate grazing management can contribute to biodiversity objectives and soil C sequestration (Batalla et al., 2015). A variety of ecological indicators including species richness, soil microbial biomass and C/N ratio benefited from sheep grazing (Garmendia et al., 2022). However, soil compaction worsened. In addition, there is cultural significance, as sheep farming has been linked to preventing the abandoning of agricultural land and preserving rural communities (Bertolozzi-Caredio et al., 2020). Similar analyses in Finland determined benefits from sheep farming for biodiversity and biogeochemical cycles yet disbenefits towards climate change, land use and water use (Uusitalo et al., 2019). In Wales, the ecosystem services provided by upland farming, which is dominated by sheep grazing, has previously been valued at £1214.56 million $year^{-1}$ (Hardaker et al., 2020). However, ecosystem disservices are likewise derived from upland farming therefore consideration of the benefits alongside the disbenefits is necessary.

2.9 Global Warming Potential methodology

Application of Global Warming Potential (GWP) is currently used to convert different GHGs to CO_2 equivalents. This aids with comparison of gases at a wider scale. The GWP refers to the level of heat trapped by a GHG within a specified time frame, in relation to CO_2 equivalent. Historically this has largely been through use of GWP_{100}, which calculates this equivalence over a 100-year time frame. However, CH_4 is a short-lived gas, persisting in the environment for approximately 12 years. A new methodology, named GWP*, has been introduced which compares emissions of short-lived gases with CO_2 by differences in their emission level relative to singular pulse emission of CO_2 (Cain et al, 2018). Comparison of GWP_{100} and GWP* under different CH_4 emissions scenarios resulted in differences in the calculated total CO_2eq. A 25% rise in CH_4 emissions resulted in 980 tCO_2eq total under GWP_{100} versus 945 tCO_2eq under GWP*. A 10% fall in CH_4 resulted in an increased difference between the methodologies with 800 tCO_2eq total under GWP_{100} versus 0 tCO_2eq under GWP*. A 25% fall in CH_4 emissions gave a value of 735 tCO_2eq under GWP_{100}, whereas GWP* resulted in calculation of a net sink of - 420 tCO_2eq.

Ruminant livestock footprints have CH_4 as a key GHG and so any alterations to the GHG aggregation methodology have implications for future footprinting. A recent Australian study assessed the GHGs arising from Australian livestock production under both GWP_{100} and GWP* (Ridoutt, 2021). Ruminant animals were calculated as having less GHG emissions under GWP* than under $GWP_{100,}$ although for non-ruminants (pigs and chickens), altering methodologies had a limited difference. For sheep specifically, emissions related to production in 2018 were 10.3 Mt CO_2eq under GWP_{100} but -2.8 Mt CO_2eq under GWP*, showing a decreased warming effect relative to the GWP_{100} baseline. A further recent assessment of European sheep systems (from 1981 - 2018) found that when emissions are substantially lowered, under GWP* the warming effect was reversed (from 72.2 – 35.8 Mt CO_2eq) (del Prado et al., 2023). However, the same work highlighted the issues with simplifying emissions metrics, and that wider consideration should be given to the complexities of emissions reductions, rather than focussing solely on reaching Net Zero under GWP_{100}.

While GWP* offers a new approach to quantifying short-lived emissions, there are criticisms of its use and how it equates the emissions with CO_2 equivalents. A key critique is its reliance on past emissions as a baseline for comparison, which can cause issues when scaling to country or product level. There is potential disparity and inequality between developing countries with lower historical emissions in comparison to countries with higher historical emissions. A study that attempted to incorporate equality into the methodology used a *per capita* emissions baseline in place of historical emissions, which resulted in NZ CH_4 emissions (CO_2eq) *per capita* being 40x larger in comparison to standard GWP* (Rogelj and Schleussner, 2019). This led to recommendations to only use the methodology for global emissions accounting, and not at a country or footprint level. It is clear further refinement is required for the GWP* methodology. It has not yet been accepted widely in the scientific community nor by the IPCC, but still represents a considerable change to how livestock systems could be studied going forward. In a UK context, it has implications for how decisions are made relating to lamb production in the future.

2.10 Options for improvement

In general, increased statistical analysis of results will aid in improving management of model uncertainties and variability in results. This can be in relation to modelling itself but also the uncertainty underpinning direct emissions measurements. More specifically, systematic uncertainty and sensitivity analyses represent an approach available to aid in final interpretation of study results (Cucurachi et al., 2022). Various methods exist by which an analysis can be performed. A common uncertainty analysis is use of Monte Carlo simulations, which utilises random inputs (within a probability range) to better quantify the effect of uncertain parameters (Sun and Ertz, 2020). Uncertainty analysis can aid in understanding how the inherent model uncertainties affect the consistency of results therefore further aiding in decision making. To illustrate the effect of this, a study based on dairy farms applied systematic uncertainty analysis to EFs used within previously completed LCA studies to determine any changes in result interpretation (Chen and Corson, 2014). Addition of uncertainty related to EFs increased the uncertainty of impacts that were previously definitive for the farm, including climate change, acidification, and eutrophication. Inclusion of uncertainty analysis across all LCA studies can aid in result trustworthiness (Cederberg et al., 2013). Sensitivity analysis is also useful in deriving the effect of input parameters on model results and can be performed by a more targeted methodology, where selected input parameters are altered by a pre-defined value to determine effects on final results (Groen et al., 2016).

Improving transparency across LCA methodologies and results offers further opportunity to improve by sharing knowledge. Insufficient transparency of LCA methodology can result in complications when comparing different livestock production systems, or when assessing switching systems (Wiedemann et al., 2015). Increased transparency combined with wider collaboration will be essential in aiding wider understanding of the LCA techniques necessary to capture the complexity of livestock systems in the best way that is technically available. This is particularly true given the significant developments of LCA in recent years. Related to this, researchers involved in LCA must better reflect that results cannot be exact due to inherent uncertainties and therefore result ranges should be presented and interpreted effectively (Cederberg et al., 2013).

The documented high variability of in-situ CH_4 and N_2O emissions spatially and temporally represents another area for improvement (Mancia et al., 2022a; Milne et al., 2014). While capturing all variation may not be possible, disaggregation of EFs between different altitudinal pastures could improve estimates and has been recommended by various studies, particularly in relation to N_2O (Marsden et al., 2019; van der Weerden et al., 2020). Differences in CH_4 emissions have been found in sheep eating lowland pasture vs. hill pasture, therefore disaggregation may also be appropriate for CH_4 estimations (Zhao et al., 2016b). Greater data collection and understanding of emissions burdens across different systems and implementing this within C calculators would therefore be beneficial in improving CF studies of lamb production systems. Development in this area will not only improve model accuracy, but clearly wider implications can be found. In general, improving integration of C and N cycling within models, and how climatic changes interact with those would be beneficial for grazing systems (Del Prado et al., 2013). Additionally, consideration for soil C given its role as sink in grass-based systems is important. As pressure increases to abate emissions towards Net Zero, modelling tools have a key role in the evaluation of mitigation strategies that are practical and appropriate for different systems (Ouatahar et al., 2021). They likewise have importance for livestock sector industry bodies. For example, Hybu Cig Cymru (the Wales meat promotion board) have highlighted reducing Welsh meat production impacts on the environment and climate change as part of their vision for the future (Hybu Cig Cymru, 2018a).

2.11 Mitigation Interventions

2.11.1 Overview

A key component in sustainability of the livestock sector and in turn the sheep sector is application of effective and economically viable mitigation interventions to decrease the sector impacts on the environment. There is currently a lack of understanding on the efficacy of intervention options and how they interact, when considering the overall GHG budget. While there is a need for further data, key options for mitigation are consistent across a variety of studies include limiting deforestation for agricultural use, increasing efficacy of N management on farm and reducing fossil fuel dependency across supply chains (Cederberg *et al.*, 2013). Strategies that target improvement of forage quality and digestibility have been shown to be effective, with a 30% reduction in enteric CH_4 estimated by improving forage quality within American beef production (Wang et al., 2015). Improving productivity and efficiency of systems is of significance, as lamb CF was found to decrease by 30.5% if the least efficient systems improved consistent with the most efficient (Hyland et al., 2016).

An aspect that makes selection of interventions difficult for lamb production is the variability between production systems. For example, upland farms have different soil types, soil pH levels and grass species than lowland farms. It is also clear that there is high variability between farms within different production systems and methods. Mitigation interventions specific to the sheep sector have previously been summarised into three categories: improvements to farm productivity and alterations in management from both an animal and land perspective (Jones et al., 2014). Further breakdown of mitigation options can be seen in Table 2.4.

Table 2.4 Mitigation options for lamb production systems, adapted from Jones et al, 2014

Strategy	**Action**
Productivity Improvements	• Increasing reproduction rates • Improving animal finishing weight and weight gain • Reducing animal mortality • Health improvements
Diet changes	• Improvements to forage quality • Changes to what feed is provided • Addition of dietary additives to reduce CH_4 production in the rumen e.g., tannins, lipids
Pasture Management	• Changes to fertiliser application • Pasture composition and reseeding • Nitrification inhibitors • Stocking rate and grazing period

Generally, mitigation options can involve improved production methods via technological advances and farm management or via reduction of GHGs in the field, for example CH_4 inhibitors being included in diets (Honan et al., 2021) and nitrification inhibitors being used to decrease N_2O emissions (Schils et al., 2013). Reduction of GHGs can also occur simply by reducing livestock numbers. Improvements in breeding and genetics can produce animals with higher productivity levels and that are naturally low CH_4 emitters (Rowe et al., 2019). Emissions can be offset by increasing C sequestration on farm, via land management to improve soil C sequestration or woodland creation (Viglizzo et al., 2019).

2.12 Summary of mitigation options for lamb production systems

2.12.1 Forage characteristics and diet manipulation

Animal diet can be directly manipulated as an option for inhibition of both CH_4 and N_2O. This can be from the feeding of supplements or from alteration of farm management via improvements to forage quality and processing. Modification of the diet can result in alteration of methanogenesis of the rumen environment itself therefore affecting CH_4 output. Altering enteric fermentation has been identified as a highly effective option to CH_4 abatement (Haque, 2018). Diet manipulation can also affect N excretion within animal excreta and the components within urine (that may behave differently in the soil) therefore inhibiting the N_2O released (Gerber et al., 2013).

Different feed additives are being tested to determine their effect on CH_4. Additives recognised to inhibit methanogenesis by directly affecting methanogens within the rumen include 3-nitroxypropanol (3NOP), nitrates and macroalgae (due to the presence of halogenated compounds within the organisms) (Honan et al., 2021). Other substances such as lipids, plant secondary compounds and essential oils affected CH_4 production by affecting rumen conditions and therefore methanogenesis, rather than the methanogens themselves. Reductions in CH_4 output have been seen by feeding fatty acids in the form of soybean oil to growing lambs (Mao et al., 2010). However, plant compounds and essential oils have had less consistent results with some reductions but also increases being seen (Honan et al., 2021). Cattle fed cottonseed oil or tannin had a reduction in CH_4 production of 14 and 11% respectively, and a 20% decrease when fed together (S. R. O. Williams et al., 2020).

Diet manipulation can reduce the N excreted in urine and dung, either by reducing crude protein levels of animal feeds or including supplements such as condensed tannins and salts (Schils et al., 2013). Additionally, inclusion of mineral supplements can result in greater urine volume and therefore reduced N concentrations being expelled to soil (Dijkstra et al., 2013). A comparison of four studies linking diet with N excretion showed that the N excretion in urine compared to dung varied significantly. Urine N excretion was determined as 3.5 times more variable than N excretion in faeces. This represents an opportunity to use diet manipulation to alter N excretion, particularly in the case of urinary N (Weiss et al., 2009). Pastures diverse in their forage species (in comparison to typical ryegrass or clover swards) have also been found to lower urine related EFs due to their ingestion resulting in an increase of non-urea N compounds and plant secondary metabolites excreted in the urine (Gardiner et al., 2016). This is due to these compounds being less susceptible to change in the soil therefore inhibiting N_2O production. Certain plant species, such as *Brachiaria*, can also naturally inhibit production of N_2O emissions by releasing nitrification inhibitors from their roots (Subbarao et al., 2013). This has potential to reduce N_2O burdens from livestock systems if grazing these pasture types.

In the context of the lamb production sector, work has been ongoing particularly in NZ on the application of CH_4 inhibitors in farms. The key issue with relying on this as an option for lamb production is the extensive aspect of systems and difficulties with introducing feed additives. Research has therefore primarily been vaccine-based and aims to improve emission reductions by 20% (Moxey and Thomson, 2021). A study to determine N excretion from sheep used forage chemical composition, N content and digestibility to create equations showing that increases to feeding level and reductions in N concentration resulted in improved N efficiency. More N excretion was shifted to faeces than urine also, representing a reduced N_2O burden (Zhao et al., 2016a). As discussed previously, the natural species found across different lamb production pastures have been determined as influencing both CH_4 and N_2O emissions therefore diet represents a mitigation option going forward from both natural pasture management and dietary additive perspectives.

2.12.2 Genetics

Genetics have long presented a solution to optimising desirable traits within a particular species. For example, the Dutch national cattle breeding programme includes 15 traits related to milk production, overall health, lifespan, and reproduction (de Haas et al., 2021). Discussion has now moved to breeding for “low emitters” – animals that may have naturally lower CH_4 output or improved N utilisation. CH_4 production (and N utilisation) by animals are genetic traits that could be included in breeding programmes. However, this does involve having a sufficient number of animals genotype to allow reliable predictions to be made on genetic inheritance.

The application of genetics to lamb production systems has not yet been significantly explored but does represent a future opportunity for mitigation. Particularly for pasture-based systems as it would remove difficulties related to distribution of diet related GHG inhibitors. There are still limits to the knowledge including how low emitting sheep are linked to faecal output and N excretion and how reduced CH_4 in the eructate and flatulence may affect CH_4 emission from manure. NZ has implemented a government led programme to assess breeding low-GHG sheep and an initial study determined that overall GHGs were lower for low CH_4 emitting sheep in comparison to high emitting sheep (Jonker et al., 2019). This effect was seen even when other GHGs (N_2O from excreta) were accounted for, indicating that improving genetics may still be an option for the future. A long-term (10 year) breeding experiment in NZ resulted in sheep with 10-12% reduced enteric CH_4 emissions when low CH_4 was selected for (Rowe et al., 2019). Additionally, genetics provides an opportunity to selectively breed for other improvements related to productivity such as feed utilisation, greater finishing LW, growth rate and reductions in lamb mortality (Alcock and Hegarty, 2006).

2.12.3 On-farm productivity improvements and farm management

Improvements to productivity are key in reducing GHGs from livestock systems (Hyland et al., 2016b). An assessment of the Scottish sheep farming sector maintained that if productivity could be increased by 17%, the number of breeding animals could then be decreased by 10% causing an overall reduction of 0.05Mt CO_2eq in the GHG inventory (Moxey and Thomson, 2021). A Welsh based study gave a high range of CF values across different farms in lowland, upland and hill areas directly linked to productivity differences (Jones et al., 2014a). In this study, the hill farms with the lowest CF were lower than the mean lowland and upland farms. This highlights that improved productivity is possible for farms in marginal areas. The most influential variables that affected the CF were determined as the number of lambs reared per ewe and the lamb growth rate (g day^{-1}). The number of ewes and replacements not mated, and the use of concentrates were also significant variables. The use of additional feeds and fertilisers can finish lambs more quickly and to a greater weight but may also negate the C benefit due to the inherent C intensity of producing animal feeds. Effective accounting of productivity measures is also helpful in terms of communication with farmers as measuring input vs output efficiency of a system is a clear way of benchmarking (Halberg et al., 2005).

Improving on-farm healthcare of sheep can improve productivity by increasing the number of productive animals in the flock as well as improving animal welfare and giving GHG savings. Improvements to sheep health were determined as resulting in 15 kg CO_2eq reduction per animal per year and saving 36 pence per animal (Eory et al., 2020). Having a 50% improvement in sheep health was calculated to save 484 kt CO_2eq emissions within the UK inventory by 2035. Sustainable pasture management options such as restriction of grazing and lowering animal stocking rates can provide additional benefits as GHG burdens are reduced and farm profitability increases. Other farm productivity improvement options related to the animals themselves include selective breeding to increase litter size, lambing ewes as yearlings rather than 2 years old and introducing creep feeding to finish lambs quicker (Alcock and Hegarty, 2006).

While moving to extensive farming systems does not guarantee an improvement to farm productivity, these systems help to accentuate the ecosystem services provided by the uplands and offer a way forward for increased farm sustainability (Ripoll-Bosch et al., 2013). Intensive systems are often correlated with improved productivity therefore reduced GHG intensity per unit product but extensive systems offer unique characteristics that can aid in GHG mitigation, for example their interaction with agroforestry and opportunities for C sequestration (Eldesouky et al., 2018). However, product output from these systems is generally lower than for more intensive systems, so this should be considered in the context of efficient land use in the future. Previous evidence shows that extensification of upland farming systems can be productive and sustainable over a 10 year time period with lamb output per ha, ewe body condition and sheep LW not decreasing in trials (Barthram et al., 2002).

Additives to the soil also represent options to decrease GHG burdens from livestock. These include nitrification inhibitors and slow-release fertilisers, both of which affect the uptake of N to plants. One nitrification inhibitor, (Dicyandiamide (DCD)) has been widely investigated and in one study gave an average 46% decrease in N_2O EF from cattle urine (Chadwick et al., 2018). However, factors such as temperature, pH, moisture content, soil organic matter and clay content all affect the efficacy of DCD (McGeough et al., 2016). For lamb production systems and given the environmental differences across upland and hill areas, it is not clear if use of inhibitors is economically viable or effective enough. It may be applicable for some areas of lamb production systems but also limited across the different pastures involved on farms. The inhibitor 3,4-dimethylpyrazole phosphate (DMPP) was deemed ineffective for inhibiting N_2O emissions from sheep urine (Marsden et al., 2017). Slow-release fertilisers can reduce overall N losses and prevent large N_2O outputs, particularly after periods of heavy rainfall. However, these fertilisers are high cost and therefore economically restricted (Ball et al., 2004). They are also applicable only to some areas of lamb production systems e.g., lowland, and improved upland pastures.

2.13 Issues with implementation of mitigation strategies

While there are a multitude of mitigation options suggested for livestock systems, it is still unclear what economic implications these may have. Marginal abatement cost curves (MACCs) attempt to rank mitigation options while considering costs of implementation. These have been developed for UK agriculture (Eory et al., 2015). Due to economic limitations, physical constraints and system trade-offs, less than 10% of what is technically feasible has been determined as realistically feasible under current economic conditions (Herrero et al., 2016). The practicality of measures represents a significant barrier as measures that are highly effective but less practical and prohibitive economically are less likely to be implemented by farmers. This is especially true without government support. Emissions reductions are particularly difficult in the agricultural sector due to the variety of systems and options having to be implementable across this diversity. Expert knowledge related to the options that are not preferable is more thorough than knowledge related to options that are best suited for systems, making consensus difficult to gain. Inclusion of farmer opinion can aid in improving this barrier (Jones et al., 2013).

It is important to consider what intervention is best placed for a particular system and one that suits a particular system may not suit another. Interventions must be contemplated from an LCA perspective, considering wider possible consequences. For example, ensuring an intervention such as a diet change to reduce CH_4 emissions in the rumen does not result in increased GHGs overall due to production and transport of the recommended diet (Garnett, 2009), or a shift of excreted N into the urine where there would be greater risk of N_2O emission. Additionally, socio-economic aspects and the reliance of developing communities on livestock products should be considered. Low- and middle-income countries produce the majority of ruminant related GHGs globally, yet any mitigation options suggested so far are aimed at developed nations. There is a lack of studies currently assessing the social, economic, and environmental side of mitigation options simultaneously (Harrison et al., 2021). Limited studies exist that are focussed on mitigation options for lamb production systems with the economic and social impacts included. Clearly, balancing GHG mitigation with delivering continuing food security is intrinsic to future strategies surrounding livestock production.

2.14 Knowledge gaps and opportunities for improvement

Lamb production remains a significant industry within UK and Welsh agricultural sectors. However, negative environmental impacts occur as a result of production with a key impact being the production of GHG emissions (namely enteric CH_4, excretal N_2O and CO_2 from energy use on farm). Given the expected increase in global population, agricultural systems must look to improve efficiencies and reduce their impact on the environment. Within the UK policy context, a target to reach Net Zero GHG emissions by 2050 is in place. The lamb production sector therefore has a part to play in reducing emissions. At the core of reducing emissions, is the quantification of emissions. LCA and CF represent methodologies that can evaluate the environmental burdens related to a product through the stages of its life cycle, with CF focussing on the C impact. However, various uncertainties exist within the methodology that affect robust quantification of emissions burdens from lamb production systems. Additionally, it is a sector that has received less study than other ruminant systems such as beef and dairy.

As lamb production occurs across a variety of pastures of differing altitudes, evidence is beginning to show emissions changes across said pastures. Enteric CH_4 has been found to vary depending on the animal's diet, in terms of feed intake and forage quality. Some plant species also contain secondary compounds that affect enteric CH_4 output. Excretal N_2O emissions have been found to decrease when moving from lowland to upland pastures, with values below default EFs being measured in these pastures. These emissions variations have implications for CF methodology, in terms of which parameters to use (e.g., forage digestibility) and appropriate EFs for different grazing systems. Additionally, the choice of FU and the inherent multifunctionality of lamb production systems presents complication within accounting methodology. Other factors that affect results include the choice of GWP value, particularly in the context of discussion around the short life of CH_4, as well as inclusion of wider system benefits and disbenefits.

Lamb production may also provide wider ecosystem services, for example biodiversity improvement and C sequestration if under the appropriate management. Social and cultural benefits exist in the form of income and maintaining rural communities. Assessment of these may therefore aid in providing a more holistic assessment of these systems. As well as accurate quantification of emissions, it is equally significant to consider strategies to reduce emissions. Strategies to reduce emissions cover a variety of actions and target different gases. Generally, they involve either directly reducing emissions e.g., by use of dietary additives to reduce enteric CH_4 or improving forage quality, or by improving farm efficiency, reducing reliance on fossil fuels, and promoting C sequestration on farms via management to improve soil C storage or introduction of trees. However, study of mitigation options in the context of altitudinal lamb production systems is limited. Priorities for improvement of assessment of these systems are therefore to improve understanding of emissions changes across altitudes and generate further data to improve robustness of estimates, particularly in relation to enteric CH_4 and N_2O. Integration of these changes in emissions within CF methodology, for example by disaggregating EFs by altitude or pasture type will aid in more accurately assessing these systems, particularly for upland and hill grazing. Finally, consideration of altitudinal changes when assessing mitigation strategies appropriate for lamb production systems will aid in further understanding emissions reductions options and aid in promoting long-term sustainability.

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Chapter 3 Greenhouse gas emissions from sheep grazing at different altitudes in a lamb production system

3.1 Introduction

The key greenhouse gases (GHGs) relevant to lamb production systems are methane (CH_4) and nitrous oxide (N_2O). CH_4 arises due to sheep being ruminants, therefore have a rumen as part of their digestive system to aid in digesting coarse plant materials. The gas is produced as a by-product of this process (Hopkins and Lobley, 2009). Additionally a proportion of CH_4 arises from animal manure, during storage or after deposition to pasture (IPCC, 2019). N_2O gas arises in livestock systems due to manure management and the deposition of urine/dung to pasture as well as fertiliser use and manure spreading for feed crops (Cardenas et al., 2019; Chadwick et al., 2018). Within lamb carbon footprints (CF), CH_4 from enteric fermentation is generally responsible for the largest portion of the overall footprint, followed by N_2O from applied and deposited manure. A recent review found in general between 50 and 75% of lamb production GHGs arose from enteric fermentation, 18-25% from excreta deposition (mostly N_2O but additionally some CH_4) and 10-15% from fertiliser and feed production (Bhatt and Abbassi, 2021). Manure related emissions are often lesser in lamb production systems due to them being predominantly grazed and therefore excreta are mostly deposited direct to pasture. Lamb production systems also result in CO_2 emissions from the production of feed and energy use on farm, but are generally a smaller proportion of the overall footprint (Jones et al., 2014). Globally, the demand for ruminant produce e.g., meat and milk are rising and projected to rise further (Henchion et al., 2021), which has been attributed to population and income increases. Additionally, they are culturally significant and have a role to play in economic security for significant numbers of communities across the world (Batalla et al., 2015). It is therefore essential to accurately understand their contribution to GHGs across a variety of production systems and how sustainability can be achieved within these (Mottet et al., 2018).

Small ruminants, particularly sheep in the UK, are generally grazed on a variety of mixed sward pastures of varying digestibility and environmental conditions. In the UK, this variety occurs due to the altitudinal aspects of sheep production, whereby animals are grazed on hill and upland areas unsuitable for arable crops and finished on lowland pastures (National Sheep Association, 2021; Rodríguez-Ortega et al., 2017). Studies have shown variability in enteric CH_4 output dependent on different environmental and dietary factors. For example, a link between the digestibility of forage and any concentrates consumed by the animal and the resulting CH_4 emissions has been determined (Hegarty et al., 2010). Differences in CH_4 emissions have been observed from animals eating different grasses, for example lambs fed winter forage rape had enteric CH_4 emission rates 22-30% lower than those fed perennial ryegrass across all trials (Sun et al., 2015). Indeed, mixed diets using plants with high levels of secondary compounds, e.g., polyphenolic compounds including tannins, to reduce methanogenesis in the rumen and therefore CH_4 are highlighted as a solution to reducing ruminant CH_4 (Provenza et al., 2019). This is potentially relevant for lamb production systems, as animals are often grazed in areas with a proportion of ericaceous species such as *Calluna vulgaris*. These plant species contain a high proportion of phenolic secondary compounds (Ropiak et al., 2016).

Full quantification of the variability of enteric CH_4 across the altitudinal gradient of lamb production systems and the implications that these differences may have on the resulting CF has not yet been achieved. At the time of writing, there exists limited published works quantifying enteric CH_4 emissions from sheep grazing different grasses and exploring how this may affect the final footprint. Studies with cattle have shown that cattle grazed on semi-improved upland areas had substantially lower dry matter intake per day, CH_4 emission output and bodyweight gain on average than cattle on improved lowland grasslands (Richmond et al., 2015). However, when expressed on a LW gain basis, the cattle on upland grazing had a higher mean daily enteric CH_4 output. This could result in the lower absolute CH_4 emission being offset by the longer finishing time required on upland pastures, therefore affecting the final footprint. Additionally, the wider contribution of ruminants to extensive grassland management e.g., biodiversity and carbon (C) sequestration should be included in any analysis to determine a holistic understanding, but comprehension of these benefits is not yet well developed.

A similar trend can be seen with N_2O emissions in that they are highly variable dependent on different environmental factors. N_2O emissions from urine and dung patches have been linked with rainfall and temperature as well as soil moisture deficit and soil type (Krol et al., 2016). Variations in nitrification and denitrification rates in the soil affect final N_2O cumulative fluxes, which is likewise affected by environmental factors (Bell et al., 2015). Thus far, some studies have shown variation in the N_2O produced between intensively managed lowland and extensively managed upland and hill areas. In an upland grassland, N_2O emission factor (EF) (EF_{3prp}) values of 0.03 ± 0.09% from real sheep urine were determined in spring and values of 0.08 ± 0.04% in the autumn (Marsden et al., 2018). For context, the current Intergovernmental Panel on Climate Change (IPCC) default EF is 0.3% for N_2O related to N deposition on soil from excreta deposited by grazing sheep (IPCC, 2019). This value is the same across all pastures and not disaggregated e.g., by altitude or soil pH. The UK employs country specific EFs for cattle but not for sheep due to limited data specific to the UK. Instead, 50% of the cattle value is used in line with IPCC 2006 guidelines and the resulting values are close to the IPCC 2019 value but also have no disaggregation (0.315% for urine and 0.097% for dung) (Brown et al., 2022; IPCC, 2006). Further work completed in a hill area derived EF_{3prp} values of 0.01 ± 0.00% (summer application) and 0.00 ± 0.00% (autumn application) for artificial sheep urine. An EF of 0.01 ± 0.02% was determined for real sheep urine (Marsden et al., 2019). These lower values were linked to the organic soils with low soil pH inhibiting nitrification. This is further supported by additional data from New Zealand, which supports disaggregation of EFs for sheep and cattle excreta by altitude, accounting for both urine/dung split and the pasture terrain (Kelliher et al., 2014; van der Weerden et al., 2020). A review of ruminant excretal N_2O emissions found a range of techniques available to quantify emissions with sheep excretal emissions having increased uncertainty in comparison to cattle (Mancia et al., 2022b). Overall, further data collection to increase robustness of data and disaggregation of EFs was recommended to improve estimates. These findings highlight a necessity to further generate site-specific altitudinal EFs for both N_2O and CH_4, particularly in support of the sheep industry's sustainability targets.

In the context of livestock CFs, the EFs mentioned above are used within calculations to determine the CF of a particular system. IPCC methodologies include a default value for N_2O emissions from the deposition of grazing animal excreta to pasture (IPCC, 2019). For ruminant CH_4, a value known as the CH_4 conversion factor is used (Ym), which describes the percentage of gross energy intake by the animal that is released as CH_4 (IPCC, 2019). Therefore, generation of site-specific EFs will also aid in more accurate CFs, by extension allowing site-specific C mitigation scenarios to be explored in better detail.

With the aim of filling this information gap and providing baseline data for LCA modelling later in this thesis (Chapters 4 and 5), this study aimed to generate excretal N_2O and ruminant CH_4 EFs from sheep grazing pastures at different altitudes. The pastures were located at Bangor University's Centre for Hill and Upland Management (CHUM), Abergwyngregyn, North Wales. Measurements were made from Welsh Mountain ewes, chosen for their widespread use in the local area. To measure enteric CH_4, a short-term methane measurement system known as GreenFeed (GF) (C-Lock Inc., Rapid City, South Dakota, USA) was used across the altitudinal gradient, whilst N_2O emissions were made from urine patches and dung pats created from excreta collected from the same ewes using manual static chambers. These measured emissions were used alongside data from published literature to assess the benefits of introducing an altitudinal aspect to sheep production CFs and GHG inventories in the future.

3.2 Methodology

3.2.1 Enteric methane emission quantification from sheep

3.2.1.1 Field site

The field sites for all measurements were based at CHUM, Abergwyngregyn, North Wales (53 ° 140N, 4 ° 010W). The land base covers 252 hectares across an altitudinal gradient, with pastures starting at just above sea level through to upland (ca. 250m asl) and hill pastures (ca. 700m asl). There were three study sites selected for use, with measurements made between 2020 and 2021, beginning in the lowland pasture in spring, and moving through to upland (late summer) or hill pasture (early winter) to simulate typical animal movement management on the farm. Additionally, starting in the lowland allowed easier access (as well as connection to mains power) to the Greenfeed small ruminant methane measurement system (GF) (C-Lock Inc., Rapid City, South Dakota, USA) to ensure correct functioning before moving to the more remote areas where the GF was powered by a generator used to charge batteries (as well as solar panels).

The initial intention was to repeat measurements across all pastures in the second year. However due to COVID-19 affecting facility access combined with technical issues and the limited time period of a PhD studentship, repeat measurements were only achieved in the lowland pasture. Details of each of the pastures can be found in Table 3.1.

Table 3.1 Enteric CH_4 emission measurements from sheep: pasture locations and site characteristics

Site	Experiment Dates	Site characteristics
Lowland	29/05/20-22/06/20 & 08/06/21 – 07/07/21	Managed lowland pasture (<50m a.s.l), seeded with *a Lolium perenne* mix (including *Trifolium repens*) and receiving moderate nitrogen fertiliser applications. The soil classification is Eutric Cambisol. The 2020 measurements were from twenty animals (22 animals ha^{-1}) and the 2021 measurements were from 8 animals (30 animals ha^{-1}).
Upland	28/10/20 - 01/12/20	Semi-improved upland pasture (>320m a.s.l), comprising mostly of British NVC classification MG6 (*Lolium perenne – cynosurus cristatus*) alongside a variety of herb species such as *Trifolium repens* and *Achillea millifolium.* The soil classification is Orthic Podzol. The measurements were from twenty animals (17 animals ha^{-1}).
Hill	03/09/21 – 30/09/21	Enclosed paddock of hill grazing (approx. 259m^2 in size, grazing 8 animals*) (53°22′N, 3°95′W) in Wales, UK on the Carneddau mountain range (>700m a.s.l). The area is common grazing land, used by graziers between the months of April and October. The pasture vegetation present is NVC classification H12 (*Calluna vulgaris, Vaccinium myrtillus* heathland). Soil types include Dystric Histosol and Humic Gleysol. The measurements were made from eight animals in total (initially the same eight from the lowland pasture, however four escaped from the paddock and were replaced with four new animals).

*The number of animals in the hill enclosure was greater than the average stocking density for a hill pasture, however this was the only available space to contain animals close to the GF and allow sufficient measurements to be made.

During 2020, both the lowland and upland measurements were made from the same group of twenty Welsh Mountain (barren) ewes (>1 year old and an average LW of 29.6 kg). For 2021, eight ewes in total (from the original group of twenty) were involved in the lowland and hill measurements (average LW of 37.7 kg). A smaller number were selected for 2021 due to the limited size of paddock available for measurements on the hill pasture. During the hill measurements, four of the eight ewes escaped the paddock, so were replaced with new animals from the wider flock.

In all sites, the animals were allowed free range of the pasture. In the upland after two weeks of measurements, animals were still not visiting the unit as frequently as was necessary. Excess availability of pasture was determined to be the cause, therefore electric fencing was installed to decrease the pasture size and availability of grazing to approximately 1/3 of the original pasture area.

3.2.1.2 Enteric methane emission measurements

The CH_4 emitted from grazing sheep was measured through use of the GF, an automated head-chamber system used to measure CH_4, and CO_2 mass fluxes that are present in ruminant animal breath. The unit measures CH_4 and CO_2 emissions through use of a nondispersive near-infrared analyser, which reports emissions every second. When an animal inserts its head into the hood of the unit, the RFID recognises the animal number and releases a specified number of bait feed pellets for a specific time (pre-set prior to the measurements beginning) (Hristov et al., 2015). CH_4 data are then automatically sent from the GF system to C-Lock for processing, who then provide processed time-stamped flux data per animal.

For all pastures, the unit was set up as per manufacturer guidelines with the unit placed on top of a wooden pallet (measured to size) at an accessible height for the sheep to eat from the hood easily. Additionally, standard sheep hurdles were set up in front of the unit to create an alleyway. This ensured only one animal visited at a time, reducing subsequent negative effects on CH_4 emission measurements (as seen in Figure 3.1). Standard maintenance on the unit, including air filter changes and gas calibrations, were performed throughout the measurement period as recommended by the manufacturers.

Figure 3.1 GreenFeed system set up in a lowland pasture, May 2020 (Photo: H Riddell)

For the lowland measurements, power was supplied to the unit via a mains connection in the field. In the upland and hill plots, three 12V 200Ah batteries were charged via a 6000E Silence generator (SDMO, Brest, France) installed with a battery voltage sensor (BVS) (GenControl, UK). This meant that when the battery voltages dropped below 11.8V, the BVS would cause the generator to start, therefore recharging the batteries. It was also possible to charge the batteries using two large solar panels (as seen in Photo 3.1), but the decision to use mains and a generator was made to maintain power supply across weather conditions. To facilitate data upload, the unit had a pre-loaded data sim inserted using the UK network 3 (Hutchison 3G UK Ltd, Berkshire, UK). Data were uploaded every hour via the mobile network connection.

For these measurements, the GF system was set to deliver each animal 5 drops of bait feed every 36 seconds, at a maximum of 4 times per each 24-hour period. This ensured the animals remained within the unit for a minimum of 3 minutes, increasing reliability of the CH_4 measurement. Additionally, it encouraged the animals to visit at different times within a 24-hour period reducing the effect of any diurnal CH_4 variations on the final results. The bait feed selected for the measurements was a small fibre pellet (<7mm in size) with the brand name of Allen and Page Fast Fibre® (Allen and Page, Norfolk, UK). The estimated digestible energy (DE) of the pellet is 8.0 MJ kg^{-1}. Further nutritional characteristics of the pellet can be seen in Table 3.2.

Table 3.2 Nutritional composition of bait feed used in experiments (data presented on a dry weight basis, and provided by the supplier on the bag*)*

Composition factor	**% Present**
Dry matter	91
Crude protein	7.5
Crude ash	9.5
Crude fibre	26
Total sugar	2.5
Starch	5.0
Omega 3	0.5
Crude oils and fats	3.0
Calcium	1.0
Sodium	0.8

The bait feed used is originally intended for use with horses that require a low-calorie diet. However, for this study it was selected due to it having lower energy and crude protein than a generic sheep feed. This selection was intended to decrease the impact of the bait feed on the resulting ruminant CH_4 emissions, allowing the focus of any differences to be on the forage consumed while grazing.

At each of the pastures, the target was to obtain a minimum of 200 ruminant CH_4 measurements across all the animals, so the duration of the measurements at each pasture reflected this. This was alongside a goal of obtaining a minimum 20 measurements per animal. This was in line with manufacturer advice and allowed sufficient CH_4 data to be collected that could be analysed across the whole population or to explore between and within animal variance. Length of experiment therefore varied, although all lasted between 25 and 34 measurement days.

3.2.1.3 Animal selection and training

Twenty-one Welsh Mountain ewes were initially selected; however, one ewe was discovered to be pregnant and removed from the group, leaving twenty. During the 2021 hill measurements, a total of eight ewes were used as the paddock available for use was deemed insufficient in size for a greater number of animals. This breed of sheep was selected due to it being a representative breed within the local area and common within grazing in upland areas.

To familiarise the ewes and train them to use the GF unit, the ewes were retained in a livestock building at CHUM. During the training period, the system was set to give the animals a drop of bait feed every 15 seconds (for 2 minutes) to entice them to use the feed hood more regularly. This rate of bait drops was gradually lowered until the feed drop number and time interval used for the trials was reached. The training period was conducted until all ewes were visiting the GF regularly throughout the day, which took approximately three weeks.

3.2.1.4 Animal liveweight

The liveweight (LW) of each ewe was recorded on the first and last days of each GF CH_4 measurement period at each pasture. This allowed any weight gain or loss to be recorded and aided in calculation of dry matter intake (DMI) within net energy calculations (as described in section 3.2.1.5).

3.2.1.5 Forage characteristics and dry matter intake determination

Samples (n=5) of the forage present in each pasture were taken. To capture variability as precisely as possible across the pasture and due to their size, the lowland and upland pastures were split into five sections. Five samples were then taken from each section, homogenised, and sent as a single sample for analysis. This resulted in 5 homogenised samples in total being analysed for both pastures. In the hill site, five samples total were taken across the whole pasture as it was a smaller area. Forage samples were collected using shears to cut the pasture to a length necessary to simulate sheep grazing (4-8cm). For most pastures, forage sampling was conducted between the second and third weeks of measurements, dependent on the number of useable CH_4 emission measurements (as confirmed by C-Lock). This differed in the upland site where two samples were taken. This was due to the grass quality visibly decreasing due to inclement weather conditions as well as the area being made smaller to encourage the animals to visit the unit. The decision to sample again was therefore taken. Fresh pasture samples were analysed by Sciantec Analytical (Cawood Scientific Ltd., North Yorkshire, UK). Factors analysed were dry matter (g kg^{-1}), crude protein (g kg^{-1}), D value (digestibility), ME (MJ kg^{-1}), NDF (g kg^{-1}), ash (g kg^{-1}), oil- A (g kg^{-1}), sugar (g kg^{-1}), nitrate nitrogen % and buffering capacity (meq kg^{-1}).

To determine DMI and GE requirements of the animal, the net energy calculations to estimate feed intake, as described in the IPCC 2019 guidelines were used. The calculations were used alongside site specific data such as animal LW and pasture characteristics e.g., digestibility to calculate the expected DMI of each animal. This in turn allowed calculation of the Ym, also described as the proportion of CH_4 emitted per unit gross energy intake (GEI), from animals grazing in each pasture. The calculations were generally applied as dictated within the IPCC guidelines; however, some elements were modified to adhere to the measurement periods (rather than annually).

3.2.2 Excretal nitrous oxide emission quantification

3.2.2.1 Field site

Urine and dung collection as well as application to the pasture and the subsequent measurements took place in the same lowland and upland pastures as described previously in Table 3.1. Urine and dung were not collected from animals grazing the hill pasture because of time constraints. An area of each pasture was fenced off 4 months prior to the measurements beginning to exclude any livestock excretal deposition occurring and affecting the N_2O fluxes from the urine and dung applications. In the lowland pasture, N_2O sampling commenced on 13/08/20 and continued until 15/02/21. In the upland pasture, sampling commenced on 23/09/20 and continued until 09/04/21.

3.2.2.2 Animal excreta collection and measurements

Prior to excreta collection, the ‘sheep visit’ data from the GF unit were studied and the 12 ewes selected for excreta collection were those who had visited the unit most. Real sheep urine and dung were collected by housing individual Welsh Mountain Ewes (n=12) in purpose built collection pens for a period of 6 hours as shown in Figure 3.2 and described in Marsden et al. (2017). The pens have slatted flooring with a tray underneath allowing urine samples to be collected when an excretion event occurred. Wire mesh covered with muslin cloth was placed on top of the collection trays to prevent dung samples contaminating urine samples. This system also allowed easier collection of dung samples. While in the pens, the animals were provided with water and forage cut from the surrounding pasture. Ethical approval from Bangor University’s College of Natural Sciences Ethics Committee was obtained for this study (approval number COESE2019HR01).

Figure 3.2 Urine collection pens in the lowland site (Photo H Riddell)

Excreta collection occurred on 30/07/20 in the lowland site and 16/09/20 in the upland site. Once collected, the urine and dung samples were weighed, and a record of each sample weight was taken. Subsamples were taken from all urine and dung samples for subsequent analysis. For the lowland site, the samples were immediately frozen until application (application was initially delayed due to COVID-19) but at the upland site samples were kept refrigerated for a week before application.

The urine and dung characteristics at each site were determined. Subsamples for all urine events were taken and the remainder filtered and bulked. The dry matter % of the dung was determined by oven-drying at 80°C for 24hrs. Organic matter content was obtained by igniting samples in a muffle furnace at 450 °C for 16hr. Total nitrogen (N) and C content within the urine was determined by using a Multi N/C 2100S analyser (AnalytikJena, Jena, Germany). Urine samples were diluted by 500x and 1000x before analysis. It was not possible to analyse the dung samples for total N content, therefore a literature value was used in calculation of the EFs (see Table 3.11).

3.2.2.3 Soil characteristics

The soil characteristics present in each pasture were determined (n = 18 at approximately 10cm depth) and can be seen in Table 3.3. Soil samples were collected on the day that excreta samples were applied to the pasture (11/08/20 in the lowland site and 28/09/20 in the upland site) and pH and EC as well as gravimetric soil moisture and organic matter content were assessed within 24 hours of sampling. The soil water-filled pore space (WFPS) % was then determined, by calculating the ratio between soil porosity and the volumetric water content.

To determine bulk density, 100cm^3 cores (0-10cm depth) were pushed fully into the soil to obtain a soil core. The cores were then oven-dried (at 105 °C for 24 hours) and once dried, they were sieved to remove any stones. The gravimetric soil moisture content was obtained by drying soil samples in an oven at 105 °C for 24hrs and weighing before and after drying. Organic matter content was obtained by igniting samples in a muffle furnace at 450 °C for 16hr. To determine pH and electrical conductivity (EC), electrodes were inserted into fresh soil samples mixed with distilled water at a 1:2.5 (w/v) ratio. It was not possible to analyse for total C and N (%), therefore literature values measured at the same site were utilised (Marsden et al., 2019, 2018, 2016).

Table 3.3 Soil characteristics (0-10cm) obtained at each site (n=18, ± standard error of the mean (SEM)). Data expressed on a dry weight basis. Data for hill pasture taken from Marsden et al., (2019).

Soil characteristics	**Lowland**	**Upland**	**Hill***	
			Summer	Autumn
Soil Type	Eutric Cambisol	Orthic Podzol	Dystric Histosol	Humic Gleysol
Bulk density (g cm^{-3})	0.86 ± 0.03	0.57 ± 0.02	0.33 ± 0.05	0.40 ± 0.04
Gravimetric moisture content (%)	28.5 ± 0.48	42.9 ± 0.35	222 ± 37	88 ± 6
Total C (%)	5.47 ± 0.77*	73.3 ± 7.0**	24.9 ± 4.6	7.7 ± 0.5
Total N (%)	0.42 ± 0.05*	5.8 ± 0.5**	1.39 ± 0.24	2.05 ± 0.04
Organic matter (%)	7.89 ± 0.25	17.05 ± 0.65	47.2 ± 8.0	14.7 ± 1.8
pH	6.41 ± 0.03	5.47 ± 0.07	4.44 ± 0.06	4.36 ± 0.04
EC (mS cm^{-1})	140.04 ± 2.41	113.67 ± 19.95	0.036 ± 0.02	0.059 ± 0.03

*Values taken from (Marsden et al., 2016), **Values taken from (Marsden et al., 2018), ***Values taken from (Marsden et al., 2019).

3.2.2.6 Nitrous oxide sampling experimental set-up

To allow measurement of N_2O fluxes, the closed static chamber technique was employed in both pastures (as detailed in Chadwick et al., 2014). The static chambers used for these measurements were designed specifically for sheep urination volume. Polypipe ® polyvinyl chloride circular piping was selected as appropriate material and then cut to 25cm sections, which would allow minimum 10cm insertion to the soil while leaving sufficient headspace for sampling. The pipe was 200mm wide, to account for sheep urine patch sizes.

Chamber lids in the form of tapered polyethylene plugs were purchased from Essentra Components® and were matched size with the piping allowing creation of an airtight seal when pushed onto the chambers at the times of gas sampling. The pipes were lined with

RS PRO 4mm thick closed-cell black foam tape supplied by RS Components®, to create a tight seal and prevent release of N_2O to the air during sampling. Each chamber lid was equipped with a Suba-seal (Sigma, Gillingham, UK) allowing manual gas sampling to occur. Each chamber was inserted 10cm into the soil, at least 1 week before sampling occurring allowing any disturbance of the soil to settle. The height of the chamber above the soil (used for determining the volume of the chamber headspace and in subsequent N_2O flux calculations) was recorded. The chambers inserted into the measurement area (in the lowland site) can be seen in Figure 3.3.

Figure 3.3 Measurement chambers installed in lowland site for excretal nitrous oxide experiment (Photo: H Riddell)

Both experimental sites had 18 plots with urine (n=6), dung (n=6) and control (n=6) replicates. The 18 plots were arranged in 3 rows of 6 and the allocation of replicates was determined via a randomised block design (Figure 3.4). Each plot consisted of one sampling chamber and soil samples were taken from the surrounding area of the chamber. Gravimetric soil moisture content was determined from these samples and measured alongside the N_2O measurement, due to the known effect of soil moisture content on resulting emissions (Krol et al., 2016). Samples were taken out with the chamber as there was not sufficient area to take multiple soil samples over the measurement period, and to avoid soil disturbance within the chamber and any effect on emissions. Soil mineral N measurements were not made, as resources were focussed on N_2O flux measurements to generate pasture specific N_2O EFs.

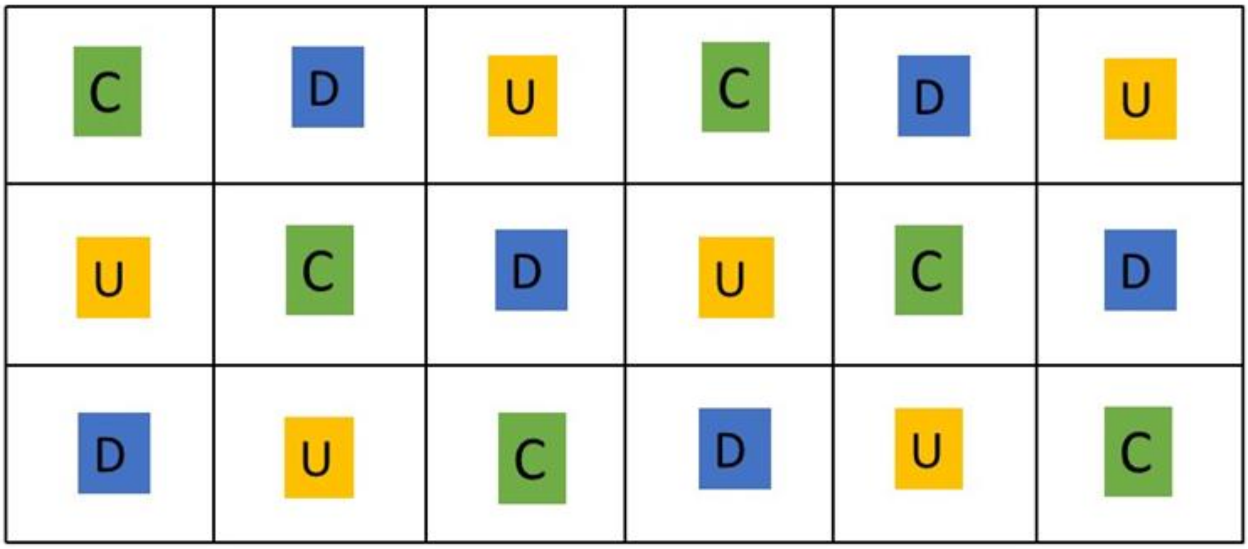

Figure 3.4 Example distribution of treatments and replicates at each pasture

Prior to application at both sites, both the urine was filtered, and the urine and dung samples were bulked separately, mixed, and portioned ready for application to the grassland. The volume and weight of urine and dung to be applied was determined by calculating the mean volume of urination and weight of dung events that occurred during the collection period. This resulted in 215ml of urine being applied to each urine plot and 57g of faeces to each dung plot in the lowland. In the upland, 185ml of urine was applied to each urine plot and 64g of faeces to each dung plot. This mean volume and weights were used to ensure sufficient volumes for uniform application weights and reduce any uncertainty related to variation in individual sheep urine composition and urination volumes.

Once portioned, urine samples were subsequently bottled, and dung samples bagged. On the day of application, portions of urine and dung were applied within the respective sampling chamber by hand. Samples were then left for a minimum period of 3 hours before gas sampling occurred. The application dates of both urine and dung samples were 17/08/20 in the lowland site and 28/09/20 in the upland site.

3.2.3 Nitrous oxide emissions sampling

3.2.3.1 Sampling frequency

Manual gas sampling occurred 2x in the week before application, allowing determination of background N_2O fluxes for each chamber. After application, the sampling occurred at an increased frequency in the initial weeks before gradually decreasing. The sampling period occurred for a total of 6 months at both sites, allowing capture of the major flux (Vangeli et al., 2022) and continued monitoring to ensure any further peaks were also captured for both urine and dung. The detailed sampling protocol can be seen in Table 3.4.

Table 3.4 Manual gas sampling frequency

Week of measurement period	Frequency per week
Week 1 & 2	3x
Week 3 & 4	2x
Week 5, 6, 7 & 8	1x
Week 9, 10, 11, 12, 13, 14, 15, 16 & 17, 18	1x every 2 weeks
Week 19, 20, 21 ,22, 23, 24, 25, 26	1x every 4 weeks

3.2.3.2 Nitrous oxide emissions monitoring

Three headspace gas samples were collected from each chamber on each sampling day at T0min, T30min and T60min, resulting in 1hr total sampling time. An ambient air sample was also collected for each day. A 20ml syringe was used to collect samples, which were then inserted into 20ml glass vials that had been pre-evacuated (on the same day prior to gas sampling).

Soil samples (0-10 cm) were obtained from the area next to each plot on each sampling day. Samples were subsequently assessed for gravimetric soil moisture content through drying in an oven for 24 hours at 105 °C.

A Varian 450 Gas Chromatograph was used to analyse the samples and final N_2O fluxes were determined via the equation defined in Scheer et al. (2014). Temperature data from the nearby COSMOS automatic weather station (NERC- Centre for Ecology and Hydrology) (53.2252, -4.0126) and Henfaes weather station were obtained for use within the flux calculations from the upland and lowland pastures, respectively.

3.2.4 Data processing and statistical analyses

3.2.4.1 Methane emissions data

Statistical analyses and graphical representation of data throughout this chapter were performed either in IBM SPSS Statistics 27 (IBM, Portsmouth, UK) or Microsoft Excel. Data were not directly compared between pastures because measurements were made at different times of the year and therefore statistical tests for differences were deemed not applicable. One instance where statistical tests were used was during analysis of the hill pasture CH_4 emissions data whereby two spreadsheets presented by C-Lock Inc. for the same trial were analysed for any differences to inform which data to use in analysis (see Appendix 3.1). Two spreadsheets were presented due to animals removing their head from the unit frequently during measurement events, therefore C-Lock estimated a possible 10% underestimation of CH_4 results. Data were tested for normality using the Shapiro Wilks test (where it was concluded data were not normally distributed), then the Mann-Whitney U test was utilised to establish if there was a difference in distribution between the provided datasets ($p<0.05$).

GF data were supplied in an Excel spreadsheet format from C-Lock, via the annual support contract. In the lowland 2020 trial, an unknown animal with no RFID number visited the unit on several occasions. As there was no identification for the animal, data points related to it were removed. Outliers related to unit malfunction or sampling error were removed by C-Lock prior to receiving the data sheet, therefore no further outlier removal was conducted.

When considering variance, both population and within animal variance was calculated. For population variance, all measurements were included. For within animal variance, only animals who had provided >20 measurements over the measurement period were included. This was in line with manufacturer guidance. For calculation of Ym, all animals that lost weight were removed due to inability to capture this within the net energy calculations causing a skew in the results. This was relevant for the lowland 2020 experiment. The CH_4 Ym calculations from the sheep on the hill pasture were calculated differently, as explained in section 3.1.4.

3.2.4.2 Nitrous oxide emissions data

As above, statistical analyses and graphical representation of data was performed in IBM SPSS Statistics 27 (IBM, Portsmouth, UK) or Microsoft Excel. Additionally, N_2O fluxes between lowland and upland pastures were not directly statistically compared because they were conducted at different times, with varying N application rates (due to site-specific urine and dung collection).

N_2O fluxes were calculated using the change in headspace N_2O concentration with information on the chamber volume and surface area as well as local air temperature, as per (Scheer et al., 2014). Cumulative N_2O were determined using trapezoidal integration of the flux data. The N_2O-N EF (EF_{3prp}) was then determined by the following equation (De Klein et al., 2003):

$$EF = \frac{Total\ N_2O - N\ (treatment) - Total\ N_2O - N\ (control)}{N\ applied} * 100$$

3.3 Results

3.3.1 Overview of methane emission quantification from sheep

3.3.1.1 Summary of results

As the experiments occurred at different times, it was deemed not possible to directly compare results from each pasture. However, the measurements were made with the intention of following the typical management of an animal within a mixed altitude farm with animals often being in lowland areas in spring and then in upland and hill pastures over summer through to late autumn/early winter. Both sets of lowland measurements were made in spring-early summer and upland and hill measurements made in late summer-autumn. Therefore, an overview of data generated in each pasture, including enteric CH_4 production and local forage chemical characteristics, is given to indicate local emission output for use within LCA modelling.

3.3.1.2 Methane output

The population daily mean enteric CH_4 (g animal^{-1} day^{-1}) output varied across the period of the experiments conducted across both years (Figure 3.5 and Figure 3.6).

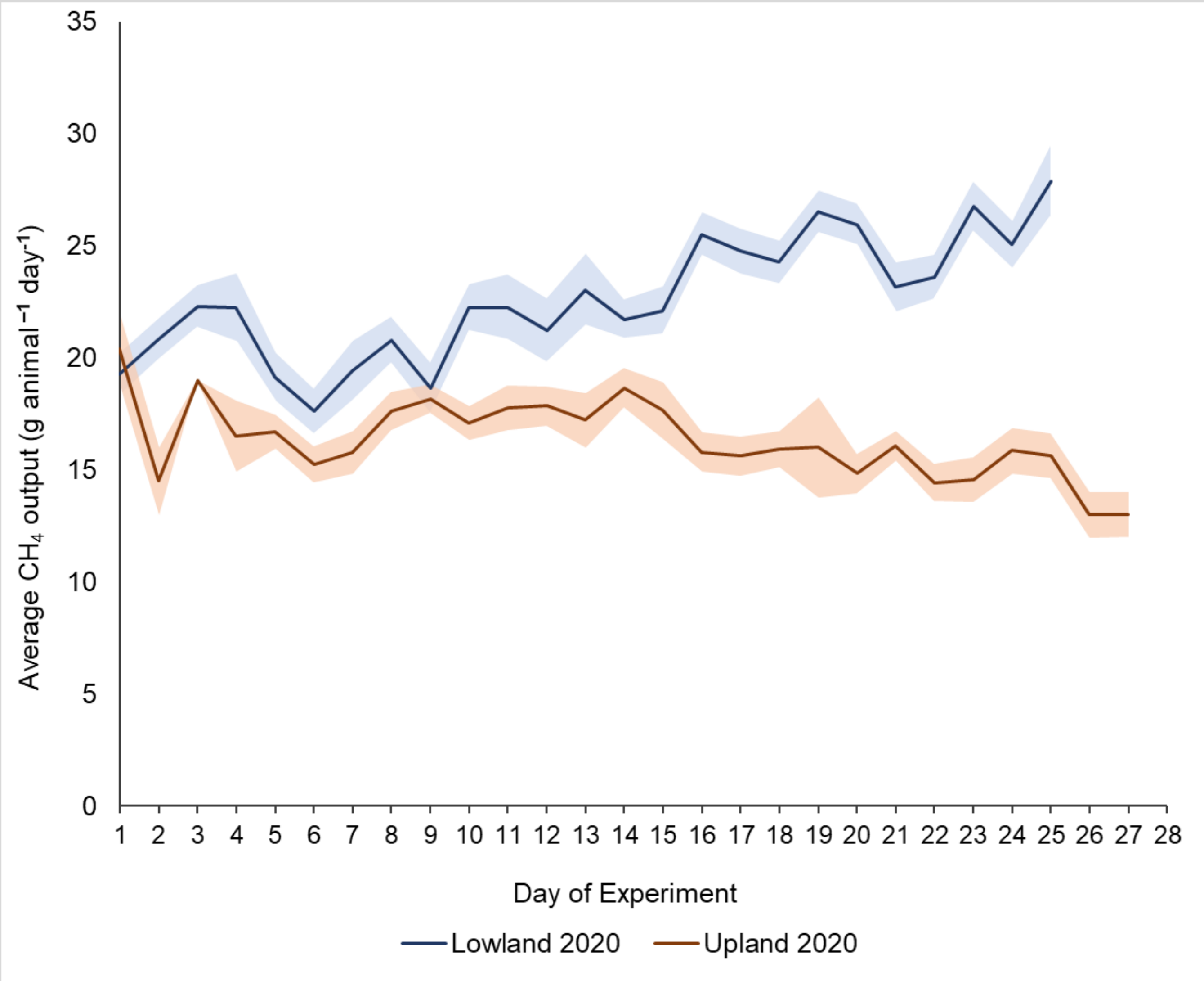

Figure 3.5 Population daily mean CH_4 (g animal^{-1} day^{-1}) measured in lowland and upland experiments conducted in 2020 (lowland experiment = 25 days of measurement, upland experiment = 27 days of measurement). Shaded area indicates daily SEM (n=20 in the lowland, n=20 in the upland).

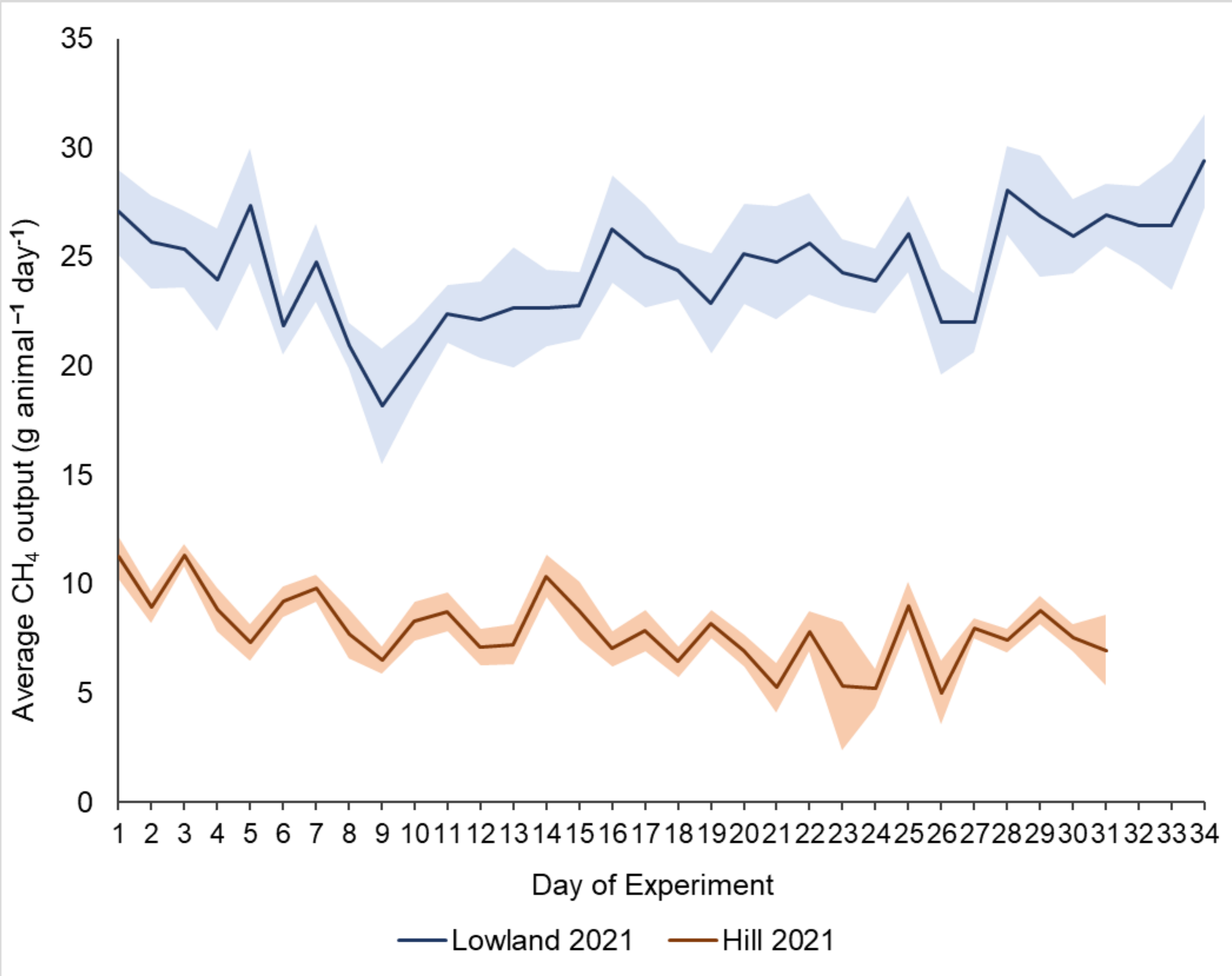

Figure 3.6 Population daily mean CH_4 (g animal^{-1} day^{-1}) measured in lowland and hill experiments conducted in 2021. Shaded area indicates daily SEM.

In both years, the sheep grazing the lowland spring pasture produced higher mean daily CH_4 (g animal^{-1} day^{-1}) across the population than in both the upland 2020 and hill 2021 autumn grazing periods, although the upland site was closer in value to the lowland than the hill site in 2021. The range, mean and annual expected CH_4 output (kg animal^{-1}) were calculated (Table 3.5).

Table 3.5 Range, mean and projected annual CH_4 values measured in each pasture (± SEM).

Measured values	Lowland		Upland	Hill
Year	**2020**	**2021**	**2020**	**2021**
Number of Measurements (n=)	754	675	682	364
Range (CH_4 g animal^{-1} measurement^{-1})	2.1 – 40.7	5.0 - 44.2	2.0 – 30.0	1.1 - 32.3
Mean CH_4 (g kg body weight (BW)$^{-1}$)	0.81 ± 0.01	0.61 ± 0.03	0.54 ± 0.01	0.21 ± 0.01
Mean CH_4 (g animal^{-1} day^{-1})	22.5 ± 1.1	23.8 ± 1.2	16.9 ± 2.0	8.0 ± 0.9
Annual CH_4 (kg animal $^{-1}$)	8.3 ± 0.2	8.9 ± 0.2	5.4 ± 0.3	2.9 ± 0.1

Both lowland measurement periods resulted in a mean CH_4 (g animal^{-1} day^{-1}) value >20 and a range beginning at 2.1 and 5.0 CH_4 (g animal^{-1} measurement^{-1}) in 2020 and 2021 respectively. Both trials then generated maximum values of >40g CH_4 animal^{-1} measurement^{-1}**.** The late summer-autumn upland and hill trials showed similar ranges. However, during the hill measurements, the lowest mean was recorded (7.9 CH_4 g animal^{-1} day^{-1}) and the lowest overall value (1.1 CH_4 g animal^{-1} day^{-1}), of all measurements made, alongside the lowest CH_4 in terms of g kg BW^{-1}.

3.3.1.3 Forage chemical composition

The mean forage chemical composition derived from forage analysis samples across the pastures can be seen in Table 3.6.

Table 3.6 Mean (n=5, ± SEM) forage chemical characteristics in the lowland, upland, and hill sites (data expressed on a dry weight basis).

Chemical Characteristic	**Lowland**		**Upland**		**Hill**
Year	2020	2021	2020 (1)	2020 (2)	2021
Dry matter (g kg^{-1})	209.6 ± 4.8	236.2 ± 14.7	256.4 ± 8.2	309.4 ± 25.2	289 ± 23.3
Crude protein (g kg^{-1})	185.6 ± 5.7	169.4 ± 13.8	163 ± 3.0	137.2 ± 8.8	129.8 ± 10.7
D value	71.3 ± 0.3	73.1 ± 0.9	70.3 ± 0.1	63.5 ± 0.4	63.3 ± 1.4
ME (MJ kg^{-1})	11.2 ± 0.1	11.48 ± 0.1	11.02 ± 0.0	9.96 ± 0.1	9.96 ± 0.2
NDF (g kg^{-1})	453.8 ± 11.3	417.4 ± 15.2	430.4 ± 14.5	524.6 ± 12.4	627 ± 40.1
Ash (g kg^{-1})	90.8 ± 1.6	82.4 ± 3.2	82.6 ± 1.8	91.8 ± 3.0	94.2 ± 2.4
Oil-A (g kg^{-1})	29.2 ± 0.4	26.6 ± 1.7	28 ± 0.6	19.2 ± 0.6	20 ± 2.0
Sugar (g kg^{-1})	122.8 ± 1.6	159.2 ± 17.4	154.2 ± 6.3	118 ± 7.7	77.2 ± 17.2
Nitrate N (%)	0.1 ± 0.0	0.08 ± 0.0	0.09 ± 0.0	0.06 ± 0.0	0.04 ± 0.0
Buffering capacity (meq kg^{-1})	378 ± 8.0	376 ± 6.2	383 ± 8.1	370 ± 0.0	370 ± 0.0

In both years, during the time of the lowland measurement, a greater digestibility value and crude protein value was determined from the forage than measured from the upland and hill forage samples taken later in the year. The two samples taken during the upland measurements showed a decrease in crude protein, digestibility, and ME but an increase in dry matter over time, indicating a drop in forage quality between autumn and winter. The hill forage appeared to have increased NDF (g kg^{-1}) than forage collected in other pastures, with the second upland sample having increased NDF (g kg^{-1}) than the original measurement. Ash (g kg^{-1}) was relatively similar across the pastures, varying between 82 and 94. Oil-A (g kg^{-1}) was similar in the lowland and first upland pasture samples, but the second upland sample was lower and similar in value to the hill pasture. The sugar (g kg^{-1}) was lowest in the hill pasture, with the lowland 2020 and second upland samples being similar. The lowland 2021 and initial upland pasture sample were similar and the highest values. Nitrate N (%) varied with both lowland values being similar to the initial upland pasture value. Values were then lower in the second upland pasture sample and hill sample. Buffering capacity (meq kg^{-1}) was similar across the sites, with the hill and second upland pasture samples having the lowest values.

3.3.2 Lowland measurements

3.3.2.1 Methane emissions

A total of 754 measurements were taken from sheep grazing the lowland pasture in 2020, with the between animal coefficient of variation (CV%) across the whole population being 28.5 and standard deviation 6.5. The mean within animal variance was 26.1. These measurements saw a drop in CH_4 until day 15, when the CH_4 emissions increased back to similar values measured on day 0. At the start of the lowland pasture measurements in 2020, there was low rainfall in comparison to the final two weeks of the measurement period, therefore this fluctuation in CH_4 output could be linked to changes in grass quality throughout this period.

A total of 675 measurements were made from the lowland pasture in 2021, with a between animal CV% of 27 and standard deviation of 6.6. The mean within animal variance was 25.4. Variation in the data was therefore similar across both lowland measurement periods.

3.3.2.2 Bodyweight, dry matter intake, gross energy intake and Ym value

Calculation of DMI and GEI using IPCC net energy equations allowed determination of a Ym value, using CH_4 emissions and animal bodyweight measurements (Table 3.7).

Table 3.7 Mean DMI, GEI, Ym and bodyweight data generated in the lowland pasture (± SEM).

Measured values	**Lowland**	
Year	2020	2021
Mean DMI (kg animal^{-1} day^{-1})	0.6 ± 0.01	0.9 ± 1.6
Mean GEI (MJ animal^{-1} day^{-1})	10.4 ± 0.3	16.2 ± 0.1
Ym%	12.8 ± 0.8	8.01 ± 0.9
Mean weight (start of trial) (kg)	28.2 ± 0.6	37.3 ± 1.01
Mean weight (end of trial) (kg)	28.1 ± 0.5	41.6 ± 0.96
Mean bodyweight change (kg)	0 ± 0.3	+4.4 ± 1.6

The lowland 2020 and 2021 measurements resulted in a Ym of 12.8% and 8.0% respectively. Animals remained largely at the same weight during the lowland 2020 measurement experiment, with a small decrease measured. Whereas, during the 2021 measurement period in the lowland, there was a mean increase in overall bodyweight of 4.4 kg. This could contribute to the higher DMI calculated in 2021.

3.3.3 Upland measurements

3.3.3.1 Methane emissions

A total of 682 measurements were made from the sheep grazing the upland pasture, with a between animal CV% of 31.2 and standard deviation of 5.1. Mean within animal variance was 28.1.

The upland measurements saw a decrease in enteric CH_4 across the duration of the measurement period. Two measurements of forage characteristics were taken during this time due to the changing season, and this gradual decrease in CH_4 could be reflected by the decrease in forage digestibility across the duration of the measurement period (see Table 3.6).

3.3.3.2 Bodyweight, dry matter intake, gross energy intake and Ym value

As completed for the lowland pasture measurements, DMI, GEI and Ym were calculated from the data measured in the upland pasture (Table 3.8).

Table 3.8 Mean DMI, GEI, Ym and bodyweight data generated in the upland pasture (± SEM).

Measured values	**Upland**
Year	2020
Mean DMI (kg animal^{-1} day^{-1})	0.8 ± 0.02
Mean GEI (MJ animal^{-1} day^{-1})	14.5 ± 0.4
Ym%	6.5 ± 0.2
Mean weight (start of trial) (kg)	28.2 ± 0.6
Mean weight (end of trial) (kg)	33.9 ± 0.5
Mean bodyweight change (kg)	+5.7 ± 0.6

Animal bodyweight increased by 5.7 kg on average, with a mean start weight of 28.2 kg and finishing weight of 33.9 kg. The resulting Ym for this measurement period was 6.5% ± 0.2.

3.3.4 Hill measurements

3.3.4.1 Methane emissions

A total of 364 measurements were made from sheep grazing the hill pasture. There was a between animal CV% of 44.9 and standard deviation of 3.5. Mean within animal CV% was 45.6. This indicates higher variability of emissions measured in the hill pasture.

3.3.4.2 Bodyweight, dry matter intake, gross energy intake and Ym value

The mean start and end weight and bodyweight change across the animals during this measurement period are detailed in Table 3.9.

Table 3.9 Mean bodyweight (start and end) and bodyweight change in the hill experiment (± SEM).

Measured values	**Hill**
Year	2021
Mean weight (start of trial) (kg)	39.8 ± 1.3
Mean weight (end of trial) (kg)	32.1 ± 1.4
Mean bodyweight change (kg)	-7.0 ± 1.4

Animals lost significant weight from the hill pasture, with a mean bodyweight loss of 7 kg over the 1-month period. As a result, there were issues with using the net energy calculations for data generated from the hill pasture, due to the animals losing significant weight and the calculations not being designed to capture this. Application of weight loss within the net energy for growth calculation resulted in negative GE and resulting Ym values.

It was therefore decided to conduct a different exploration of Ym values for this pasture so that an indicative value could be used within the proceeding LCA modelling. Data from the final four days of the measurement period for 7 animals were used to indicate a DMI, GEI and Ym based on the finishing bodyweight of the animals (Table 3.10). All measurements across the 4 days were included due to the smaller number of data points measured. It was chosen to analyse the final days of data as all animals' final weights were recorded within

7 days of the CH_4 measurements concluding, therefore minimal weight changes should have occurred.

Table 3.10 Indicative DMI, GEI and mean Ym% (across the four final days) values from CH_4 data generated in the hill pasture across the 7 animals (± SEM).

Animal Number	DMI (kg animal^{-1} day^{-1})	GE (MJ animal^{-1} day^{-1})	Mean Ym%
1	0.6	11.3	3.5 ± 0.2
2	0.6	11.3	4.6 ± 0.4
3	0.67	12.5	4.6 ± 0.6
4	0.5	9.6	5.2 ± 0.9
5	0.6	11.6	3.2 ± 0.1
6	0.56	10.3	4.2 ± 0.3
7	0.58	10.8	3.9 ± 0.4
Mean	0.6 ± 0.02	11.03 ± 0.4	4.2 ± 0.2

3.3.5 Excretal nitrous oxide emission quantification from sheep

3.3.5.1 Urine and dung characteristics at each site

The total N (g N l^{-1}), total C (g C l^{-1}) and N loading rate (as well as the DM% and OM% for dung samples) applied at each site for both urine and dung are shown in Table 3.11.

Table 3.11 Urine and dung characteristics at each site (± SEM). Dung total N value taken from van der Weerden et al. (2011) (as described in Section 3.2.2.2).

Urine	**Lowland (n=21)**	**Upland (n=19)**
Total N (g N l^{-1})	2.1 ± 0.3	3.7 ± 0.3
Total C (g C l^{-1})	3.0 ± 0.4	5.8 ± 0.5
N loading rate (kg N ha^{-1})	164.2	217.9
Dung	**Lowland (n=29)**	**Upland (n=20)**
Total N (g N kg^{-1})	6.5 ± 0.6	6.5 ± 0.6
N loading rate (kg N ha^{-1})	118.1	132.6
Dry matter (DM) %	19.5 ± 0.8	15.6 ± 0.5
Organic matter (OM) %	11.7 ± 1.2	7.6 ± 0.6

The urine N content was comparable between both pastures, although with a fairly wide range. In the lowland, total N ranged from 0.7 – 6.6 g N l^{-1}. In the upland, it ranged from 1.4 – 7.5 g N l^{-1}. In terms of total C, in the lowland it ranged from 0.6 – 7.8 and in the upland, 2.16 – 10.32 g C l^{-1}. Generally, there was a higher N loading rate derived for the upland measurements in comparison to the lowland experiments across both excreta types.

3.3.5.2 Measured nitrous oxide fluxes

The measured N_2O fluxes for the sheep urine and dung applied to the lowland pasture can be seen in Figure 3.7 (as well as the flux from control plots). The urine treatment had the highest flux between day 0 and day 20 after application, showing the greatest peak on day 4 before decreasing slightly, then peaking again on day 9. After this peak period, the urine treatment returned to control levels showing little fluctuation in calculated emissions thereafter. Both the control and dung treatments had lower fluxes, with the dung treatment resulting in a small peak just before day 40 after application.

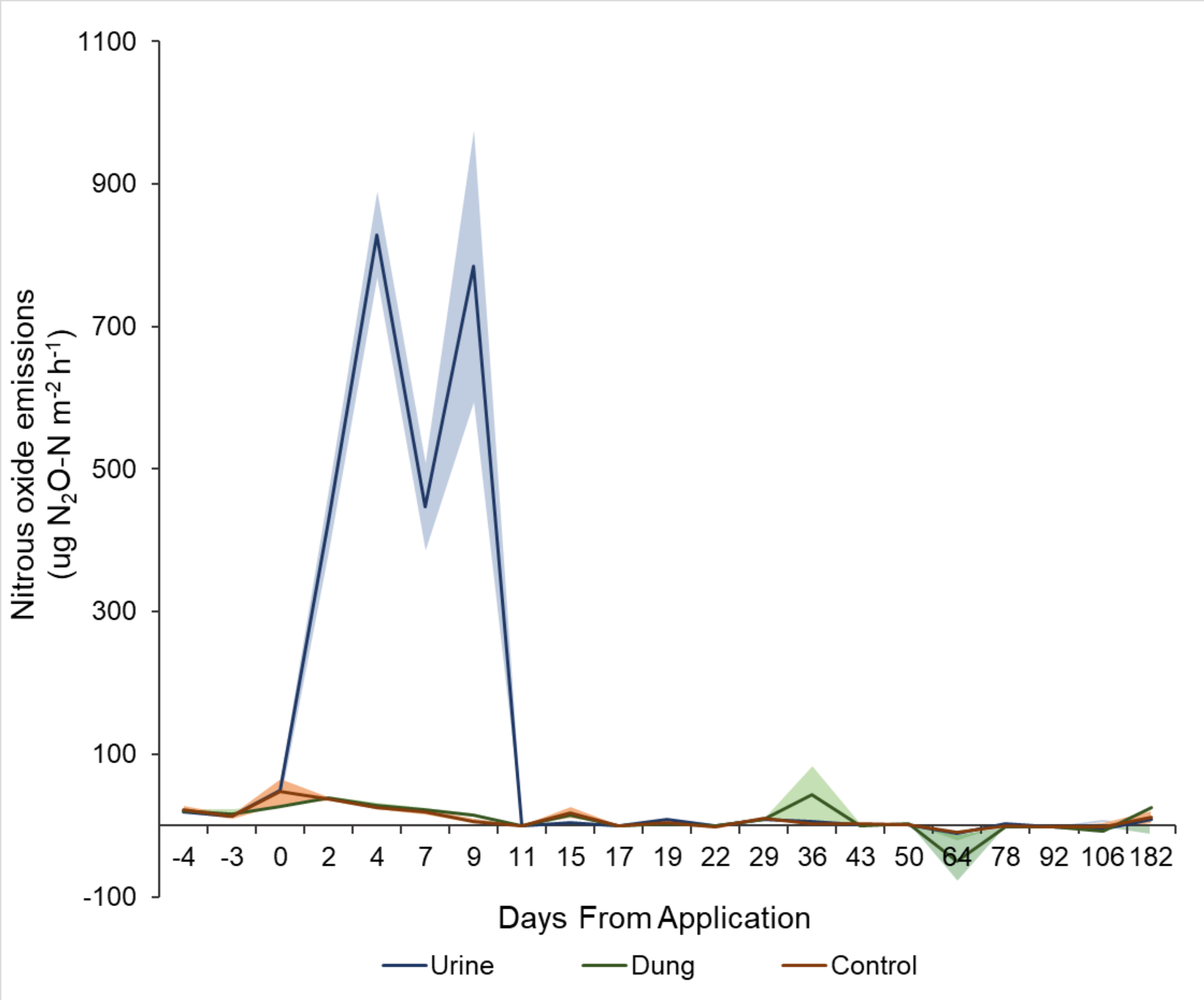

Figure 3.7 N_2O emission flux (µg N_2O-N m^{-2} h^{-1}) (real sheep urine, dung, and control) measured in the lowland pasture from 2020-2021. Shaded area represents the SEM (n=6), with the line representing the mean of the six replicates for each treatment.

The measured N_2O fluxes for the sheep urine and dung applied to the upland pasture can be seen in Figure 3.8 (as well as the flux from control plots).

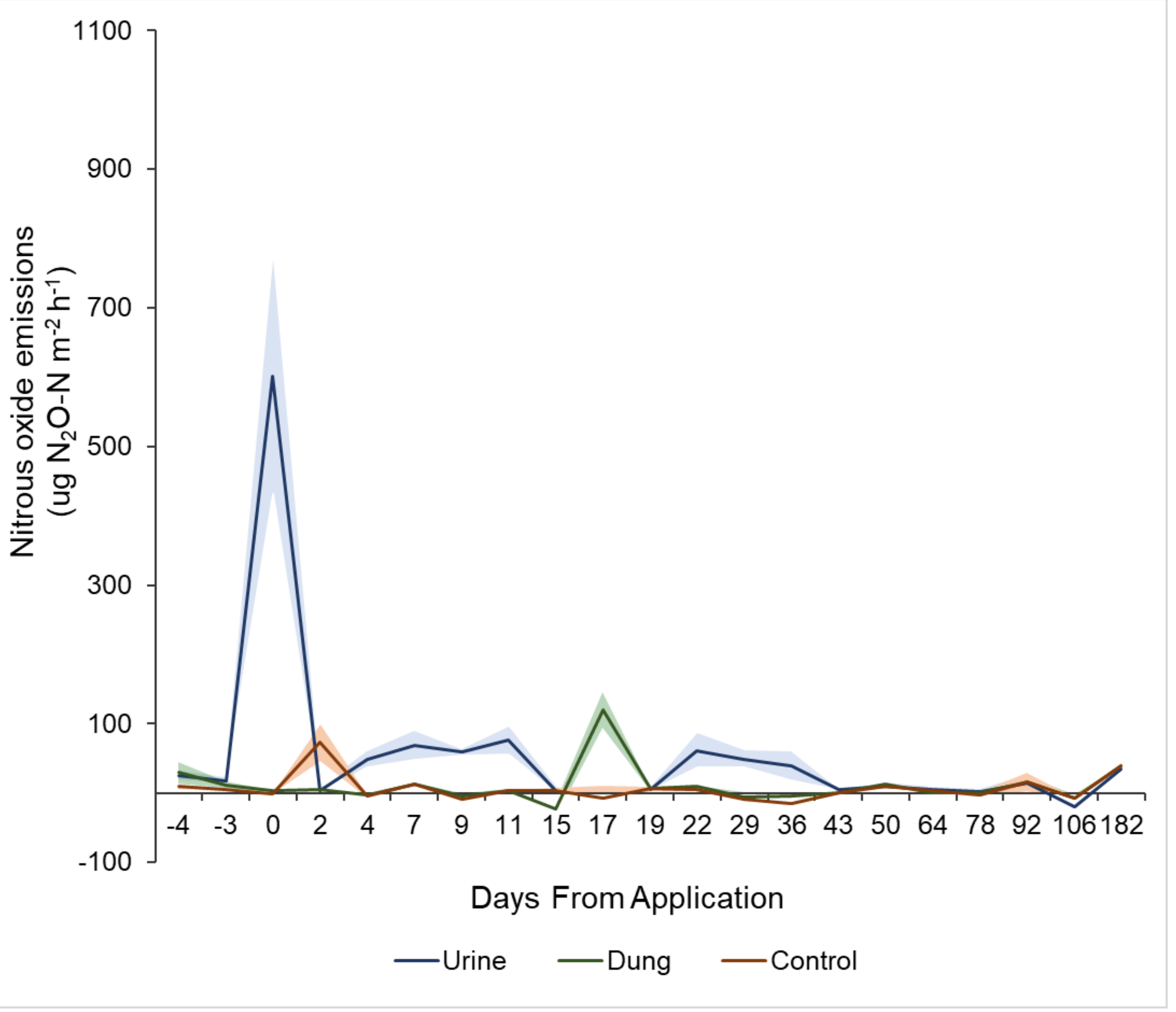

Figure 3.8 N_2O emission flux (µg N_2O-N m^{-2} h^{-1}) (real sheep urine, dung, and control) measured in the upland pasture from 2020-2021. Shaded area represents the SEM (n=6), with the line representing the mean of the six replicates for each treatment.

The urine treatment peaked quickly, with the highest value recorded on the day of application indicating immediate volatilisation of the urine. The peak then dropped off with two much smaller peaks (<100 µg N_2O-N m^{-2} h^{-1}) then occurring before levelling off for the remainder of the 6-month sampling period. As expected, a lower flux was seen from the dung and control plots with a small peak in the control plots generated on day 2 after application, and a small peak in the dung plots generated on day 17 after application.

The soil WFPS (%) was recorded at each pasture across the measurement period (Figure 3.9).

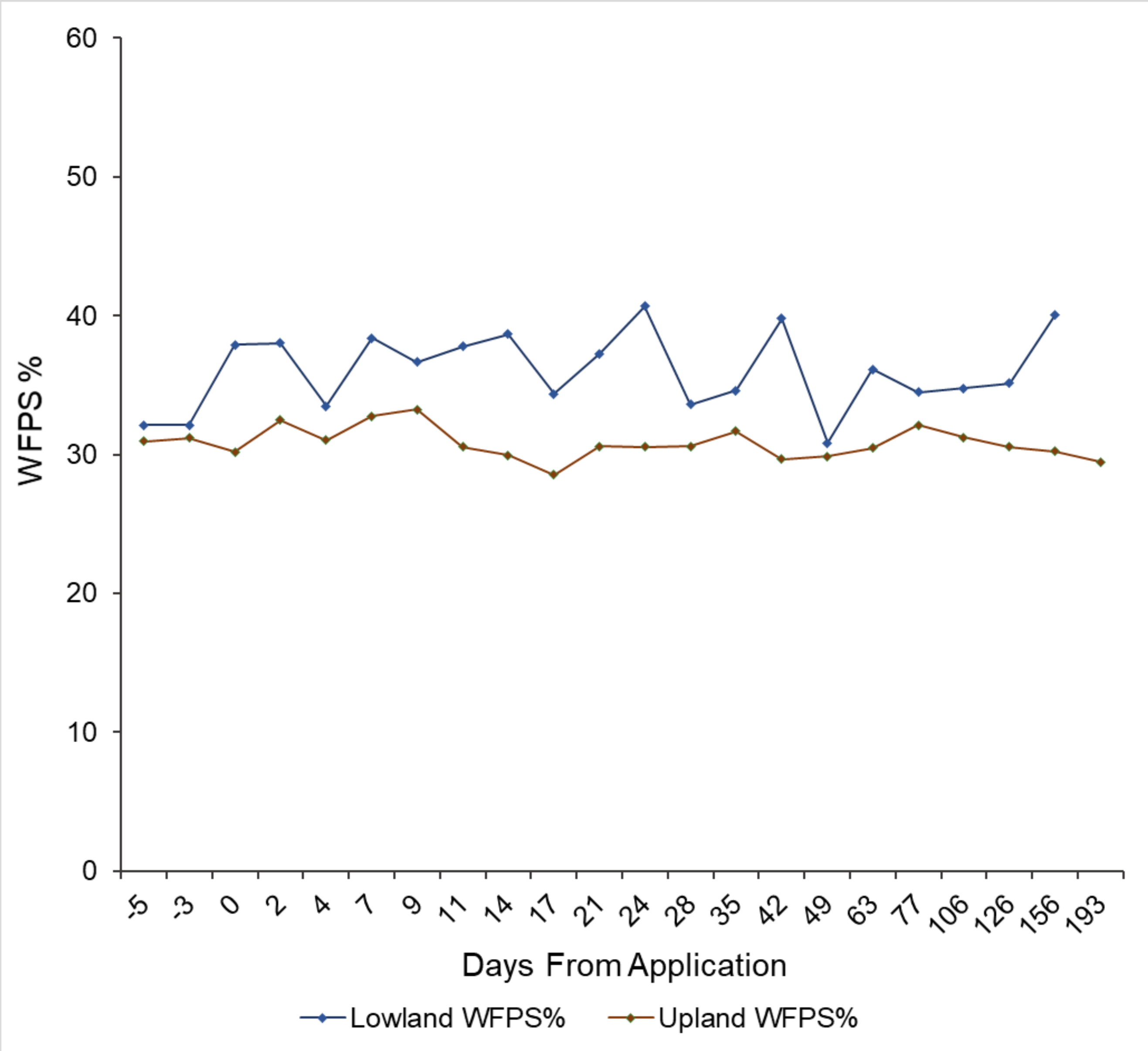

Figure 3.9 Soil WFPS% (0-10cm) in lowland and upland pastures across measurement period, after application of urine and dung samples.

In the lowland pasture, the WFPS% ranged from 30.8% to 40.7% across the measurement period. In the upland, it ranged from 28.5% to 33.3%. The mean value in the lowland pasture was 36% and in the upland pasture 30%. The lowland pasture showed more variability across the measurement period than the upland pasture (CV% 7.8 vs. 3.0).

3.3.5.3 Cumulative nitrous oxide emissions and nitrous oxide emission factor

Cumulative N_2O emissions across replicates in the lowland pasture proceeding treatment application ranged from 0.3 - 1.8 kg N_2O-N ha^{-1} for urine, -0.6 - 0.7 kg N_2O-N ha^{-1} for dung and -0.02 - 0.7 kg N_2O-N ha^{-1} for control plots. Cumulative N_2O emissions measured across replicates in the upland pasture varied from -0.07 - 1.2 kg N_2O-N ha^{-1} for the urine treatment, -0.05 - 0.6 kg N_2O-N ha^{-1} for dung and -0.06 - 0.5 kg N_2O-N ha^{-1} for the control treatment. The mean cumulative N_2O and N_2O-N EF values (± SEM) measured in both pastures are shown in Table 3.12.

Table 3.12 Cumulative N_2O emissions and N_2O-N EF calculated from the urine, dung, and control treatments in the lowland experiment (± SEM)

Treatment	Cumulative N_2O emissions (kg N_2O-N ha^{-1})	N_2O-N EF (% of N applied)
Lowland		
Urine	1.2 ± 0.2	0.6 ± 0.1
Dung	0.08 ± 0.2	-0.07 ± 0.2
Control	0.2 ± 0.1	-
Upland		
Urine	0.7 ± 0.2	0.20 ± 0.08
Dung	0.2 ± 0.09	0.00 ± 0.07
Control	0.2 ± 0.1	-

Urine application generally was associated with a higher EF than dung. The dung treatment produced a slight negative EF therefore emissions can be considered negligible from this source. In the upland site, urine application was likewise associated with a higher EF than dung application. The dung application EF was calculated as 0.00, showing negligible emissions from this source.

The combined excretal N_2O-N EF for both lowland and upland pastures are shown in Table 3.13.

Table 3.13 The combined excretal N_2O-N EF (EF_{3prp}) (% of N applied) for both lowland and upland pastures (assuming a 60:40 urine to dung ratio as per (Webb and Misselbrook, 2004) (n=6, ± SEM).

Pasture	Combined excretal N_2O-N EF (EF_{3prp})
Lowland	0.36 ± 0.15
Upland	0.12 ± 0.03

3.4 Discussion

3.4.1 Enteric methane changes across pasture types

Differences in CH_4 output between different forage sources and pastures is well known (Grainger and Beauchemin, 2011). We hypothesised that emissions would vary in each pasture with upland and hill sites having lower emissions, and while statistical significance of the results here cannot be determined because measurements were made at different times, the results appear to follow this hypothesis. The enteric CH_4 emissions decreased with increasing altitude in terms of CH_4 output g animal^{-1} day^{-1} and g BW^{-1} basis. This aligns with results in other work. One study assessed CH_4 emissions from lambs zero-grazed on both freshly cut ryegrass and a mixed extensively managed permanent pasture representative of upland grazing. The lambs were measured as having a daily CH_4 output of 16.1 g animal^{-1} day^{-1} when fed ryegrass vs. a reduced output of 12.9 g animal^{-1} day^{-1} when fed with forage from the extensive pasture (Fraser et al., 2015). In a different study, fresh hill grass resulted in the lowest CH_4 output when fed to hill breed ewes in comparison to pelleted ryegrass and fresh lowland grass (Zhao et al., 2017). The values reported by Zhao et al. (2017) ranged from 11.8 g animal^{-1} day^{-1} when feeding hill grass to 22.4 g animal^{-1} day^{-1} when feeding pelleted ryegrass, reflecting fairly closely to the results generated with the GF here (7.8 g animal^{-1} day^{-1} in the hill – 22.7/24.3 g animal^{-1} day^{-1} in the lowland). The aforementioned study made a link between consumption of hill grasses and reduced DMI, which may in part explain the lower CH_4 emissions measured here as output is directly linked to DMI. This trend has also been seen in cattle, with lowland cattle having 10% increased DMI and 15% increased CH_4 (g animal^{-1} day^{-1}) in comparison to upland cattle (Richmond et al., 2015).

The reduced CH_4 output determined in the hill pasture during this study combined with the weight loss measured indicates that some of the reduced output in the hill may be due to the animals eating less (the hill site also had lowest estimated DMI at 0.6 kg day^{-1}) and not solely from environmental or forage characteristics. However, the potential reduced DMI in this study may have arisen due to the animals being confined in a small, enclosed paddock, therefore unable to roam and graze. Indeed, in all aforementioned studies, the permanent pasture or hill grass areas assessed appeared to have higher CH_4 output when related to DMI or GEI. There are potential implications for hill farming as a result as the hardy hill sheep breeds, such as the Welsh Mountain, have generally also been measured as less productive than other breeds in terms of growth rate, finishing weight and lambing percentage (Wolf et al., 2014). Therefore, the potential for higher CH_4 emissions in relation to their feed intake may further affect the suitability of these systems. Crossbreeding with larger breeds has been shown to improve flock productivity whilst retaining the natural hardiness of the Welsh Mountain ewes. However, the effect of changing breeding and producing bigger sheep on CH_4 output has potential additional impacts on CH_4 emissions. Improved breeding to increase ewe size has been linked to increased GHG emissions, therefore future breeding will have to consider GHG reductions against increasing economic value (Lambe et al., 2014).

Certain plant compounds are being assessed for their potential to reduce ruminant CH_4, including condensed tannins, essential oils, sapponins and flavonoids (Ku-Vera et al., 2020). Tannins were seen to cause a 30-35% reduction in ruminant CH_4 emissions from cattle and the effect is linked to how the compounds alter the rumen microbial environment. Recent experiments conducted with sheep found that feeding tropical tanniferous legumes resulted in a decrease in CH_4 in comparison to other diets (Dias Moreira et al., 2013). Common species identified in the upland and hill site in this study were *Trifolium repens* and *Calluna vulgaris* respectively. Both of these species contain condensed tannins and are being investigated as a natural CH_4 inhibitor for livestock grazing (Roldan et al., 2020; Varga et al., 2021).

Calluna in particular has been assessed for its effect on enteric CH_4 from grazing sheep. It is an ericaceous species and a study assessing effect of these plant types on ruminant CH_4 production found that an increase in consumption of ericaceous species resulted in a decrease in sheep and deer enteric emissions (Pérez-Barbería et al., 2020). This equated to a potential reduction of 63.8kt CH_4 $year^{-1}$ based on current sheep and deer European populations against a diet consisting of grass, suggesting that extensive pastures with high ericaceous species levels may have a natural CH_4 inhibiting effect. However, studies have shown sheep to be selective grazers with <10% of their diet attributed to consumption of *C. Vulgaris* (Fraser et al., 2009). This effect may therefore be limited dependent on what the animals naturally consume. These results further discourage use of generic information when assessing these pastures, as the wider potential benefits are not yet fully assessed. There are multiple factors that could affect changes in enteric CH_4 across pastures, but a lack of evidence is present for fully quantifying these within CF models and GHG inventories.

Regarding emissions variation, the between animal CV% increased with altitude. This corroborates with other results where cattle grazing in upland areas had higher CV% than those in lowland sites (Richmond et al., 2015). This could be related to the greater diversity of grass species and likely higher variation of DMI consumed by the animals in the extensive pastures. There could also be some additional variation introduced as a result of the use of the GF unit itself. The aforementioned study in cattle utilised the sulphur hexafluoride (SF_6) measurement technique and historically, it has been determined that this technique produces higher variability than a respiration chamber (Grainger et al., 2007; Pinares-Patiño et al., 2011). However, work comparing the GF with the SF_6 has found the GF to be less variable than both SF_6 and respiration chambers (Alemu et al., 2017; Hristov et al., 2016). The variation measured in the upland and hill sites here is higher than others using GF, although no published studies have used the GF with small ruminants across different pastures in this way (Hegarty, 2013). The lowland CV% values of 28.5 and 27 have been seen when measuring CH_4 with other methods in cattle, so are closer to values derived previously (Garnsworthy et al., 2012). There is a potential for environmental factors to alter GF accuracy and therefore emissions variability, particularly windspeed. However, this can be mitigated by fitting the unit with an anemometer and utilising that data to correct the CH_4 output for windspeed (Hristov et al., 2018; Huhtanen et al., 2015). The unit used here did have an anemometer and recorded wind speed across all three sites.

3.4.2 The Ym Value: How appropriate are default values for UK lamb production?

The Ym values generated in this study ranged from 4.2% in the hill site to 12.8% in the lowland site, with individual pasture ranges varying dependent on the pasture measured. All values in the study are comparable to previously published ranges for sheep and cattle: 3.8-12 with a mean of 8 (Ma et al., 2019). A different study assessing Ym against different forage levels offered to sheep determined Ym values of 7.3-9 (de Azevedo et al., 2021). Cattle studies have reported values of 2-12 (Jaurena et al., 2015).

The IPCC Ym (6.7% ± 0.9) is derived from data generated in New Zealand over six years using respiration chambers (IPCC, 2019). Significant work has been completed in New Zealand to better quantify ruminant CH_4, given its high contribution to their GHG inventory (Swainson et al., 2016). A key limitation when applying these values to UK production systems, is that the experiments were conducted by feeding the animals ryegrass swards only (Muetzel and Clark, 2015). The forages varied in terms of quality but did not cover the variety of species that upland and hill sheep in the UK may consume. A significant limiting factor in the predictive power of models is the frequent use of limited datasets used to create model parameters (Moraes et al., 2014). However, the upland Ym value of 6.47% ± 0.2 reflects quite closely with the IPCC default value and it is also comparable with a New Zealand generated Ym of 6.3% for adult sheep (Swainson et al., 2016). The higher Ym calculated in the lowland pastures (12.8 ± 0.8 in 2020 and 8.01 ± 0.9 in 2021) could be attributed to the higher CH_4 generated here in comparison to the estimated GEI, whereas in the upland pasture animals had lower enteric CH_4 emissions. Changes in pasture digestibility between the sites will likewise alter estimations of GEI. The higher value in 2020 compared to 2021 could be attributed to the decreased GEI as animals had lower LW gain.

The lowest recorded Ym across all the pastures was recorded from the hill pasture (4.17 ± 0.2). However, the robustness of this value is questionable because of the measured weight loss of the animals that grazed this pasture. The inability of the calculations to capture significant weight loss affects the accuracy of the GE calculated from the animal weight data, in turn affecting the accuracy of the Ym value when related to DMI and GE (and by extension Ym). Limited studies exist assessing Ym of sheep consuming hill grass, but one study found the permanent pasture had a higher value (5.2% vs 5.6%) (Fraser et al., 2015). Conversely, Zhao et al 2017 found similar values for hill ewes fed lowland grass and hill grass, with a slight decrease in hill grass (5.7% vs 5.6%). Estimation of animal intake is particularly difficult in pastures where the animals roam, particularly due to current techniques interfering with animal behaviour (Giovanetti et al., 2020). Further work is necessary to improve assessment of DMI and GEI in upland and hill sites, alongside the potential for reduced CH_4 output to increase confidence in Ym and enteric CH_4 predictions.

The use of the Ym has received critique in recent years, due to its inability to capture variations in enteric CH_4 dependent on diet factors, such as the presence of structural carbohydrates. Thus far, prediction equations have been determined as poor at estimating enteric CH_4, particularly Tier 1 estimates or those that did not account for diet variations (Ellis et al., 2010). IPCC Tier 2 equations fared slightly better in their predictive ability when assessing individual animal output. There is potential for high uncertainty being introduced to GHG inventory estimates due to this, as well as conclusions on potential mitigation methodologies being incorrect (Patra and Lalhriatpuii, 2016). Equations should therefore be developed to include a wider range of factors that may cause enteric CH_4 to vary, underpinned by larger datasets across different grazing systems.

3.4.3 Differences across altitudes in nitrogen loading rates, cumulative nitrous oxide emissions and nitrous oxide emission factors in comparison to other literature

It is known that the urine N concentration and urination volume can affect the N loading rate and therefore N_2O emissions following excreta deposition (Marsden et al., 2016). Mean urine N concentration in this study was measured as 2.4 ± 0.3 g N l^{-1} in the lowland and 3.7 ± 0.4 g N l^{-1} in the upland. Recent studies report mean values across seasons of between 3.3 and 9.3 g N l^{-1} in lowland and hill pastures and mean values of 4.3 – 6.4 g N l^{-1} in upland pastures only (Mancia et al., 2022a; Marsden et al., 2018). While the results of this study are slightly lower in comparison, the results are similar to those reported in a global meta-analysis giving a urine N concentration range of 2.6 g N l^{-1} to 14.6 g N l^{-1} across a variety of studies (although this does include all types of livestock excreta) (López-Aizpún et al., 2020). Mean urine volumes in this study were 215ml in the lowland and 185ml in the upland. The urine volumes measured are both higher than the reference often cited as a representative sheep urine volume of 150ml (and used by Mancia et al., 2022) (Doak, 1952). However, they are similar to other reference values of between 100 and 200ml (Haynes and Williams, 1993). Work in the same site determined mean urine volume values of 177, 239 and 377 ml in spring, summer and autumn respectively in semi-improved upland pasture with a mean value of 364 in Autumn in the lowland pasture (Marsden et al., 2020). The values in this study are similar to the lower values reported by Marsden et al. (2020). Further study that estimated urine volumes via tri-axial accelerometers in the same breed of sheep used in this study (Welsh Mountain), obtained an average value of 159 ± 1ml, with a range of 17 – 745 ml (Marsden et al., 2021). The urine volumes in this study fit within this range. The urine N loading rate was 164.2 kg N ha^{-1} and 217.9 kg N ha-1 for the lowland and upland respectively and the dung N loading rate was 118.1 kg N ha^{-1} and 132.6 kg N ha^{-1} for the lowland and upland respectively. The values are within the range reported by a New Zealand meta-analysis, although both lower than the mean value presented (van der Weerden et al., 2020). Another study determined N loading rate values ranging from 171 to 336 kg N ha^{-1} for sheep, which are similar to this study although still higher than the lowest values reported here (Luo et al., 2013).

Cumulative N_2O emissions for sheep urine measured in this study were 1.2 kg N_2O-N ha^{-1} in the lowland pasture and 0.7 kg N_2O-N ha^{-1} in the upland pasture. A similar study measured excretal N_2O from sheep grazing lowland and hill areas in Ireland (Mancia et al., 2022a). Cumulative N_2O post urine application was on average 0.26 ± 0.05 kg N_2O-N ha^{-1} in the lowland and 0.01 ± 0.02 kg N_2O-N ha^{-1} in the hill. Other data measured in an extensive/upland pasture similar to the pasture in this study had values between 2.8 and 3.8 mg N_2O-N m^{-2} (0.03 - 0.04 kg N_2O-N ha^{-1}) (Marsden et al., 2018). These values are similar to that determined in the study in Ireland. An assessment of cumulative N_2O emissions in a hill grazing pasture resulted in values of 0.28 – 0.62 mg N_2O-N m^{-2} (<0.00 kg N_2O-N ha^{-1}) (Marsden et al., 2019). The results of this thesis are higher than both of the above studies, highlighting the variability in emissions across different works. However, values are within the ranges reported by Luo et al., 2013 for sheep urine across different sites and seasons: 0.3 - 7.99 kg N_2O-N ha^{-1} for flat grazing areas and -0.01-1.2 kg N_2O-N ha^{-1} for medium slope areas. For dung, fewer data are published although the study in Ireland found values of 0.3 kg N_2O-N ha^{-1} in the lowland and -0.01 kg N_2O-N ha^{-1} in the upland (Mancia et al., 2022a). The values determined in this study are between the published values and are 0.08 and 0.2 kg N_2O-N ha^{-1} in the lowland and upland site respectively. Higher cumulative emissions were measured in the upland, which conflicts with the findings of Mancia et al. (2022). However, as with urine, the values calculated for dung are approximately within the range of published values of 0.1 - 0.7 kg N_2O-N ha^{-1}, although this experiment is on the lower end of that range (Luo et al., 2015).

In general, values for urine N concentration, urine volume, N loading rate and cumulative emissions were similar to the wider published literature. It was hypothesised that a difference would be determined between the lowland and upland site as it is well known that N_2O emissions are largely driven by a variety of environmental factors (Krol et al., 2016). This appears to have been the case with differences in the measured data across the lowland and upland pastures. While it is difficult to determine contributing factors within the soil to emissions due to soil sampling not being taken throughout the measurement period, soil WFPS% is one of these factors and was measured alongside N_2O in this study due to its known association. WFPS% was higher in the lowland than in the upland throughout the duration of both experiments, which is a possible contributor to the higher peak in the lowland. In addition, as mentioned, soil pH is a highlighted factor in affecting N_2O output in that acidic soils inhibit soil nitrification (Marsden et al., 2019). The upland site in this study had a more acidic pH than the lowland.

3.4.4 Disaggregation of excretal nitrous oxide emission factors by altitude

Disaggregation of excretal N_2O EFs is not currently common practice globally, but discussions centred on how to do this most appropriately have been increasing. An option for EF_{3prp} disaggregation for lamb production systems would be by altitude. The combined excretal EFs generated in this study were 0.34% in the lowland site and 0.12% in the upland site, which correlates with other work conducted across a variety of locations that compared emissions in different altitudes. The values in this study are similar to those presented for the same sites during experiments conducted in previous years (Marsden et al., 2019). Additionally, field experiments conducted in New Zealand over a period of 13 years found that where excreta was deposited in areas with greater than a 12° slope, the EF was significantly less (Kelliher et al., 2014). The derived values for sheep urine were 0.55% ± 0.19 in lowland areas, 0.40% ± 0.10 in hill areas with a low slope and 0.16% ± 0.05 in hill areas with a medium-steep slope. A recent meta-analysis has stated sheep urine EFs in New Zealand should be 0.5% in lowland areas and 0.08% in areas with a medium-steep slope (van der Weerden et al., 2020). These values are relatively similar to the values determined in this study. Other work has measured combined sheep excretal N_2O EFs that are negligible, with values of -0.01% and -0.03% for lowland and hill pastures respectively (Mancia et al., 2022a). This highlights that while values are in general similar, there is a potential for much lower EFs across different sites in different seasons.

To accurately make decisions regarding future land use, it is important to utilise representative evidence when making said decisions. For UK lamb production, it is particularly important to consider how altitude may affect emissions burdens as approximately 60% of the breeding flock are present on UK uplands. Additionally, there are currently conflicts around upland land use, partly surrounding reaching UK afforestation targets (Fraser, 2008; Hardaker, 2018). Over estimation or under estimation of the environmental impacts of a land use may lead to incorrect conclusions being drawn that may hamper sustainability efforts. For example, applying the above disaggregated EFs to the New Zealand national inventory resulted in a 30% decrease of excretal N_2O, leading to disaggregation being recommended for use in the New Zealand inventory. Likewise, in the UK a 45% decrease was estimated for the UK inventory from sheep related excretal N_2O (based on the methodologies employed in the inventory at the time of publication) (Marsden et al., 2019; van der Weerden et al., 2020). In addition, the IPCC currently recommends 0.3% EF to be used for sheep systems (IPCC, 2019), revised from the previous recommendation of 1%. As mentioned previously, the UK inventory currently employs a value similar to the IPCC default for the whole sheep population (0.3%) (Brown et al., 2022). This may be appropriate for some systems yet is higher than all EFs reported here for upland and hill systems. These findings support the work of Marsden et al. (2019) and suggest that the N_2O burdens of sheep systems may be currently overestimated in the UK.

While evidence points to disaggregation being an appropriate way forward, further work would be needed across UK regions to fully assess changes in N_2O EFs before applying a change to the UK inventory. Marsden et al, (2019) also suggest that sheep urine EFs could be disaggregated by soil pH and anaerobicity, if livestock grazing numbers can be disaggregated accurately for such soil conditions. Disaggregation could potentially enhance accuracy of emission estimates but requires a greater resolution and breadth of collected data (Mancia et al., 2022b). Not only do these findings have implications for the national inventories, but also for farm level CF. Historically, upland and hill lamb has been associated with a higher CF than lowland lamb (Jones et al., 2014; Taylor et al., 2010). The impact of disaggregated EFs has not yet been assessed for lamb production CFs, but upland and hill lamb may have a different footprint than previously calculated. However, excretal N_2O tends to comprise a smaller portion of lamb production footprints than enteric CH_4 (Bhatt and Abbassi, 2021).

3.5. Conclusions

Lamb production systems continue to be a key source of both enteric CH_4 and excretal N_2O emissions in Welsh and UK agriculture. In order to reach Net Zero goals, it is essential to understand emissions burdens across a variety of system types. Lamb production by its nature involves exploiting different pasture types (lowland, upland, and hill) with differing environmental factors and diets, which can all affect emissions. The data collected in this study adds to the existing evidence base of there being differences in both enteric CH_4 and excretal N_2O (EF_{3prp}) between lowland, upland, and hill pastures. Both enteric CH_4 and excretal N_2O appeared to decrease with increasing altitudinal gradient, although further data is required in particular for enteric CH_4 before definitive conclusions are drawn. The variation in both EF_{3prp} and CH_4 Ym suggest that use of the singular IPCC default and country-specific values may not be appropriate across the variety of grazing areas within Wales and the UK. With the current data available, it would appear sensible to disaggregate EFs based on altitude although other factors such as soil type could be chosen. However, there is a lack of robust data available for factors that affect emissions variability. For example, there is little information that details animal diets across the upland and hill pastures and how this affects enteric CH_4. Additionally, while knowledge of the factors that affect N_2O cycling are known to an extent, the magnitude of their effect could be better understood. Particularly in extensive pastures with higher levels of organic soil and soil pH values that are much lower than grazed pastures on most mineral soils. Further work to quantify emissions will therefore aid in refining estimates and understanding the underlying variability across the pasture types. Additionally, work to investigate further the factors that cause the variability in emissions would be beneficial. This information can then contribute to potential disaggregation of EFs and improving national greenhouse gas inventory reporting as well as CF.

3.6 Reference list

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Chapter 4 Development of a carbon calculator accounting for the altitudinal variability of lamb production systems

4.1 Introduction

Lamb production results in the emission of greenhouse gases (GHGs), therefore contributes to climate change. A key gas produced is methane (CH_4) from enteric fermentation (Miljan et al., 2021), but also nitrous oxide (N_2O) from deposition of excreta onto grassland and fertiliser use (Broucek, 2018). Carbon dioxide (CO_2) is also released due to on-farm activities, for example use of fossil fuels in farm vehicles (Grossi et al., 2019). In the context of Net Zero targets to reach Net Zero by 2050, pressure is increasing on the sector and agriculture as a whole to reduce GHG emissions. Effective measuring of emissions and identifying where negative environmental impacts can be reduced are key to achieving future sustainability (CIEL, 2022).

An important methodology for assessing the carbon (C) impact of a product, that has been applied to agriculture, is life cycle assessment (LCA) (Haas et al., 2000). It can provide an estimation of the environmental cost and resource use of a farm and its product outputs, as well as aid comparison of different systems. Use of LCA provides evidence that can contribute towards decision making in environmental management and environmental impact reduction options (Caffrey and Veal, 2013). Carbon footprinting (CF) is an analysis based on LCA principles and has been utilised for a variety of sheep production systems, including lamb, wool and milk as products. These analyses have been conducted globally with a range of results, including in the United Kingdom (UK), New Zealand, Australia, Africa and across the Mediterranean (Batalla et al., 2015; Ibidhi et al., 2017; Jones et al., 2014b; Wiedemann et al., 2016). Throughout all studies, the greatest emissions burden of lamb production derives from CH_4, primarily due to enteric fermentation (Bhatt and Abbassi, 2021; Cottle and Cowie, 2016). In terms of UK-specific studies, a 2014 Wales based study gives average values of 10.9 kg CO_2eq kg liveweight^{-1} (LW) for lowland lamb, 12.9 kg CO_2eq kg LW^{-1} for upland lamb and 17.9 kg CO_2eq kg LW^{-1} for hill lamb (Jones et al., 2014b). An older Wales-based study gave a range of 8.1-31.7 and 20.3-143.5 kg CO_2eq kg LW^{-1} for two case study farms (one upland mixed sheep and cattle farm and one hill mixed sheep and cattle farm) (Edwards-Jones et al, 2009). However, the Intergovernmental Panel on Climate Change (IPCC), which produces guidelines for agricultural GHGs estimation, has produced a refinement to guidelines since these studies were published (IPCC, 2019).

LCA by its nature involves uncertainty. The inherent uncertainty related to LCA is further complicated by the high level of variability, both spatially and temporally, within agricultural systems. Variability of emissions themselves due to differing environmental conditions across grazing pastures and variability in farm-specific activity data, for example level of farm inputs (e.g., fertiliser use), animal breeds and diets have all been shown to affect CF results (Ledgard et al., 2020). This may result in reporting of inaccurate results and incorrect conclusions, dependent on the initial question asked. It is difficult to develop a representative farm production system due to the variety of farms and management systems present in different countries and in different regions of the same country. Uncertainty can arise due to methodologies chosen within the model as well as limitations in the input data, in terms of production characteristics and differences in management between systems. Variability in animal populations may also be present, in both a spatial and temporal manner (Chen and Corson, 2014).

Other areas where results can be influenced within LCA studies include selection of a functional unit and the allocation of environmental impacts to the different products that may be generated by livestock systems. Use of different functional units can make comparison of results of different studies problematic, or even impossible. The issue of allocation of impacts to co-products is particularly important for lamb production systems, as many produce both lamb and wool (Wiedemann et al., 2015). Additionally, use of different impact categories and methods, for example global warming potential (GWP) and eutrophication potential, can hinder effective comparison (Röös and Nylinder, 2013). The breed of sheep and type of enterprise involved affects final quantities and quality of the lamb and wool outputs, increasing the effect of differing allocation options on the final footprint dependent on what is chosen (Cottle and Cowie, 2016). However, LCA still represents an effective approach for measuring the environmental impacts of a production system, allowing identification of areas where impacts can be mitigated.

Until recently, there were no peer-reviewed review articles summarising and comparing the latest LCA research on lamb production systems. This has made it difficult to draw conclusions regarding the environmental performance of the sector, from both an academic and policy perspective. However, there is now one article summarising recent LCA application to such systems (Bhatt and Abbassi, 2021). Use of LCA has mostly focussed on climate change impacts via quantification of production related GHG emissions and generally enteric CH_4 was the key emission source in lamb production footprints (comprising 50-75% of system emissions). The system boundary has mostly been the restricted to emissions produced on-farm with a lack of assessment of post-farm emissions, for example emissions related to processing and transportation of products. It was noted that the diversity of methodology and results within lamb production system LCA studies made comparison of studies difficult. Additionally, there are large differences in farm management practices and efficiencies of production within this sector.

Another consideration is that production of lamb is associated with a variety of wider system benefits, making these systems unique (Batalla et al., 2015). For example, the presence of ruminant grazing has encouraged grassland conservation and in turn may encourage continued sequestration of C within some grasslands, up to approx. 1 t CO_2 ha^{-1} $year^{-1}$ (Janssens et al., 2005). Livestock are synonymous with the appearance of certain landscapes, especially in the UK, and grassland-based systems are linked to a variety of ecosystem services including regulating and cultural services (Ripoll-Bosch et al., 2013). Despite this, a standardised methodology of assessing these additional benefits is not present within LCA and therefore can result in a conflict between GHG mitigation and other environmental aims. The previously mentioned review of lamb production LCA includes the quantification of system benefits alongside wider environmental implications as a recommendation for developing lamb production LCA (Bhatt and Abbassi, 2021).

UK lamb production systems are often operated on a tier-based system, taking advantage of a variety of altitudes to optimise production. This approach enables exploitation of different breed characteristics suited to their environmental and climatic surroundings (National Sheep Association, 2021). The hardier hill breeds survive in hill areas with older hill ewes moved to upland areas and crossed with upland breeds to produce mule lambs. The ewe crosses are then bred with a lowland terminal sire breed, resulting in larger lambs. Animals can then be sent to the lowland and sometimes upland areas to be fattened, before being sent for slaughter (Sargison, 2008).

This stratified system results in lamb production systems being highly variable as animals graze different pastures with varying forage and environmental factors known to affect emissions. For example, forage consumed by the animals is known to affect enteric CH_4 and environmental conditions such as rainfall are known to affect excretal N_2O (Fraser et al., 2015; Mancia et al., 2022b). There is a growing body of evidence that suggests absolute emissions vary across altitudes in lamb production systems due to these different environmental and forage characteristics. In a recent study, measurement of N_2O fluxes from sheep urine on an upland and hill site resulted in lower urine N_2O-N EFs (EF_{3prp}) than UK and IPCC default factors (Marsden et al., 2019). This was linked to differences in nitrification rates in these sites due to acidic conditions. A similar study assessing N_2O emissions from sheep excreta showed negligible EF_{3prp} in both lowland and hill pastures with higher cumulative N_2O emissions in the lowland pasture than the hill pasture (Mancia et al., 2022a). For enteric CH_4, it is known that diet influences production of the gas from the rumen (Shibata and Terada, 2010). The frequent movement of animals across pastures exposes animals to a variety of forage species with different characteristics. There has been evidence to suggest different CH_4 output from different forages, which may significantly influence lamb footprints (Archimède et al., 2018; Zhao et al., 2017, 2016). In the case of upland and hill sheep, heather species (*Calluna vulgaris*) may form a portion of their diet (Fraser et al., 2009). These forages are known to contain condensed tannins and recent studies suggest supplementation of condensed tannins resulted in a 51-60% decrease in rumen CH_4 output from Bapedi sheep (Ng'ambi and Brown, 2022; Ropiak et al., 2016). There were no recorded ill effects to the sheep and the antioxidant level of the resulting meat product was greater than non-supplemented sheep.

This variation in emissions output could then result in variation within modelling emission factors (EFs). The two EFs of most concern within this study are i) Ym, also known as the methane conversion factor, which is the proportion of gross energy intake that is expelled as CH_4 and is used for calculation of enteric fermentation CH_4 and ii) EF_{3prp}, the emission factor used for calculation of N_2O emissions that arise as a result of urine and dung deposition from grazing animals to pasture (kg N_2O-N) (IPCC, 2019). Despite the growing evidence, IPCC 2006 and 2019 guidelines do not contain any disaggregation, which could be by altitude in lamb production systems, when considering livestock GHG related EFs. A blanket value for both Ym and EF_{3prp} is used across the whole animal population, highlighting the need for progression in this area. In New Zealand (NZ), a country specific EF_{3prp} value disaggregated by altitude and livestock type has been developed within the inventory (van der Weerden et al., 2020).

In comparison to beef and dairy systems, lamb production remains an area of agriculture with fewer published CF and LCA studies quantifying its environmental impact (Lal et al., 2022). Capturing the variability of pasture types, altitudes, and environmental factors within EFs as well as the frequent movement of animals is a key challenge when applying CF to this sector and has not yet been assessed. This chapter aims to detail development of a disaggregated LCA model that allows differentiation across animal and farm management characteristics, and EFs, by altitude. Site-specific EFs were generated from data collected within Chapter 3 of this thesis and are included in the model. Model development is followed by an assessment of how the addition of disaggregation in the model may affect final farm footprints and annual GHGs using input data from Bangor University's commercial farm at the Centre for Hill and Upland Management (CHUM), Abergwyngregyn, North Wales (UK).

Finally, site-specific inputs are used to determine what aspects of input data have a greater effect on model outputs, therefore highlighting where future efforts to retrieve additional data should be focussed.

The results of this chapter are intended to highlight where improvements could be made in lamb footprinting methodologies. The experiment-based values used are not intended to be taken as definitive footprints of lamb production at different altitudes. Instead, they are intended to show how emissions may differ through space and time within stratified farm systems and how these differences may be better represented within footprinting tools.

4.2. Model methodology development

4.2.1 Overview of model

4.2.1.1 Model aims

The overarching aim throughout model development, was to create a structure that would allow differentiation in GHG (CH_4, N_2O and CO_2) emissions by altitude and time (month) in lamb production systems – reflecting the frequent movements of animals across the year within a typical sheep farm. The (Microsoft Excel spreadsheet) model was created with the intention of being used as a footprinting tool at a farm level. Additionally, options to choose different EFs based on IPCC Guidelines for National Greenhouse Gas Inventories from 2006 (IPCC, 2006) or 2019 (IPCC, 2019) as well as those based on measurements made in this study from pastures at different altitudes was included in the model. This allowed assessment of the effect of EF choice and disaggregation on the resulting lamb footprint. Given the high uncertainty and lack of replicates in relation to the experimental EFs (Chapter 3) and input data, these footprints are not intended to be conclusive. However, analysis is included to show how site-specific inputs may affect final footprints.

4.2.1.2 Calculation of footprint

The model was completed via a desk-based study, adapting existing Microsoft Excel models used in previous farm footprinting (Mazzetto et al., 2020; Soteriades et al., 2019; Styles et al., 2018). The core model was initially used for cattle production systems but was altered to adhere to sheep-specific information.

The cohorts considered within the model include adult ewes (>1 years old), adult rams (>1 years old), replacement ewes (>1 years old), male lambs (<1 years old) and female lambs (<1 years old). Each cohort had a separate worksheet in the model and within each worksheet a calculation table was present for each altitude (lowland, upland, and hill) and an additional table for housed animals (if any are present on the farm). A monthly stocking diary page was added similar to that of Hyland et al. (2016), detailing monthly animal inputs and outputs per altitudinal location and cohort. This structure was what allowed the planned disaggregation by altitude as per the overarching aim.

GWP were reported according to IPCC Fourth Assessment Report values, to maintain consistency with the UK inventory approach in the context of improving UK and Welsh lamb production footprinting. These are 1 kg CH_4 = 25 kg CO_2 equivalents (CO_2eq) and 1 kg N_2O = 298 kg CO_2eq based on the warming effect of the gases in comparison to CO_2 over a 100-year time frame (Pachauri et al., 2015). The IPCC Fifth and Sixth Assessment Report values (1 kg CH_4 = 28, 1 kg N_2O =265; 1 kg CH_4 = 27, 1 kg N_2O = 273) were included within the model development, although only used for analysis later in this thesis (see Chapter 5) (IPCC, 2021).

4.2.1.3 Farm inputs

Bangor University's commercial farm (the Centre for Hill and Upland Management, CHUM) was used as a case study for the model. Information regarding farm inputs and outputs were collected for use within the altitudinally disaggregated model. The centre is a field research facility, encompassing 252 hectares of diverse landscapes ranging from sea level to mountain areas of the Carneddau mountains. The farm includes a commercial sheep operation, historically consisting of up to 1650 Welsh Mountain ewes that graze across the altitudinal gradient present on the farm (Bangor University, 2022). Further information on the case study farm is summarised in Table 4.1.

Table 4.1 Description of the case study farm, Centre for Hill and Upland Management (CHUM)

Farm Characteristic	**Information**
Location	Centre for Hill and Upland Management (CHUM), Henfaes Research Centre, Abergwyngregyn, Gwynedd, Wales.
Grazing area (ha)	Lowland (improved) 52.9 ha, upland (unimproved/semi-improved) 148.8 ha and hill 490 ha (common grazing).
Number of ewes	1237 – 1554 (variable throughout the year)
Lambs born	965
Lambing average lambs ewe^{-1}	0.78
Store lambs sold	565
Culled ewes sold	261
Concentrate use kg year^{-1}	4390
Diesel l year^{-1}	2752
Petrol l year^{-1}	554
Nitrogen kg N year^{-1}	757
Phosphorus kg P year^{-1}	212
Potassium kg K year^{-1}	271
Lime kg year^{-1}	1200
Herbicide kg active ingredient year^{-1}	32

Farm data were collected via a questionnaire, the inputs of which were discussed in a formal meeting with farm staff. The questions involved determining further information regarding animal characteristics, e.g., animal numbers, LW, monthly stocking numbers, lambing average, scanning percentages and movements across different pastures. Additionally, further farm information was collated including concentrate use and which animals were fed concentrate as well as fertiliser type and quantity used, fuel use, manure management and wool production. Data were collected for the year 2020 and encompassed the full 12 months of the production year (April – April).

4.2.1.4 Farm characteristics

CHUM operates over a total area of 284.5 ha (192.8 ha permanent grassland, 5.2 ha temporary/reseeded grassland) with additional grazing rights on common land area totalling 446 ha, allowing grazing of up to 1880 sheep in total. The farm operates across diverse landscapes ranging from sea level to mountain pasture (>800m a.s.l) on the Carneddau mountains (Bangor University, 2022). The primary operation is the production of store lambs for sale. Breeding ewe numbers ranged from 1237-1554 throughout 2020 with ram numbers ranging from 33 to 55 during tupping. Ewe and ram store lambs were sold mostly between August and September with some sold in November and December. Culled ewes and rams were sold across the same months. Animals were grazed across lowland and upland sites all year round, with ewes and ewe lambs only grazing the common land from April to October. A small number of ewes (200) were housed for 6 weeks in February.

Other farm inputs included a small amount of straw and silage being fed to animals from January – April in the lowland only. Concentrates are used minimally with 4.39 tonnes fed across the farm only in the lowland during March. In terms of fertiliser and energy, Henfaes applies fertiliser in the lowland areas only with the upland area being unfertilised for over 20 years. It is mostly different variants of NPK fertilisers with a small amount of ammonium nitrate used in one field. Additionally, agrochemicals in the form of Thistle X (Dow AgroSciences, Cambridge, UK) and Pasture Master (Nufarm, Wyke, UK) are applied. Electricity use is minimal and not recorded. Red and white diesel and petrol was 2105, 467 and 554 litres respectively for farm activity as well as research. However, it was not possible to distinguish between the activities from the information given, so all consumption was attributed to the farm within the model.

4.2.1.5 Functional unit

The functional unit used in the model is kg LW leaving the farm. LW refers to the total weight of animals that are produced and sold by the farm. This unit was chosen due to the main output of the farm being store lambs, therefore animals are not sent straight to slaughter. It is possible within the model to obtain kg carcass weight (CW) leaving the farm. This is the fraction of LW present as usable carcass, determined by the national average killing out percentage for sheep – here defined as 46 and 44% for adult sheep and lambs respectively as per the UK national GHG inventory (Brown et al., 2021). Results for CHUM displayed in this chapter were expressed only in kg LW and (kg farm^{-1}) for reasons stated above.

4.2.2 System boundary

The system boundary of the model (Figure 4.1) is defined as cradle to farm gate. The model therefore includes all emissions from an on-farm activity basis, and those arising upstream from production and transport of farm imports but not emissions that arise after the product(s) leave the farm gate; for example, downstream transport and processing. The emissions sources included are:

1. Emissions relating to production and transport of farm inputs and the use of these inputs, such as fertiliser and feed on farm.
2. Emissions relating to the animals on farm, for example enteric fermentation CH_4, manure management CH_4 and direct and indirect N_2O.

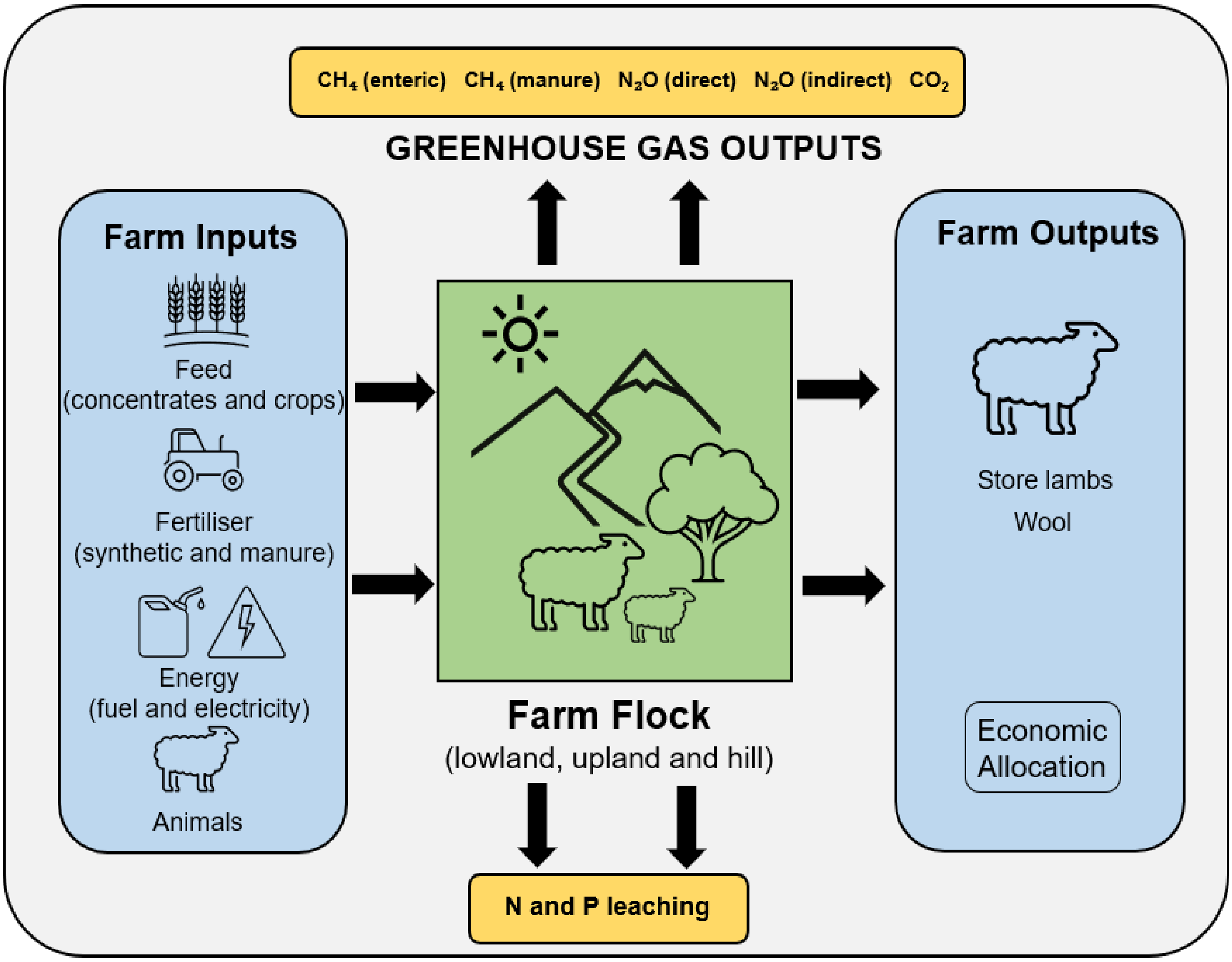

Figure 4.1 System boundary of the LCA model

GHG outputs are defined as CH_4 from enteric fermentation and manure management, direct and indirect N_2O and CO_2 arising from energy use on farm. Farm outputs are store lambs and wool; although in the case of CHUM wool remains a minor output of the farm, therefore it was not deemed necessary to allocate emissions to wool as a product. It is possible to allocate emissions within the model via economic allocation if required.

4.2.3 Disaggregation of model

4.2.3.1 Overview

The model was structured to allow disaggregation by altitude of various factors within the production system. A summary of the factors included for disaggregation can be seen in Table 4.2.

Table 4.2 Summary of factors disaggregated by altitude present in the model

Factor	
Livestock activity data	Monthly number of animals by altitude from each cohort Average weight of animals by altitude from each cohort (kg head^{-1}) Average weight gain partitioned by month across the year for growing animal cohorts (kg head^{-1} month^{-1})
Forage composition and intake (enteric CH_4 emissions)	Representative forage digestibility by altitude (digestible energy (DE) %) Gross energy intake (GEI, MJ day^{-1}) Energy coefficients within net energy for activity calculations Methane conversion factor, Ym (% GEI emitted as CH_4)
Excretal N_2O emissions	EF for estimation of N_2O from deposition of animal excreta, EF_{3prp} (kg N_2O-N kg N^{-1} input) Nitrogen (N) excretion (linked to altitudinal DE% and crude protein (CP) %) Manure management system Forage crude protein by altitude (CP%)

A flow diagram, detailing the main data inputs, calculations, and areas of disaggregation within the model can be seen in Figure 4.2. Additionally, the links between the data inputs and the different calculations within the model are highlighted. For example, forage digestibility feeds into ratio of net energy available in diet for maintenance to digestible energy consumed (REM) and ratio of net energy available for growth in a diet to digestible energy consumed (REG) equations, which in turn affect final calculation of CH_4 from enteric fermentation (IPCC, 2019).

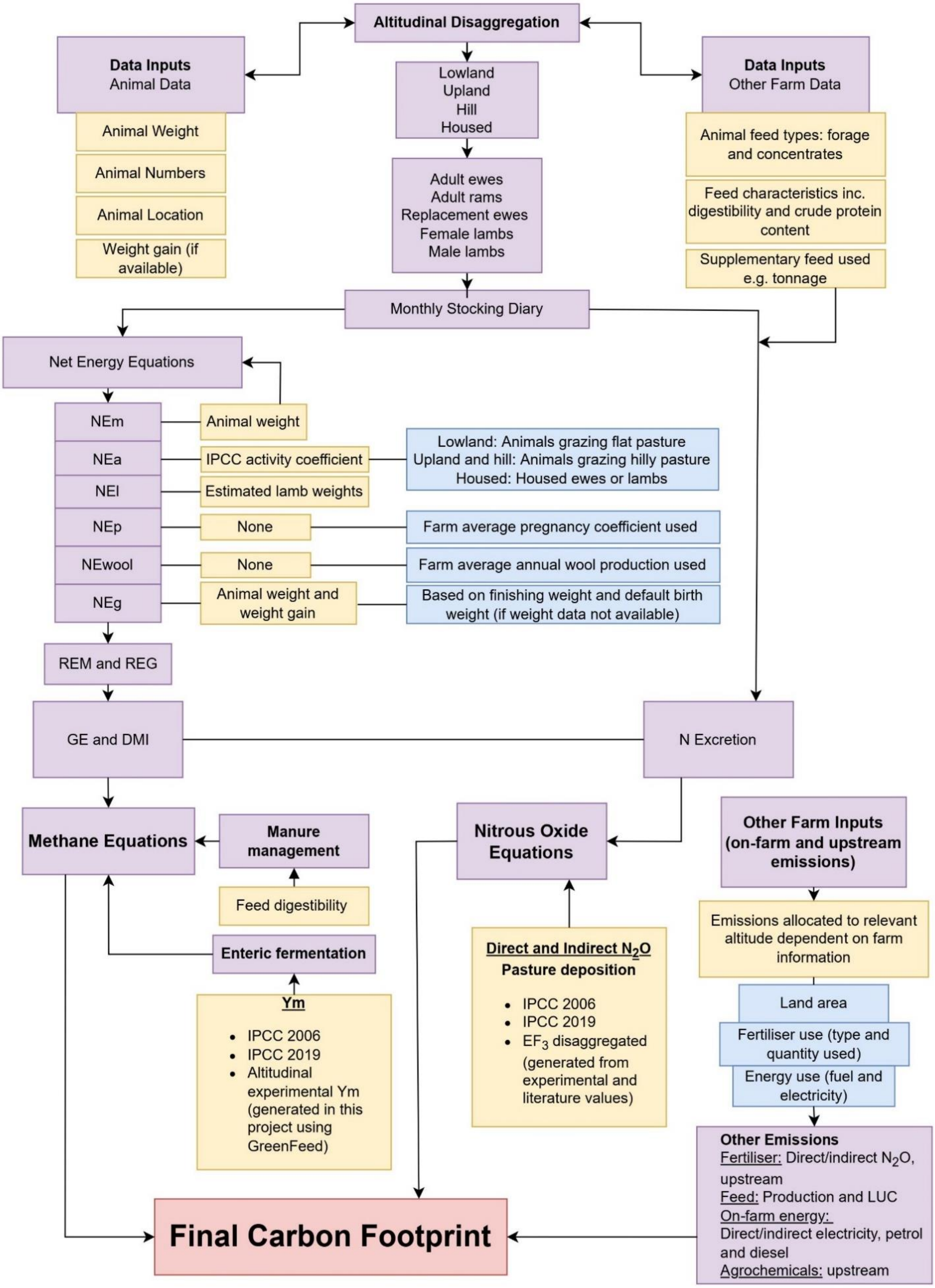

Figure 4.2 Schematic flowchart detailing model structure, data inputs and areas of disaggregation *(Yellow shading is related to disaggregated inputs, blue further disaggregation information and purple the main calculations and sheets within the model).*

4.2.3.2 Stocking diary

Functionality for several of the above factors was made possible by addition of a monthly stocking diary to the model, comprising a table allowing input of farm specific data on animal births, deaths, and movements. A key improvement gained from addition of the stocking diary was to allow flexibility, as different farm enterprises move their sheep in different ways across the year. For example, some farms keep stock on the upland and hill plots all year round, whereas others may remove them in certain months. Some animals move between the altitudes and others may remain in each altitude for the duration of their productive lives. The stocking diary allowed this functionality as the number of animals from each cohort and each pasture type in a given month of the year could be added. Each cohort was split by altitude and rows to input the number of animals that were bought, sold and in stock for each cohort and altitude added. Additionally, a row was added for specification of the average weight and the average weight gain for each subdivided cohort of growing animals.

An additional table was added to the stocking diary, consisting of the average start, and finishing weight (kg) for each cohort, and a calculation cell allowing the average weight gain (kg) to be determined with reference to start, weaning, and finishing weights during average animal lifetimes. CHUM was unable to provide lamb birth weight because this is not routinely measured across the flock, therefore a default value of 4 kg was chosen in line with Wales-specific information (Hybu Cig Cymru, 2018). However, it is possible to specify lamb birth weight within the model when specific data are available for a farm. As CHUM did not regularly weigh animals, default weight values were also applied to sold animals from the literature (Hybu Cig Cymru, 2018).

4.2.3.3 Disaggregation of forage digestibility and crude protein content

Given the relationship between forage digestibility and CH_4 output, it was deemed necessary to disaggregate forage digestibility across altitudes (Eugène et al, 2021). This was also decided for forage crude protein content owing to its influence on N excretion, and thus N_2O emissions (Marsden et al., 2020). Disaggregation of forage digestibility and crude protein content between the altitudes was enabled by linking the lowland, upland, and hill calculation matrices in each cohort sheet to different cells related to these two forage characteristics. An 'IF' function was then added to the cell, allowing IPCC 2006, IPCC 2019, and UK specific default/site-specific values to be included via a drop-down menu on the initial model introduction page.

Site-specific values refer to forage data collected during pasture analysis completed in Chapter 3 of this thesis and collated with other CHUM measurements where data allowed. Lowland digestibility is based on field experiments across two years (in spring/summer) and is similar to UK-derived literature values (MAFF, 1977). The upland digestibility is based on a September and late November collection, alongside data collected in other field experiments on the same site (in preceding years) (Williams, 2020). The hill pasture forage data are based on measurements taken in September and due to project constraints only one experiment took place on this pasture. This data is therefore less robust, although is within the range suggested by other reports for hill forages (HFRO, 1979). This was added to aid in assessing the impact of different methodologies on the final footprint. Any calculation that involved either forage characteristic was then linked to the 'IF' function cell, for example GEI calculations.

IPCC forage digestibility data for both IPCC 2006 Guidelines (IPCC, 2006) and IPCC 2019 Guidelines (IPCC, 2019) were added to the model. IPCC forage digestibility are presented as a range in the inventory reporting guidelines therefore the min, median and max values were included, although when modelling the IPCC footprints, the median values were used. IPCC digestibility value for pasture/mixed diet animals and animals fed low quality forage were added, with lowland animals linked to the pasture/mixed diet value and upland/hill animals linked to the low-quality forage value. The UK-specific value used for crude protein (CP%) is based on the UK average used within the UK inventory (S Anthony 2022, personal communication, 4th May). The values used are summarised in Table 4.3.

Table 4.3 Summary of forage characteristics (forage digestibility and crude protein) used in the model. IPCC forage digestibility values refer to the median of the range presented in the guidelines.

Data source	Lowland value	Upland value	Hill value
Forage digestibility (DE%)			
IPCC 2006	65	50	50
IPCC 2019	68	50	50
Site-specific	73	67	63
Crude protein (CP%)			
UK-specific	14	14	14
Site-specific	18	15	13

4.2.3.4 Disaggregation of emission factors

Different cells were added for each altitude for the EFs included in the disaggregation (Ym and EF_{3prp}). These EFs were chosen for disaggregation because they are linked to the main sources of CH_4 and N_2O from lamb production systems. Additionally, as discussed above, evidence suggests a difference in CH_4 dependent on the grazing (pasture quality) available and differences in N_2O across pastures dependent on different environmental factors.

To allow easier comparison of disaggregation of footprint results with IPCC default footprint results, an 'IF' function was added to the cell that the different sections of each cohort were linked to. This 'IF' function allowed selection of the IPCC 2006 and IPCC 2019 default EFs, or site-specific experimental values. This was enabled via a drop-down menu in the introduction section of the model, where the user can select the data source. Changing between IPCC 2006, 2019 and site-specific values within this drop-down menu results in the corresponding EF changing within the cell linked to the emissions calculation for that altitude and cohort. The site-specific value option includes a summary of data collected in field trials throughout this thesis (Ym calculated via CH_4 emission measurements using the GreenFeed (GF) system and EF_{3prp} calculated via N_2O measurements using the static chamber method and real sheep urine/dung) combined with data from relevant published literature from similar sites.

4.2.4 Net energy calculations

The calculations used for determination of net energy requirements per animal followed the IPCC Tier 2 methodology from the IPCC Guidelines for National Greenhouse Gas Inventories. Both the original 2006 methodology and the 2019 updated methodologies were incorporated into the tool for comparison. UK specific values were incorporated as and when they were appropriate (Brown et al., 2021).

Net energy requirements were displayed initially in a megajoule (MJ) per animal per day format for all net energy calculations. Once the gross energy requirements were calculated for each cohort, these were multiplied by the number of animals in each cohort in each month in each altitude section. The energy calculations for each cohort are displayed in Table 4.4.

Table 4.4 Summary of relevant factors included in net energy equations for main cohorts considered within the model

Cohort	Net energy equation factors
Adult Ewes	$NE_m + NE_a + NE_l + NE_p + NE_{wool}$
Adult Rams	$NE_m + NE_a + NE_{wool}$
Replacement Ewes	$NE_m + NE_a + NE_g + NE_{wool}$
Female Lambs	$NE_m + NE_a + NE_g + NE_{wool}$
Male Lambs	$NE_m + NE_a + NE_g + NE_{wool}$

**(where NEm is the net energy required for maintenance, NEa the net energy required for activity, NEl the net energy required for lactation, NEp the net energy required for pregnancy, NEg the net energy required for growth and NEwool the net energy required for wool growth)*

Adult ewes are the only cohort presumed to be pregnant and lactating throughout the year. Replacement ewes are defined as ewes >1 year old but have not joined the farm breeding cycle. This cohort was included for completeness but due to CHUM not collecting sufficient data to allow allocation of animals to this cohort, all ewes >1 year old were added to the adult ewe cohort.

4.2.4.1 Net energy for maintenance

The net energy for maintenance (NEm) required by each animal cohort was calculated by use of IPCC 2019 calculation 10.4 (IPCC, 2019). No further changes were made to the calculation.

4.2.4.2 Net energy for pregnancy

Net energy for pregnancy (NEp) was applied to adult ewes only using IPCC 2019 calculation 10.13 (IPCC, 2019). It was included only in the months that ewes are generally expected to be pregnant in the system present at CHUM (December-March, with lambing in April). While gestation periods vary in sheep, to simplify model inputs it was assumed that ewes would be pregnant for four months based on UK farmer guidance (AHDB, 2016a).

Due to the equations having a constant that is weighted to the annual average as a function of the maintenance calculation, it was decided to calculate the NEp over the course of the year and then partition the total between the four months that ewes are presumed to be pregnant. The total was also multiplied by the percentage of animals that undergo gestation, as per IPCC guidelines for sheep specifically. This was done to preserve the monthly aspect of the model. As CHUM has a lambing average of 0.78 lambs per ewe, the coefficient for single births was used throughout as advised within IPCC guidelines (IPCC, 2019).

4.2.4.3 Net energy for lactation

Net energy for lactation was applied to adult ewes only because this cohort is the only one involved in active feeding of lambs. IPCC calculation 10.10 was used, assuming milk production is unknown as Welsh/UK systems do not tend to produce milk for food (IPCC, 2019).

Ewes were assumed to be lactating for 91 days (3 months) beginning in April (or the time between birth and weaning). The 91 days was based on average weaning times presented in farmer guidance (Hybu Cig Cymru, 2018). To account for the monthly aspect of the model, net energy for lactation was applied only to the months ewes are expected to be lactating (April, May, and June). If a farm varies from this average, months lactating can be manually changed. The calculation was then divided by 91 rather than the 365 days present in IPCC guidelines, in order to be applied at a relevant daily net energy requirement.

4.2.4.4 Net energy for growth

Net energy for growth was applied only to the replacement ewes and male and female lamb categories, using IPCC 2019 equation 10.7 (IPCC, 2019). No alterations were made to the calculations; however, a monthly weight and weight gain was calculated within the stocking diary to reflect changes in weight to a greater resolution than an annual average. This was achieved by use of default birth, weaning weight and average finishing weights for each cohort (derived from literature and industry data) to partition weight gain across the 12 months of the year.

The weight at weaning was taken from industry benchmark data for the different altitudes. Growing animals were assumed to have reached the average finishing weight by 9 months of age as per the UK inventory (AHDB, 2016; Brown et al., 2021).

4.2.4.5 Net energy for wool

Net energy for wool was applied to all cohorts based on IPCC calculation 10.12, with no further changes made. Annual wool production was estimated via information provided by the farm manager at CHUM. This value was halved for male and female lambs in line with wider model guidance (Clark, 2011).

4.2.4.6 Gross energy calculations

Gross energy requirement was split between grazed forage, concentrates and hay/straw depending on the cohort and time of year. GEI from concentrates was estimated by estimating daily dry matter intake (DMI) (kg) as a function of body weight across each altitude and then multiplying by the feed energy, digestibility and amount of concentrate consumed by said cohort (provided by the farm).

The remaining GEI was assumed to derive from forage consumption. This was calculated via IPCC equation 10.16 (IPCC, 2019), minus the GEI determined to be from concentrates. These were then summed to give the GEI total. DMI was then calculated by dividing the total GEI by the IPCC defaults feed energy of 18.45MJ (IPCC, 2019).

4.2.5 Nitrogen balance and excretion

A summary of how the N balance and the resulting excretion was calculated is given below. The N excretion is defined as the N intake from feed by the animal minus any N retained in body tissues or secretions as follows:

$$N_{ex} = N_i - (N_{rm} + N_g + N_{wool})$$

Where:

N_{ex} = N excretion (k g animal^{-1} day^{-1})

N_i = N intake (k g animal^{-1} day^{-1})

N_{rm} = N retention (milk) (k g animal^{-1} day^{-1})

N_g = N retention (growth) (k g animal^{-1} day^{-1})

N_{wool} = the N retained in wool (k g animal^{-1} day^{-1})

The same process was followed for each cohort dependent on the applicable N retention factors. As the IPCC methodologies did not include calculations for N retention for sheep, Agriculture and Food Research Council (AFRC) equations were used in line with the UK inventory methodologies (Cottrill and Alderman, 1993; Brown et al., 2022).

4.2.5.1 Nitrogen intake

N intake of the animals was calculated via IPCC equation 10.32, resulting in an aggregate value for N derived from concentrates, grass, milk, or other forage dependent on the cohort (IPCC, 2019). Calculations were then linked to the relevant altitude specific CP content in forages or CP content of milk, listed within the references page. True protein content of milk was taken as 4.89% as described in equation 89 of the AFRC guidelines (Cottrill and Alderman, 1993).

4.2.5.2 Nitrogen retained in milk

N retention in milk was calculated as directed in equation 79 of AFRC guidelines (Cottrill and Alderman, 1993).

As milk yield is not measured as standard on farm, a UK derived default value of 100l per ewe per lamb for milk yield was used. This default was based on data from UK medium sized hill sheep rearing a single lamb, therefore was deemed representative for the footprinted system (Clark, 2011). This value was used to estimate N intake from milk for lambs as described in Section 4.2.5.1.

4.2.5.3 Nitrogen retained in wool

N retention in wool was calculated as per AFRC equation 79 (Cottrill and Alderman, 1993).

4.2.5.4 Nitrogen retained in body tissue

N retention in body tissue (growth) was calculated as per AFRC equation 98 (for males and castrates) and 99 (for females) (Cottrill and Alderman, 1993).

4.2.6 Calculation of greenhouse gas emissions

The IPCC Guidelines for National Greenhouse Gas Inventories methodologies were used for calculation of GHGs. Both the 2006 and 2019 EFs for enteric CH_4 and excretal N_2O were included alongside experimental values that were measured in Chapter 3 of this thesis. Total emissions were calculated monthly and then summarised per year. This allowed identification of emission intensive months (expected around lambing periods) as well as overall annual footprints.

4.2.6.1 Methane emissions

The methane conversion factor, or Ym value, describes the extent that energy from feed is released as CH_4. Specifically, it is the fraction of gross energy intake that is released as CH_4 (IPCC, 2019). The Ym values included within the model are summarised in Table 4.5. Upland and hill values are not applicable to IPCC data sources due to the lack of disaggregated values within the methodology. Experimental values refer to Ym values calculated in Chapter 3 using the GF system across lowland, upland and hill pastures.

Table 4.5 Ym values used within the model

Data source	Lowland value	Upland value	Hill value
IPCC 2006	6.5% (4.5% for lambs)	N/A	N/A
IPCC 2019	6.7%	N/A	N/A
Experimental values	11.1%*	6.5%	4.2%

**The lowland experimental Ym value is a combined mean Ym derived from two measurement periods in 2020 and 2021.*

For young animals on a milk-based diet, the IPCC recommends that a Ym value of zero is used. The age at which animals start becoming less reliant on milk is variable dependent on the pasture quality each year. However, generally lambs rely more on grass than milk from 8 weeks of age. For this reason, it was assumed that lambs 8 weeks old or less predominately consumed milk therefore a Ym value of 0 was used for those months only (AHDB, 2020; IPCC, 2019).

CH_4 generated from sheep manure management practices were calculated using IPCC equation 10.23. Lowland, upland, and hill animals were assumed to deposit 100% of their excreta to pasture. For the small number of animals that were housed (only relevant for the month of February), EFs for manure storage were used.

4.2.6.2 Nitrous oxide emissions

Direct N_2O emissions from sheep excretal deposition to soil were calculated as per IPCC equation 11.1 (IPCC, 2019). The EF_{3prp} values included within the model are summarised in Table 4.6. Values for upland and hill pastures are not available within IPCC Guidelines. Experimental values refer to EF_{3prp} values measured in Chapter 3 across lowland and upland pastures. The experimental value for the hill pasture was measured during previous work at CHUM (Marsden et al., 2019).

Table 4.6 Sheep EF_{3prp} values from different sources applied to the three altitudes considered within the model. Hill experimental value taken from Marsden et al. (2019)

Data source	Lowland value	Upland value	Hill value
IPCC 2006	1%	N/A	N/A
IPCC 2019	0.3%	N/A	N/A
Experimental values	0.36%	0.12%	0.08%

Indirect N losses due to volatilisation and leaching were calculated as per IPCC equation 10.26 and 10.27. EF_4 and EF_5 from both IPCC 2006 and 2019 were added for comparison.

4.2.7 Other emissions burdens

4.2.7.1 Fertiliser use

Direct and indirect emissions from fertiliser use were calculated using IPCC equations 11.1, 11.10 and 11.11 using default emission factors (IPCC, 2019, 2006).

Fertiliser upstream emissions were calculated by multiplying the quantity of each type of fertiliser applied by standard EFs related to the emissions related to the production and transport of that type of fertiliser (Ecoinvent v 3.5) (Wernet et al., 2016).

4.2.7.2 On-farm energy and agrochemicals use

Emissions related to on-farm energy use were derived from input data and standard EFs taken from the Ecoinvent v.3.5 database (Wernet et al., 2016). The quantity of diesel (l), petrol (l) and electricity (kWh) used across the farm were multiplied by the corresponding EF for the fuel source (direct and indirect emissions). The same process was followed for agrochemicals after quantifying their active ingredient contents.

4.2.7.3 Concentrate use

The emissions related to concentrate were calculated by multiplying input data with standard EFs for CO_2eq arising from production of the concentrate and any land-use change that results from production (Wernet et al., 2016).

Most of the concentrate use was applied to the ewe category as indicated by the farm. The remaining concentrate was spread across the cohorts on a pro-rata basis based on animal numbers. In the case of CHUM, concentrate was mostly consumed in March with minimal use throughout the rest of the year.

4.2.8 Model outputs

Total GHG emissions (t CO_2eq) across the farm were summarised on both a monthly and an annual basis. Total GHGs were split into six categories: enteric CH_4, manure management CH_4, N_2O emitted from pasture deposition of excreta, fertiliser application, concentrate use, and fertiliser upstream and on-farm energy use related emissions.

The CF was then determined by summing the monthly GHGs and dividing by the farm production (kg LW leaving the farm each month). This allowed generation of an annual CF. There were three footprints in total created: a non-altitudinally disaggregated (NAD) footprint, a disaggregated footprint, and a site-specific footprint.

In the NAD footprint, farm inputs were not disaggregated, and the same EFs across all three altitudes were used. The disaggregated footprint used IPCC default values for EFs and forage characteristics but had animals disaggregated (in terms of numbers and LW) between the three production tiers (lowland, upland, and hill). The site-specific footprint used EFs and forage characteristics that were specific to the farm and generated from experimental data in Chapter 3. The NAD and disaggregated footprints are presented using both IPCC 2006 and IPCC 2019 inventory guidelines to determine potential differences between the 2006 guidelines and the 2019 refinement. The site-specific footprint was compared to the NAD and disaggregated footprints using IPCC 2019 only as the 2019 guidelines are the most recent therefore the most valid comparison.

4.3 Results

4.3.1 Non-altitudinally disaggregated footprint results

The baseline CF results were calculated as 14.5 kg CO_2eq kg LW^{-1} leaving the farm and 13.6 kg CO_2eq kg LW^{-1} leaving the farm, following IPCC 2006 and 2019 respectively with no altitudinal disaggregation. This shows a slight decrease (6%) when moving from IPCC 2006 to 2019 methodology.

4.3.2 Effect of addition of altitudinal disaggregation

Animal numbers were shared between altitudes and a whole farm footprint produced to compare with the NAD footprint detailed above. The addition of altitudinal variation caused a change in footprint for both IPCC 2006 and 2019 guidelines (Table 4.7).

Table 4.7 NAD vs. disaggregated footprint results (kg CO_2eq kg LW^{-1})

Footprint Inputs	NAD footprint	Disaggregated footprint	% Change
IPCC 2006 (kg CO_2eq kg LW^{-1})	14.5	21.1	4% increase
IPCC 2019 (kg CO_2eq kg LW^{-1})	13.6	20.4	50% increase

The NAD footprint resulted in a lower CF value when using both 2006 and 2019 IPCC methodology. The disaggregated footprint resulted in CF values being 45 and 50% larger respectively, in comparison to the NAD footprint.

4.3.3 Effect of addition of disaggregated emission factors and site-specific farm inputs

Altitudinal disaggregation based on IPCC default guidelines increased the CF in comparison to the NAD footprint. However, when site-specific inputs and disaggregated

EFs are included, the difference is less clear. A comparison between the three footprints is presented in Table 4.8.

Table 4.8 Comparison of NAD, disaggregated and site-specific footprints (kg CO_2eq kg LW^{-1})

	NAD footprint	**Disaggregated footprint**	**Site-specific footprint**
Footprint results (kg CO_2eq kg LW^{-1})	13.6	20.4	13.8

Addition of site-specific information resulted in the CF value decreasing to a value close to that of the NAD footprint. To further assess where footprint changes occurred with addition of site-specific information, inputs were altered individually to quantify which had the biggest impact: EFs, forage digestibility, or forage crude protein content (Table 4.9).

Table 4.9 Footprint variation with added disaggregated EFs and site-specific inputs (kg CO_2eq kg LW^{-1})

Footprint	**Disaggregated footprint (IPCC 2019)**	**Site-specific EFs**	**Site-specific digestibility**	**Site-specific forage CP%**	**Site-specific footprint**
Disaggregated footprint (CO_2eq kg LW^{-1})	20.4	19.4	13.9	20.2	13.8
Change from disaggregated footprint (%)		4.7% decrease	32% decrease	1% decrease	33% decrease

Altering only the EFs to site-specific made a small difference to the footprint. When site-specific digestibility was used, larger effects on the footprint were seen. The measured digestibility in each pasture at CHUM was higher than the median digestibility suggested in IPCC methodologies (see Table 4.2), which likely contributed to this change. When site-specific forage protein content was added, a further small decrease was seen. This could be due to the upland and hill sites having lower protein content in the forage than the UK default used within this modelling, therefore reducing N intake and excretion of the animals. This in turn would affect excretal N_2O emissions. The site-specific footprint was overall 33% smaller than the disaggregated footprint.

4.3.4 Footprint composition

Emissions sources contribution to the overall footprint (enteric CH_4, manure management CH_4, N_2O from pasture deposition, fertiliser application, feed and upstream & on-farm energy use) are shown both by value (kg CO_2eq kg LW^{-1}) and percentage contribution to the total in Figure 4.3.

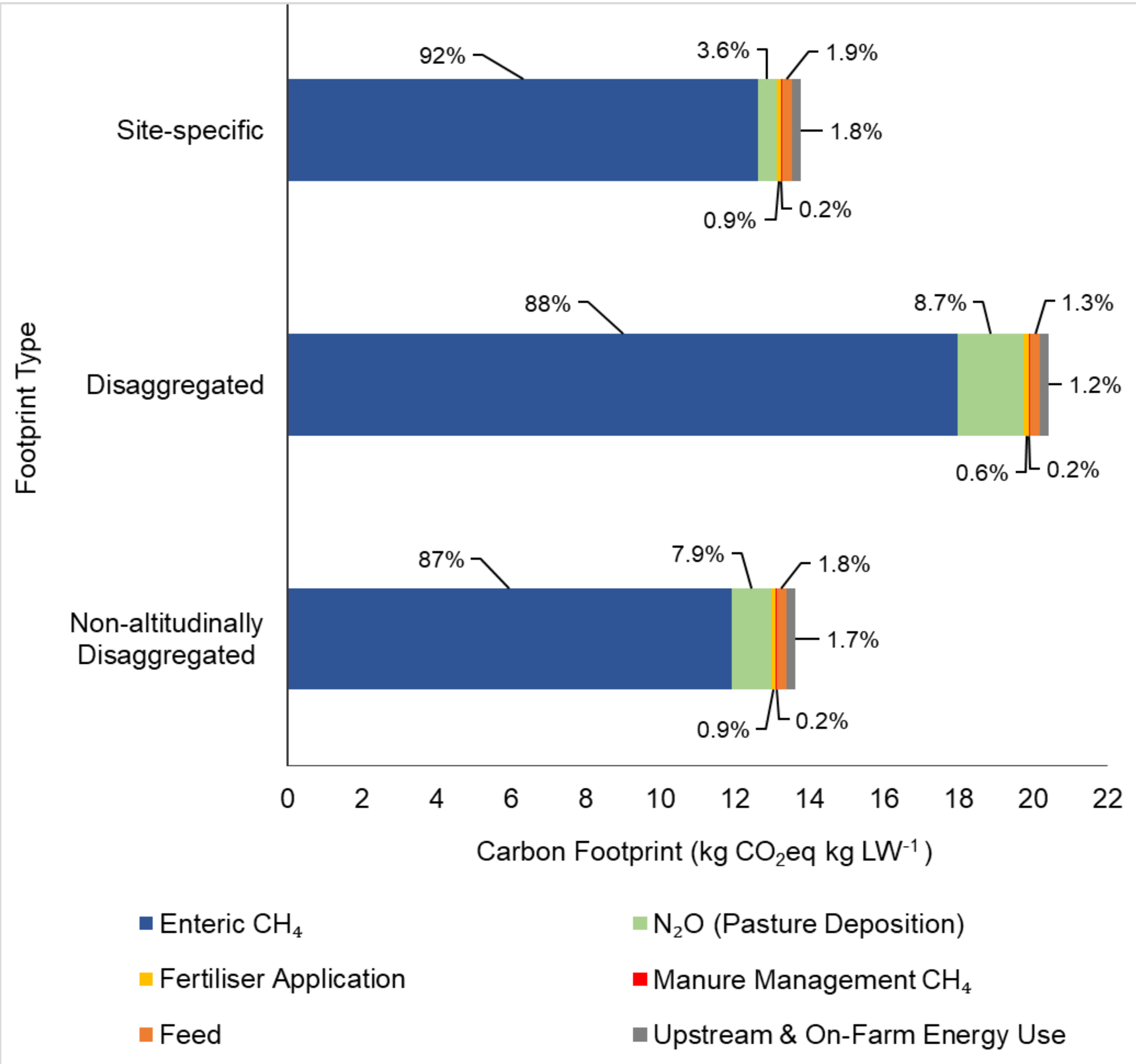

Figure 4.3 Emission sources of footprint results and their percentage contribution to overall CF value

Enteric CH_4 dominated all footprints with N_2O from pasture deposition being the next largest contributor. Enteric CH_4 comprised between 87 and 92% of the total footprint, with N_2O from pasture deposition comprising between 3 and 8.7%. Excretal N_2O being a smaller part of the footprint for the site-specific footprint reflects the smaller EF_{3prp} that has been measured in upland and hill sites. Feed, manure management CH_4, fertiliser application, upstream and on-farm energy use comprised a much smaller portion across all footprints, with the value and percentage contribution being relatively similar.

4.3.5 Whole farm annual emissions profiles

Total calculated greenhouse gas output for the entire farm unit was 448 tCO_2eq and 398 tCO_2eq using 2006 and 2019 IPCC default methodologies respectively (in terms of emission factor, digestibility range and Ym values). Disaggregated footprint resulted in 606 tCO_2eq and the site-specific footprint resulted in 332 tCO_2eq in total annually. Emissions peaks across the year were highest between April and July for all methodologies (likely reflecting the increase in animal numbers during this time due to lambing). Emissions were then lowest across the winter months (Figure 4.4).

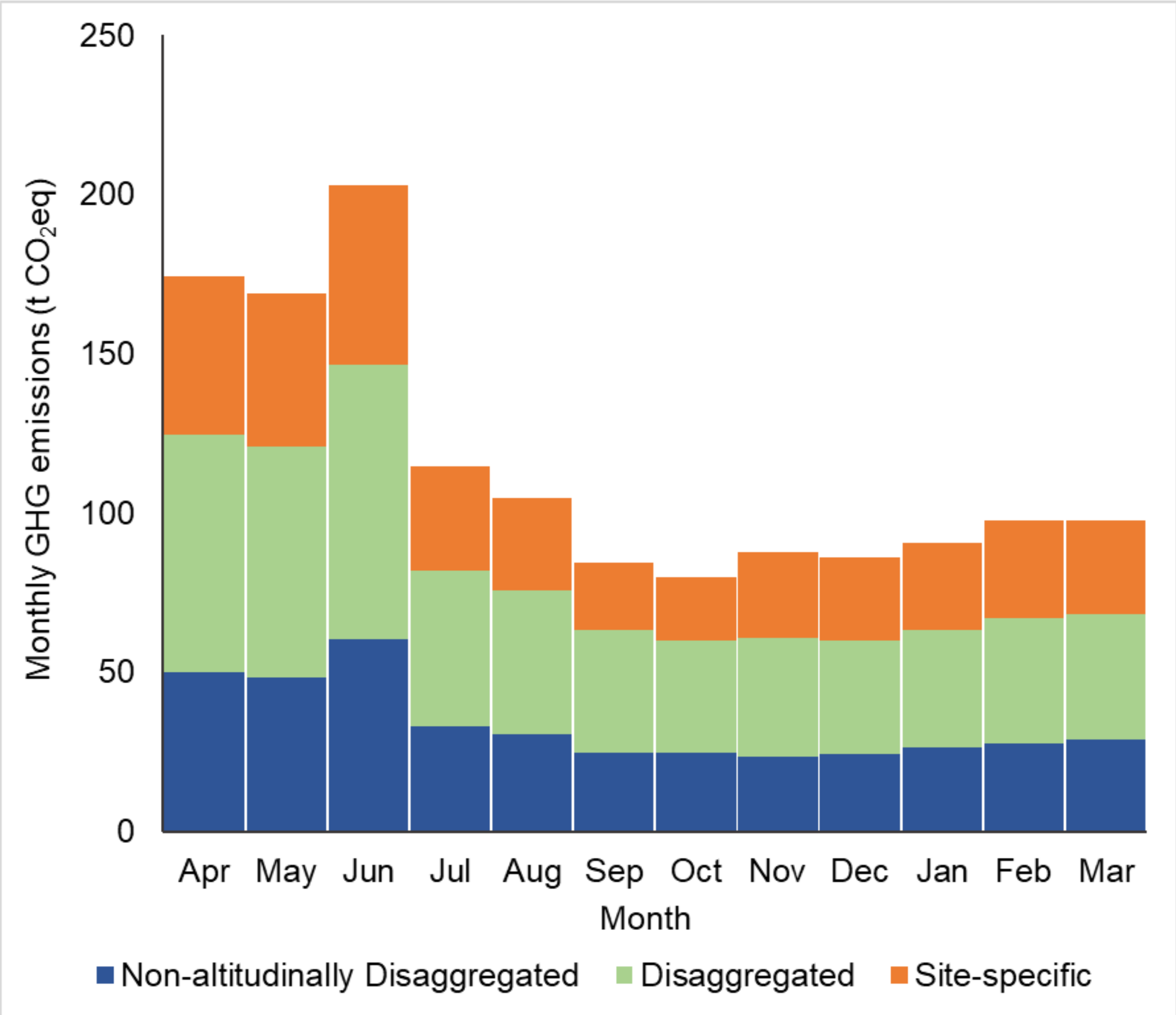

Figure 4.4 Monthly farm emissions (t CO_2eq) calculated in the non-altitudinally disaggregated (NAD) footprint, disaggregated footprint, and site-specific footprint

Across all footprints, monthly GHG emissions were largely similar in general profile throughout the year, increasing and decreasing at similar times. However, the site-specific footprint had the lowest total monthly values across the year. Monthly composition of emissions for the disaggregated footprint are shown in Figure 4.5.

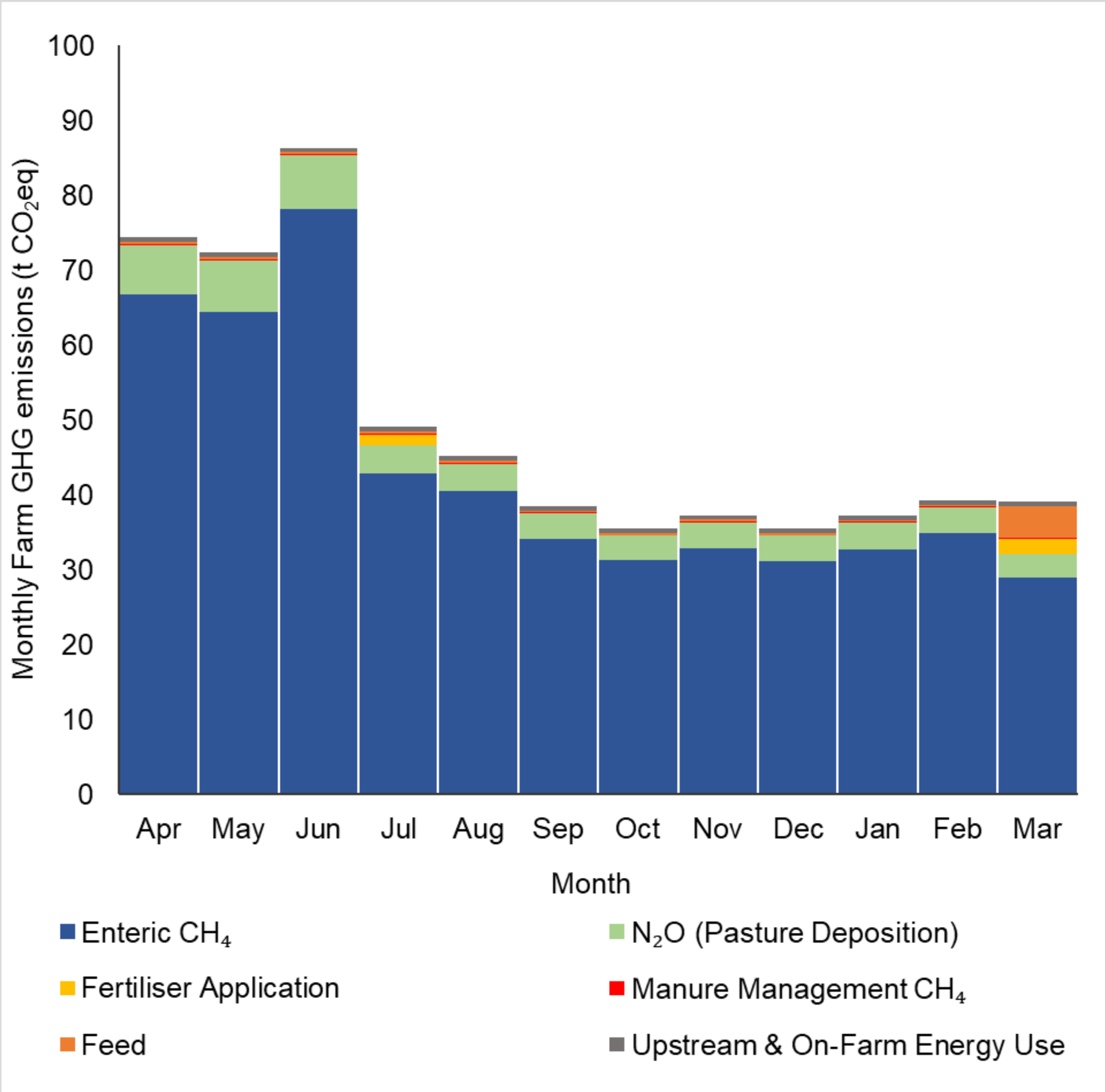

Figure 4.5 Monthly composition of emissions sources to the total (t CO_2eq) based on the disaggregated footprint

Annual emissions composition shows the slight peak in March and July from fertiliser emissions, which goes alongside the date of application. Most feed emissions occur in March, reflecting that most of the total concentrate used onsite is fed in March (as described by information from CHUM). Emissions being highest in June reflects lambs born in April growing larger and beginning to produce CH_4. The drop in emissions in July is attributed to adult ewes being assumed to no longer be lactating, therefore having reduced energy requirements and CH_4 emissions. This is despite the lamb cohorts growing and beginning to produce CH_4. As adult ewes are the largest cohort on the farm, changes to this group have a greater effect on emissions. Animal numbers then decrease between August and December, explaining the dip in emissions between these two months.

The annual enteric CH_4 kg animal$^{\mathbf{-1}}$ year$^{\mathbf{-1}}$ (mean value across all cohorts) was calculated for reference against Tier 1 estimates, with differences in emissions found between the footprints (Table 4.10).

Table 4.10 Annual enteric CH_4 (kg animal^{-1} year^{-1}) across all cohorts for the different footprints

Footprint	**NAD footprint**	**Disaggregated footprint**	**Site-specific footprint**
Annual enteric CH_4 (kg animal^{-1} year^{-1})	8.5	8.5	6.1

The site-specific footprint had the lowest annual enteric CH_4 with IPCC 2019 footprint producing the highest number. This is likely due to the improved forage digestibility in comparison to the IPCC median of the ranges, as well as the lower Ym value used for the hill pasture.

4.4 Discussion

4.4.1 Footprint values

The footprints assessed within this study ranged from 13.6 – 21.1 kg CO_2eq kg LW^{-1}, depending on whether it was the NAD, disaggregated or site-specific footprint being assessed and whether the footprint followed IPCC 2006 or 2019 inventory guidelines. An assessment of lamb CF in NZ determined a value of 6.01 kg CO_2eq kg LW^{-1}, which is considerably smaller than all footprints determined in this study (Mazzetto et al., 2023). A CF of four sheep breeds in India found a range of 5.7 – 9.5 kg CO_2eq kg LW^{-1}, similar to that of the NZ study and again lower than the footprints in this study (Lal et al., 2022). A mean of 11 kg CO_2eq kg LW^{-1} was generated across twenty farms in the most recent (albeit not peer-reviewed) assessment of Welsh beef and lamb (Williams et al., 2020). This is fairly close in value to the site-specific footprint (13.6 kg CO_2eq kg LW^{-1}), indicating possible similarity with local values. The variety of CF results across these studies highlights the variability in lamb production system footprints globally.

Overall, the CF values reported here fall within the 3.5 - 25 kg CO_2eq kg LW^{-1} value as reported in a recent review of sheep system related LCA, although the values are on the mid to higher end of this range (Bhatt and Abbassi, 2021). A possible contributor to this could be that while known as a popular and important breed for the local area, Welsh Mountain sheep have been found to be comparatively less productive in terms of LW gain and number of lambs produced per ewe than bigger crossbreeds or terminal breeds (Wolf et al., 2014). This may have reduced output and caused an increased footprint in comparison to other types of farm.

Enteric CH_4 accounted for 87-92% of emissions. This value is consistent with other sheep system LCA in that enteric CH_4 has been identified as the main contributor towards the footprint. However, the composition of the footprint that is enteric CH_4 is higher than the range presented for other studies at 50-75% (Bhatt and Abbassi, 2021). This is likely due to CHUM using a comparatively small amount of concentrate feed and fertiliser throughout the year, therefore other farms may have different compositions of emissions.

4.4.2 Reflections for effective use of Intergovernmental Panel on Climate Change guidelines

A variety of factors were reassessed between IPCC 2006 and 2019 guidelines for sheep systems. EF_{3prp} for pasture excreta N_2O is a key revision (1% in 2006 methodologies revised to 0.3% in IPCC 2019 methodologies for sheep urine). Additionally, the max value of the range of digestibility for pasture systems is higher in the 2019 refinement (80% vs. 75%). This resulted in a higher median digestibility being used in analysis. Additionally, EF_5 (emissions associated with leaching and run-off) is lower in the IPCC 2019 refinement as well as the manure conversion factor (MCF) (0.0047 in 2019 vs. 0.01 in 2006). A reduction in footprint is seen despite an increased Ym value in IPCC 2019 (6.5% for adult sheep in 2006 vs. 6.7% in 2019) (IPCC, 2006) (IPCC, 2019). These differences are likely to have simultaneously caused the overall reduction in the footprint between the 2006 and 2019 guidelines.

Assessment of the IPCC guidelines is relevant as the most recent peer-reviewed lamb production footprints in Wales (and the UK to an extent) have followed these guidelines, in particular Tier 1 EFs (Hyland et al., 2016; Jones et al., 2014b). These studies followed IPCC 2006 guidelines of 8 kg CH_4 animal^{-1} year^{-1} (IPCC, 2006). IPCC 2019 guidelines have an updated Tier 1 estimate of 9 kg CH_4 animal^{-1} year^{-1} for high productivity systems and 5 kg CH_4 animal^{-1} year^{-1} for low productivity systems for all animals including those that are not yet mature. The guidelines recommend that countries in Europe should use the high productivity estimate (IPCC, 2019). However, the UK inventory has developed country-specific Tier 1 estimates using UK data and now uses revised EFs of approximately 5kg CH_4 animal^{-1} year^{-1} (Brown et al., 2022).

The annual enteric CH_4 per animal calculated in this study ranged from 6.1 – 8.5 kg CH_4 animal^{-1} year^{-1}. In general, site-specific inputs resulted in the lowest annual CH_4 animal^{-1} year^{-1} of 6.1 kg as a farm average, highlighting the potential for variations within farms depending on the animal cohort and model inputs. Overall, the range determined here is lower than the recommended IPCC Tier 1 estimate of 9 kg CH_4 animal^{-1} year^{-1}. Using this estimate therefore has the potential to overestimate enteric CH_4 emissions, depending on the system and cohort being analysed. In contrast, Italian studies of sheep systems have found values of 12.0 kg CH_4 animal^{-1} year^{-1} and 13.6 kg CH_4 animal^{-1} year^{-1} indicating that higher values are also possible for different systems and model methodologies (Atzori et al., 2014; Vagnoni and Franca, 2018). These comparisons further cement the inherent uncertainty still present when estimating on-farm emissions. However, it is recognised that improving emission estimates is a difficult task and has already been the subject of many years of research. Nevertheless, caution should be exercised when making decisions at a farm and country level using default emissions estimates in place of site-specific information.

4.4.3 Differences in farm footprints when using site-specific information

As different input factors became more site-specific in this modelling study, overall farm emissions and CF decreased. This highlights that, while the model inputs used for the site-specific footprint here were based on values generated across a limited time frame, there are periods of time (at least) where CF of different systems are not being adequately assessed. This conclusion has been drawn before – the high variability of data across regions underpinning dairy CF was determined to strengthen the significance of using information specific to the farm instead of averaged data (Uddin et al., 2022). It has also been concluded out with livestock systems, where differences in terms of soil and farm type were seen to affect the CF of two organic farms producing a variety of crops. This lead to the recommendation of farm footprints being “farm-specific” and the impact of input factors needing to be analysed at a farm level to fully understand emission hotspots (Adewale et al., 2019).

When altering the EFs and forage characteristics from default values to those that were site-specific (of the case study used here), site-specific forage digestibility had the greatest effect on changing the CF result. Within this model, the median of the IPCC range for forage digestibility given for grazing pasture and low-quality forages/in rangelands (taken within this study as being applicable to upland/hill sites) is lower in all cases than values measured on-site for lowland, upland, and hill pastures. This indicates that pasture quality is higher at the case study farm than the default values suggested. Selecting digestibility values for extensive pastures is complicated by the lack of evidence available for upland and hill forages. The digestibility of acid grassland species (*Agrostis festuca* specifically) has been found to vary between 40 and 76% throughout the year (HFRO, 1979), whilst an average digestibility value of 64% has been measured for generic extensive pasture (MAFF, 1975). These values are presented in comparison to the 68% and 63% measured within the upland and hill pastures respectively in this case study. The assessments above are outdated and do not reflect the significant changes in UK agricultural practices since the 1970s, but there are no updated reports available at the time of writing. It is also not common practice to assess forage quality across pasture varieties when applying CF to extensive systems, therefore this lack of recent data poses challenges for accurate estimation of emissions from such systems.

The EFs used in the site-specific modelling also vary between altitudes. The lowland Ym value used was higher than the IPCC default Ym, with the upland value being similar and the hill value lower. The hill Ym value used is associated with high uncertainty because it is based on only one experiment, and there are no similar studies to compare sheep enteric CH_4 EFs with. Similarly, the measured EF_{3prp} for the lowland pasture was similar to the IPCC default, whereas upland and hill pastured had lower EF_{3prp} values. The site-specific EFs while having a small effect on changing the footprint (4.7% decrease), still did not have as much effect as site-specific digestibility inputs (32%). This finding is supported by other modelling exercises, where results were found to be less sensitive to EFs and more sensitive to farm inputs and activity data (Chen and Corson, 2014).

Improving agricultural sustainability requires improving GHG inventory methods and a key aspect of this is development of appropriate EFs. Development of EFs for specific sites is unlikely to be possible but efforts to obtain better site-specific information would be a step forward. These results highlight that there is a potential for the use of system averages and default IPCC methodologies to result in both an overestimation and underestimation of footprints. While there is limited evidence behind site-specific factors that could be implemented into this model, every effort was taken to choose values that were as representative of the systems as possible. In the context of lamb production, there may be an underestimation of lowland footprints due to a lower default Ym used in farm footprint models than found from experimental measurements in this study. However, upland and hill footprints may be unfairly assessed due to a lack of representative data for pasture quality in those areas. This has wider implications for upland and hill farms in their contribution to wider ecosystem services (which are not adequately assessed in the literature either) (Ripoll-Bosch et al., 2013).

It is not practical to expect full assessment of site-specific information given the resources required, nor may it be practically possible to capture every variability and uncertainty present within a farm. However, it may be possible and more realistic to better understand select model inputs that models are known to have high sensitivity to. An assessment of two dairy farms (one in Sweden and one in NZ) showed that the within farm variation when the CH_4 EF was varied created bigger variation in the footprint than was present between the farms in the two countries. DMI, enteric CH_4 EF, nitrogen application and direct N_2O

from soil EFs were identified as the priority inputs for improved estimation (Flysjö et al., 2011). At the very least, inclusion of an uncertainty analysis of these inputs, while potentially making differences between systems less clear, can aid in providing a more robust and representative knowledgebase for effective decision making (Chen and Corson, 2014).

4.4.4 Recommendations to improve carbon footprinting of lamb

Given the comparative lack of study within CF for lamb production systems, in relation to beef and dairy systems, there exist improvements that could be made to better quantify impacts of the systems (Jones et al., 2014a). Given the known effect of forage characteristics on footprint results, and the strong influence that forage digestibility had in this study, it is recommended in the first instance to improve understanding of forage characteristics across system types and over seasons to improve accuracy of CF. If possible, when footprinting a specific farm, an understanding of the changes in forage quality across the farm during the grazing season would be beneficial. This is particularly relevant for upland and hill sites, where less forage analysis has been undertaken and uncertainty remains high. In addition to forage characteristics, improvement of understanding of farm characteristics (collection of farm data), DMI of animals and how Ym and EF_{3prp} vary across the different pastures will aid in improving footprint accuracy, though greater clarity on these inputs would require field experiments over a period of years to generate.

For lamb production systems in particular, assessment of the wider implications of the systems would be beneficial so that system benefits could be captured, e.g., by adopting an LCA approach that accounts for the different pastures (altitudes) grazed. While extensive sheep systems appear to have higher global warming impact than intensive systems, they had less impact on other impact categories including eutrophication, acidification and land use change (Geß et al., 2022). Extensive grazing is linked to a variety of ecosystem services including C sequestration provision, environmental grazing to promote sustainability of natural resources, biodiversity improvements and contribution to social and cultural services. Development of a common approach to accounting for soil carbon sequestration within modelling is particularly relevant for extensive grazing systems (Batalla et al., 2015). As demonstrated by Batalla et al (2015), inclusion of C sequestration into CF altered the ranking of GHG impact between the systems being assessed. A reduction in the impact of extensive systems was calculated due to the C sequestration offsetting emissions, making them comparable to intensive systems. Another study found the CF value for a hill sheep system changed from 14.2 kg CO_2eq kg LW^{-1} to 7.0 kg CO_2eq kg LW^{-1} when carbon sequestration of the grassland was accounted for, highlighting the impact this can have on extensive grazing systems (O'Brien et al., 2016). In general, inclusion of ecosystems services as outputs in CF and LCA can alter conclusions on the comparative environmental efficiency of different systems, therefore considering these can be important to draw broader conclusions on the long-term sustainability of lamb production systems in relation to provision of multiple goods (Bernués et al., 2017; Ripoll-Bosch et al., 2013).

An additional aspect that is being discussed, particularly for grazing systems, is to consider the presence of "baseline" emissions, which are emissions related to wild herbivore populations that may exist in that area (and are not currently calculated in IPCC methodologies) (Pardo et al., 2022). As livestock are fulfilling the role of the wild herbivores within the ecosystem, only emissions above the "baseline" could be considered anthropogenic. In following this approach, Pardo et al, 2022 found a 30% decrease in lamb footprints. Wider analysis of African savannah found emissions related to wild animals to be estimated as similar to those under managed grazing (76.2 in comparison to 76.5 Mg CO_2eq km^{-2}) (Manzano et al., 2023). As such, the authors concluded that strategies that may encourage grazing abandonment may not result in substantial decrease in animal-related direct GHGs. Considering the baseline emissions therefore can have implications for consideration of grazing systems and their sustainability.

4.5 Conclusions

Environmental assessments, in particular CF, have become a common tool for assessing the impacts of livestock systems on the environment. Livestock systems are inherently difficult to accurately assess due to the high levels of variation in animal and farm characteristics, for example animal breed, LW, pasture type and use of farm inputs. This is particularly true for lamb production systems, as they involve frequent movement of animals across different types of pasture. Until now, no assessment of site-specific footprinting has been completed for this sector. Initially, when following IPCC guidelines and applying default factors, disaggregation of GHG emission calculations across the three main altitudes caused CF to increase. However, when site-specific inputs and altitudinally disaggregated EFs were applied, footprints decreased. The footprint was particularly sensitive to forage digestibility, which is a known but sometimes overlooked primary driver of CH_4 emissions. While there is uncertainty related to the site-specific inputs and EFs due to a lack of data, it is still clear that site-specific inputs have the potential to alter footprints in comparison to default methodologies, thereby altering conclusions about effective management of lamb systems to meet climate targets. Overall, improved characterisation of lamb farm systems, and better understanding of temporal and spatial (altitudinal) variability in forage characteristics and EFs, will allow more accurate CF of lamb production. Additionally, assessment of wider ecosystem services that may arise from the systems in a UK context will aid in policy decisions for sustainable management of extensive pastures going forward. Enteric CH_4 remains a key contributor to all footprints within this study, therefore, represents a key focus for reducing lamb CF and wider GHG mitigation within the sector.

4.6 Reference list

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Chapter 5 Lamb production carbon footprinting and site-specific modelling: a sensitivity analysis

5.1 Introduction

Sensitivity analysis allows the effect of uncertainty in model parameters on final results to be quantified. The International Organisation for Standardisation (ISO) states that its use is a vital part of life cycle assessment (LCA) studies (ISO 14044, 2006). Use of sensitivity analysis has been highlighted as an improvement to LCA for several reasons (Lacirignola et al., 2017). It allows the impact of uncertainty of different model parameters to be quantified, aids in model development, increases reliability of results and can aid in making decisions related to the model. It can also aid in simplifying model processes, as the inputs that have greatest effect can be focussed on while default values can be used for less influential variables (Wei et al., 2015).

A lack of data availability and field experiments within lamb production systems makes this an issue of even greater concern. Data input variability could have a significant effect on model results; however, this is not often captured by default methodologies. It has been shown that greenhouse gases (GHGs) from ruminant systems, including enteric methane (CH_4) and excretal nitrous oxide (N_2O), may vary across different pasture types. However, there is a lack of evidence across these pastures both temporally and spatially. Additionally other model parameters including animal numbers, animal weights and forage characteristics, which may vary across altitudes (see Chapter 3).

The global warming potential (GWP) allows the warming effect of N_2O and CH_4 to be assessed with reference to CO_2, over a specified time frame. The choice of GWP value can have an effect on the final footprint. GWP_{100} (which assesses gases over a 100 year time frame) is the most commonly adopted method, despite it being critiqued for its limitations in how it assigns a value to the warming effect and captures the effect of climate feedbacks (Levasseur et al., 2016). This is particularly relevant for CH_4 due to it being a short-lived GHG, persisting in the atmosphere for approximately 12 years (Balcombe et al., 2018). The UK inventory currently uses GWP_{100} values from the IPCC Fourth Assessment Report (Brown et al., 2022). However, these have since been updated in the Fifth and Sixth Assessment Report. It has been recommended that carbon footprint (CF) studies assess the effect of updates to the IPCC methodologies on the final footprint (Reisinger et al., 2017).

As shown in Chapter 4, altering model inputs from default numbers to inputs specific to the farm had an effect on changing the footprint. This includes inputs such as farm activity data (for example animal numbers and weight) or model parameters (for example emission factors (EFs) and forage digestibility data). However, the sensitivity of the model to these inputs was not fully assessed. Therefore, the aim of this chapter is to fully assess the sensitivity of the altitudinal model to different farm inputs. This will support recommendations for refinement of priority model parameters to improve the accuracy of CF for altitudinal lamb production systems. A second aim was to assess the sensitivity of the model to the site-specific EFs generated in Chapter 3 in isolation and when combined with wider literature data to inform subsequent work on farm mitigation strategies. Finally, the effect of subsequent updates to the IPCC GWP_{100} (AR4, AR5 and AR6) values on the lamb production footprint were quantified.

5.2 Methodology

5.2.1 Sensitivity analysis of selected model parameters

5.2.1.1 Overview of sensitivity analysis methodology

The initial sensitivity analysis was conducted via a sequential methodology, which involved altering selected model inputs one at a time by a pre-specified percentage. This was to understand the relative effect that different inputs and parameters had on the model outputs. The alteration chosen was an increase of 25% of each parameter. The change in the final CF and product output in comparison to the baseline footprint was then quantified as a percentage.

In Chapter 4, a model was developed that could account for disaggregation of selected model parameters and farm activity data related to a lamb production system. Analysis was conducted using this disaggregated model, with IPCC default model parameters (including EFs) and farm activity data from the Centre of Hill and Upland Management (CHUM). IPCC default model parameters were used because the purpose of this analysis was to assess the effect on the footprint of changing specific parameters and not to assess the parameters themselves. The site-specific footprint and EFs generated in Chapter 3 and 4 of this thesis are explored further in Section 5.2.2. Parameters were changed at each altitude simultaneously. The baseline footprint values are shown in Table 5.1.

Table 5.1 Baseline footprint results used for comparison

Output	Baseline
Carbon footprint (kg CO_2eq kg LW^{-1})	20.4
Annual farm GHGs (tCO_2eq)	590.4

The same change was selected for all parameters so that the relative effect on footprint results was directly comparable. Comparisons were also completed using AR4 GWP_{100} values, as is consistent with the rest of this thesis (except the section that specifically explores GWP values).

5.2.1.2 Selected model parameters

To estimate sensitivity of the model to different inputs, the inputs were split into two categories: EFs and farm activity data. The EF category included the two main EFs relevant for lamb production systems: the enteric CH_4 Ym and excretal N_2O EF_{3prp}. The farm activity data included different farm characteristics such as forage digestibility and animal weight. A full list of model parameters assessed can be seen in Table 5.2.

Table 5.2 Model parameters assessed within the sensitivity analysis

Farm characteristics	EFs
Forage digestibility (DE%)	CH_4 conversion factor (Ym)
Forage protein content (CP%)	EF_{3prp}
Animal weight (kg)	
Animal numbers	

5.2.1.3 Effect of Global Warming Potential value on carbon footprint results

As described in Chapter 4, this study has utilised the IPCC GWP_{100} values from the Fourth Assessment Report as this is consistent with the UK inventory (Brown et al., 2022). However, both the Fifth and Sixth Assessment Reports have been published and the GWP values updated (Table 5.3).

Table 5.3 GWP values taken from IPCC Assessment Reports (IPCC, 2021)

GHG	AR4	AR5	AR6
CH_4	25	28	27
N_2O	298	265	273

The differences in footprint results using each value were therefore assessed. This comparison is made in the context of countries updating GWP values within future inventory reporting.

5.2.2 Sensitivity analysis of measured emissions data

5.2.2.1 Overview of sensitivity analysis methodology

Further sensitivity analysis was likewise conducted to assess sensitivity of the model to variation within the data collected in Chapter 3. Site-specific EFs were generated for both enteric CH_4 (Ym) and excretal N_2O (combined EF_{3prp} from urine and dung) in lowland, upland, and hill sites. Only lowland and upland sites were quantified for excretal N_2O emissions and the combined (urine and dung) EF_{3prp} is presented as the model is built to utilise a combined EF. Analysis was conducted using the disaggregated model with site-specific model parameters (including EFs and forage characteristics) and farm activity data from the Centre of Hill and Upland Management (CHUM).

The sensitivity of the farm footprint to the calculated maximum and minimum of the uncertainty range (± standard error of the mean (SEM)) of the Ym and EF_{3prp} generated in the field experiments is explored. The values calculated are shown in Table 5.4 and 5.5. Due to there being no experiment conducted to quantify EF_{3prp} in the hill plot, data from previous experimental work in the same site were used (Marsden et al., 2019). However, data from this work quantified the urine N_2O EF only and did not include emissions from dung. Emissions from dung deposited on hill pasture were therefore not assessed in this analysis, due to a lack of available data.

Table 5.4 Experimental Ym values with associated uncertainties (± SEM)

Ym				
	Value	**Uncertainty range**	**Lower value**	**Upper value**
Lowland 2020	12.8	±0.8	12.0	13.6
Lowland 2021	8	±0.9	7.1	8.9
Upland	6.5	±0.2	6.3	6.7
Hill	4.2	±0.2	4	4.4
Aggregated lowland*	11.1	±0.8	10.3	11.9

*Aggregated lowland values refer to the combined mean and uncertainty range of the lowland 2020 and 2021 measurement periods.

Table 5.5 Experimental EF_{3prp} values with associated uncertainties (± SEM)

Combined EF_{3prp}				
	Value	**Uncertainty range**	**Lower value**	**Upper value**
Lowland	0.36	±0.2	0.16	0.56
Upland	0.12	±0.1	-0.01	0.25
Hill*	0.08	Not reported	-	-

**value taken from (Marsden et al., 2019)*

Footprints were generated using the lower (mean – SEM), mean, and upper (mean + SEM) estimate of each parameter (Ym and EF_{3prp}). Ym and EF_{3prp} were assessed separately to determine the relative effect of each parameter on the final footprint result. Footprints are presented in kg CO_2eq kg LW^{-1}.

5.2.2.2 Site-specific data from the wider literature

To further assess potential site-specific EFs, the mean values obtained from field experiments were combined with wider literature to derive a range for both Ym and EF_{3prp}. The sensitivity of the model to these wider literature values was then tested.

The sources for these wider literature values are described in this section. For Ym, lowland values included experiments conducted in the UK and New Zealand (n=8) (Table 5.6). Data from New Zealand were included, as these data are what IPCC guidelines are based upon. Only data using UK origin sheep breeds and ryegrass type swards were included. Very little data existed for upland pastures (n=4) and no other data were found (at the time of writing) for hill pastures specifically, therefore the hill site uses the GreenFeed data only.

Table 5.6 Ym data sources from wider literature

Data source	Ym%	Country of origin	Sheep breed and sward type	Reference
Lowland	12.8	Wales, UK	Welsh Mountain, ryegrass mix	Measured data (Chapter 3)
	8.0	Wales, UK	Welsh Mountain, ryegrass mix	Measured data (Chapter 3)
	5.7	Northern Ireland, UK	Scottish Blackface/Scottish Blackface Cross, ryegrass	(Zhao et al., 2017)
	6.3	New Zealand	Romney, ryegrass dominant	(Swainson et al., 2016)
	6.2	Northern Ireland, UK	Scottish Blackface/Scottish Blackface Cross, ryegrass	(Zhao et al., 2016)
	5.2	Wales, UK	Welsh Mountain/Welsh Mule X Texel, ryegrass	(Fraser et al., 2015)
	7.2	New Zealand	Romney, ryegrass	(Sun et al., 2012)
	7.5	New Zealand	Romney, ryegrass	(Pinares-Patino et al., 2003)
Upland	6.4	Wales, UK	Welsh Mountain, semi-improved upland ryegrass/white clover sward with other herb species	Measured data (see chapter 2)
	5.6	Northern Ireland, UK	Scottish Blackface/Scottish Blackface Cross, mixed species semi-natural upland grassland	(Zhao et al., 2017)
	5.6	Wales, UK	Welsh Mountain/Welsh Mule X Texel, extensive grass, and herb mix pasture	(Fraser et al., 2015)
	4.1	New Zealand	Romney, mixed species extensive pasture	(Ulyatt et al., 2005)
Hill	4.2	Wales, UK	Welsh Mountain, heather heathland >700m a.s.l	Measured data (Chapter 3)

For EF_{3prp}, data also originates mostly in the UK or New Zealand (Table 5.7). Data were only used from UK-origin sheep breeds. Data for sheep EF_{3prp} are limited in general, however particularly limited for hill sites.

Table 5.7 Data sources for disaggregated EF_{3prp}

Data source	EF_{3prp}	Country of origin	Sheep breed and sward type	Reference
Lowland	0.36%	Wales, UK	Welsh Mountain, ryegrass mix	Measured data (Chapter 3)
	0.30%	Global	Aggregated value in IPCC, considered representative of typical lowland pastures in this work	(IPCC, 2019a)
	0.37%	New Zealand	Meta-analysis of experimental data conducted across New Zealand	(van der Weerden et al., 2020)
Upland	0.12%	Wales, UK	Welsh Mountain, semi-improved upland ryegrass/white clover sward with other herb species	Measured data (Chapter 3)
	0.11%	Wales, UK	Welsh Mountain, semi-improved upland grassland (British NVC U4 and M56)	(Marsden et al., 2019; Marsden et al., 2018)
	0.09%	New Zealand	Meta-analysis of experimental data conducted across New Zealand (medium/steep slope)	(van der Weerden et al., 2020)
Hill	0.08%	Wales, UK	Welsh Mountain, heather heathland	(Marsden et al., 2019)
	0.09%	New Zealand	Meta-analysis of experimental data conducted across New Zealand (medium/steep slope)	(van der Weerden et al., 2020)

The values were then averaged by altitudinal pasture and implemented within the model for both Ym and EF_{3prp} (Table 5.8).

Table 5.8 Final combined disaggregated Ym and EF_{3prp} values

Ym			
	Mean	**Lower value**	**Upper value**
Lowland	7.4	5.2	12.8
Upland	5.4	4.1	6.4
Hill*	4.2	-	-
EF_{3prp}			
	Mean	**Lower value**	**Upper value**
Lowland	0.25	-0.01	0.37
Upland	0.07	-0.02	0.12
Hill	0.09	0.08	0.09

**Only experimental data was utilised for hill pastures, due to a lack of published data*

Values used within the model for both Ym and EF_{3prp} in hill pastures were considered less robust, due to a lack of available published data at the time of analysis. Only the collected experimental data were available for Ym in the hill pasture, and limited studies were available (2 in addition to the experimental data) were available for EF_{3prp}. Further data collection in these pastures would aid in improving robustness.

5.3 Results

5.3.1 Sensitivity analysis of selected model parameters

The sensitivity analysis results are presented in Table 5.9.

Table 5.9 Footprint result and annual GHGs when altering selected model parameters as well as the percentage change

Input data	Carbon footprint (kg CO_2eq kg LW^{-1})	Effect on footprint (% change)	Annual GHGs (t CO_2eq)	Effect on total farm GHGs (% change)
Baseline footprint	**20.4**		**590.4**	
Ym	24.9	22	720.3	22
EF_{3prp}	20.6	1	596.9	1.1
Forage digestibility	14.1	31	407.4	31
Forage protein content	21.0	2.8	607.5	2.9
Animal weight	20.8	2	690.8	17
Animal numbers	20.5	0.5	732.1	24

Manipulation of Ym had a greater effect on both the CF and annual farm GHGs than N_2O EF_{3prp} - altering Ym caused a 22% change in both the footprint and total farm GHGs. In comparison, altering EF_{3prp} caused a 1% change in footprint and 1.1% change in total GHGs. This would be expected for lamb production footprints given the smaller proportion of the footprint that is attributed to N_2O emissions from excreta deposition. Enteric CH_4 comprises a larger portion of the footprint as discussed previously.

Manipulation of forage digestibility had a greater effect on both the CF and annual GHGs, in comparison to the other farm characteristics. Altering forage digestibility resulted in a 31% change in both the footprint and total GHGs. Forage protein content alteration caused a 2.8 and 2.9% change for the CF and total GHGs respectively. Altering animal weight had only a 2% change in the footprint but resulted in a 17% change in total farm GHGs. Animal numbers resulted in a 0.5% change in footprint yet a 24% change in total farm GHGs.

Overall, forage digestibility had the greatest effect on both the footprint and annual farm GHGs. Conversely, EF_{3prp} had the least effect on both metrics. Animal numbers had little effect on the footprint but the second most profound effect on annual farm GHGs. This is likely due to more animals on farm resulting in production of more GHGs, yet more liveweight (LW) leaving the farm. The production output increase offsets the increase in emissions slightly, resulting in a lesser effect being seen on the footprint. The same effect can be seen with animal weight alterations.

5.3.2 Effect of Global Warming Potential values on footprint

The resulting CF (kg CO_2eq kg LW^{-1}) and total annual GHGs produced on farm when using different GWP methodology can be seen in Table 5.10.

Table 5.10 Effect of GWP values on footprint

GHG	AR4	AR5	AR6
Carbon footprint (kg CO_2eq kg LW^{-1})	20.5	22.5	21.8
Total GHGs (tCO_2eq)	592.1	648.7	629.3

Changing from AR4 to AR5 values results in a 9.8% increase in farm footprint and 9.6% increase in overall farm annual GHGs. AR6 values then result in a 3.1% decrease from AR5, yet 6.3% increase from AR4. Total GHGs also drop approximately 3% moving from AR5 to AR6 with a 6.3% increase in comparison to AR4. These changes indicate that farm footprint and GHG accounting are affected by the choice of global warming metric.

5.3.3 Analysis of the effect of site-specific data uncertainty on footprint

The footprint values when the ranges of Ym are applied varied (Figure 5.1). Three footprint ranges are presented: one using the lowland 2020 Ym range (± SEM) combined with the upland and hill values, one using the lowland 2021 Ym range (± SEM) combined with the upland and hill values, and one aggregating the lowland data to have a combined mean and SEM alongside the other pasture data. Only one experiment was completed in the upland and hill site, therefore only one range is presented. It was deemed necessary to separate the lowland 2020 and 2021 ranges to fully demonstrate variability in footprint results using data across the experiments.

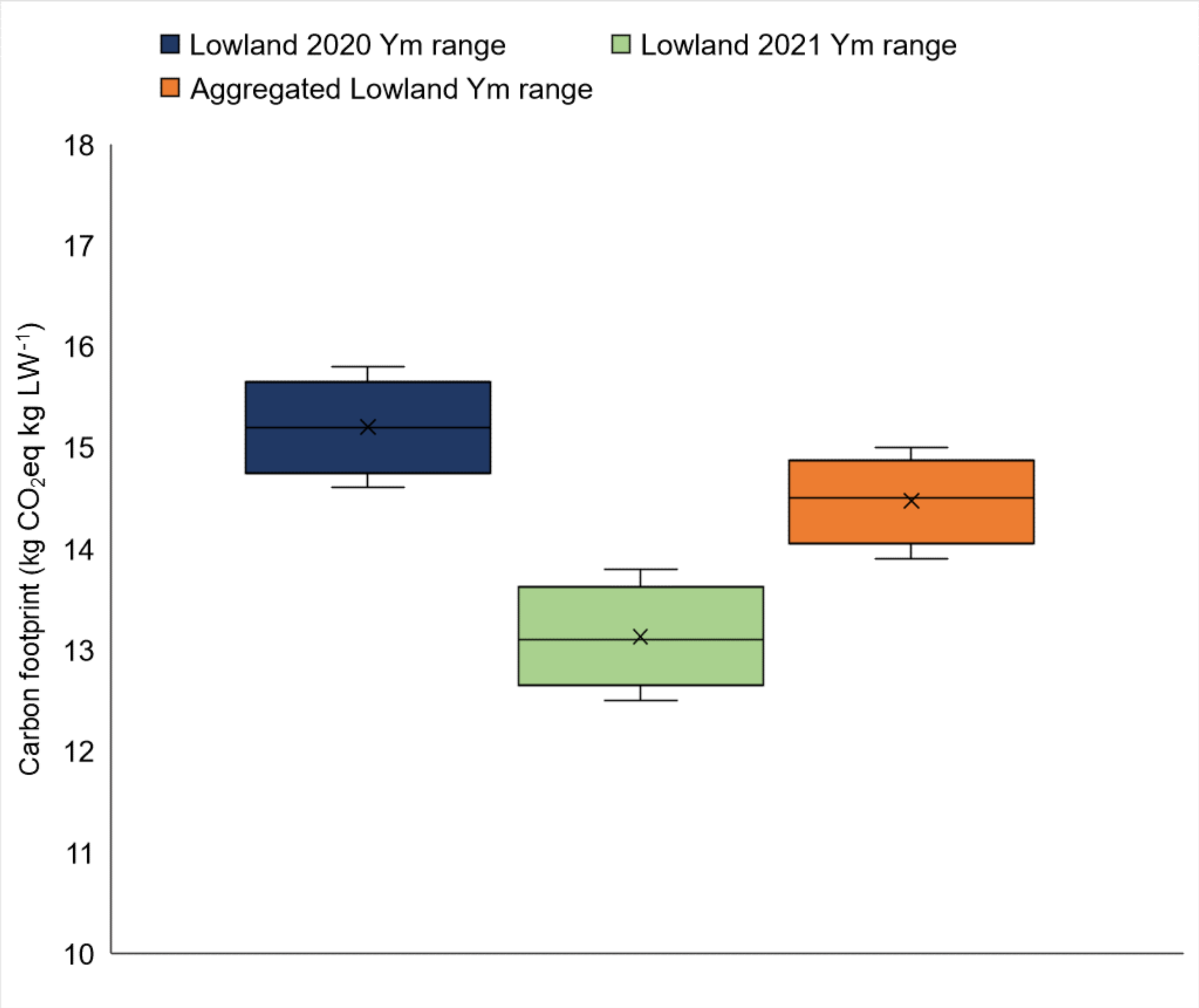

Figure 5.1 Footprint results when using experimental Ym values (± SEM). All footprints utilise the same Ym range for upland and hill pastures, as only one measurement period was conducted in those pastures.

Footprints varied between 12.5 – 15.8 kg CO_2eq kg LW^{-1} when applying the different ranges of experimental Ym values: a 26% change. In comparison, the effect of the experimental range of EF_{3prp} was smaller (Table 5.11).

Table 5.11 Footprint values when using experimental EF_{3prp} values (± SEM)

Range value	Footprint value (kg CO_2eq kg LW^{-1})
Lower	13.6
Mean	13.8
Upper	14

Footprints varied between 13.6 - 14 kg CO_2eq kg LW^{-1} when applying the different ranges of EF_{3prp}, showing a smaller effect on footprint range than altering the Ym value (~3%).

5.3.4 Effect of using wider literature data on footprint

Footprint results varied further when experimental data was used in combination with wider literature (Table 5.12).

Table 5.12 Footprint results using EFs derived from experimental work and wider literature

	Ym	EF_{3prp}
	Footprint value (kg CO_2eq kg LW^{-1})	**Footprint value (kg CO_2eq kg LW^{-1})**
Lower	10.2	13.5
Mean	11.9	13.6
Upper	14.3	13.8

When varying Ym, footprint values ranged from 10.2 – 14.3 kg CO_2eq kg LW^{-1}, showing a marked response to changes in Ym (4.1 kg CO_2eq kg LW^{-1}). A lesser response was seen (alongside smaller values of experimental uncertainty) when altering EF_{3prp} where footprints ranged from 13.5 – 13.8 kg CO_2eq kg LW^{-1} showing a difference of 0.3 kg CO_2eq kg LW^{-1}.

Overall, between experimental data uncertainty and combining with literature data, footprints ranged from 10.2 – 15.8 kg CO_2eq kg LW^{-1} when altering Ym and 13.5 – 14 kg CO_2eq kg LW^{-1} when altering EF_{3prp}.

5.4 Discussion

5.4.1 Effect of model parameters on footprint

Full understanding of the sensitivity of different models to inputs has many advantages in terms of decision making at both farm level and in a policy context, as well as improving estimates of GHG output. High quality input data are essential in improving LCA estimates, and sensitivity analysis provides an insight into the relationship between the input data and CF estimates (Van Middelaar et al., 2013).

The method used here of changing input parameters “one at a time” is a straightforward and less resource intensive route to determining the relative influence of different model parameters to the outputs. However, it does have limitations in that it does not fully explore compounding effects or interactions between model parameters. Additionally, it may not fully capture the entire range of uncertainty within parameters (Groen et al., 2016). However, in this case the method has provided a starting point of model parameters that may be useful to focus on to allow wider data collection and quantification of parameter uncertainty.

Forage digestibility appears to be the input that most strongly influences both the CF result and the annual farm GHGs emissions. This conclusion is congruent to that of Chapter 4, where inclusion of site-specific forage digestibility had an important effect on final CF results. Forage digestibility is known to have a significant effect on CH_4 output from enteric fermentation, which has been measured in many in-vitro studies of direct emission output from the animal (Hegarty et al., 2010; Sun et al., 2015; Dong et al., 2019). This input is consequently a key driver within the IPCC equations for enteric CH_4 calculation (IPCC, 2019, 2006). It is therefore logical that the model is sensitive to this input, given its importance in predicting enteric CH_4 emissions.

In general, accurate quantification of what the animals eat is an important aspect of modelling. An assessment of different grass-based beef systems in Brazil identified feed quantity and quality as being an important parameter for increasing accuracy of CF estimates (Ruviaro et al., 2015). A recent Monte Carlo simulation aimed to assess variation in feed digestibility and crude protein content and its effect on footprints. It found that footprints that did take the two feed characteristics into account differed significantly from footprints that use default methodologies, leading to a recommendation of including the feed characteristics in CF where possible (March et al., 2021). Another study concluded that determining what forage the animals were consuming had higher significance for enteric CH_4 output than what breed they were (Fraser et al., 2015). Further work in New Zealand found inclusion of feed characteristics considerably enhanced the accuracy of model estimates of enteric CH_4 emission in sheep (Muetzel and Clark, 2015).

Ym in isolation similarly had a pronounced effect on the CF value, and a still marked effect on annual GHGs. This is seen in other studies where sheep CF estimates were determined as particularly sensitive to variations in the enteric CH_4 EF (Tsakiridis et al., 2020). Other livestock footprints, including dairy and beef, were likewise sensitive to CH_4 EF variation but to a reduced extent in comparison. This has similarly been concluded at an inventory level, as similar work on the UK inventory concluded that the factor that most affected national enteric CH_4 estimates was the uncertainty behind enteric CH_4 emission factors (Milne et al., 2014). A sensitivity analysis of the Canadian national livestock model found Ym to be an area of key uncertainty within the model, with an uncertainty value of ±50% (Karimi-Zindashty et al., 2012). Disaggregation of IPCC default model inputs to the Canadian province and animal subcategory level resulted in a significant drop in uncertainty for enteric fermentation estimates (reduced to 20% of the mean for province and 13% of the mean for animal subcategory).

The smaller effect on results of EF_{3prp}, animal weight and animal numbers from a CF perspective does suggest they should not be the inputs of focus for improving estimates of lamb production systems. This of course may not be applicable to other systems where excretal N_2O may form a greater proportion of the footprint. However, it has been shown that using average EFs in comparison to site-specific inputs results in differing accuracy for CF results and also inconsistent conclusions from a policy perspective (Adewale et al., 2018). Development of better temporal and spatial EFs is therefore crucial and EF_{3prp} should not be ignored but improving robustness of data underpinning Ym may be of greater importance for these particular systems. The assessment of site-specific collected data both in this chapter and the previous chapter further supports this observation.

5.4.2 Implications for lamb production footprints

It is important to consider the desired outcome and potential applications of the data when presenting results. This sensitivity analysis has shown that for some model parameters, alteration resulted in a small change in the CF yet a large change in the annual GHGs. This has the potential to link to farm management objectives. For example, an increase in animal numbers had little effect on the CF yet a 24% increase in annual farm GHGs. This indicates that while the system retained efficiency in terms of its CF, the actual GHG burden and climate effect increased. Policy is focussed on reducing national GHGs, therefore this finding represents a trade-off between production and reaching climate change emissions targets. Trade-offs such as these will become increasingly important as the margin towards net zero grows smaller.

Assessment through the uncertainty range of data collection in field trials resulted in quite different footprint results, particularly for Ym. It is therefore important to recognise the inherent uncertainty within current Ym estimations and that uncertainty behind data inputs should be considered when drawing conclusions from LCA data. Attempting to quantify uncertainty or sensitivity within any footprinting work and being transparent about the methodologies used will aid in improving GHG estimations.

5.4.3 Choice of global warming metrics

Use of standard GWP_{100} is a global and recognised method, yet concern surrounds its application for different systems and if it may potentially result in erroneous conclusions (Levasseur et al., 2016). This is particularly applicable for the system described here that is dominated by a biogenic source of CH_4 (a short-lived gas). This study considered only the three IPCC iterations of the GWP_{100} value, all of which caused a change in overall CF and annual GHGs. A New Zealand based assessment of dairy farms found that important deductions related to different systems could change considerably with GHG metric (Reisinger et al., 2017). For example, when comparing AR4 and AR5 values (with regards to CF results between dairy farms with different input levels), a change in ranking occurred. Low-input farms initially were 3% lower than high-input farms in terms of CF value yet became only 1% lower at the change of metric. Another study on beef farms found a 9% change in footprint between methods, similar to values determined here (Rotz et al., 2019). A study on milk from New Zealand found a difference of 5% (Ledgard et al., 2020). Farm management and policy decisions could therefore change depending on the type of metric used to display results and may encourage practices that would not necessarily be most advantageous from a climate perspective.

Other methodologies have been developed, which may have implications for ruminant based systems. GWP* has been developed in an attempt to capture the warming effect of short-lived gases more accurately, such as CH_4, and is more sensitive to changes in these gases over short time periods. Applying GWP_{100} to the Australian livestock sector resulted in the sheep industry having a GHG output of 10.3 Mt CO_2eq and a CF of 6.34 kg CO_2eq kg LW^{-1} (Ridoutt, 2021). Changing the GHG metric to GWP* resulted in the sheep industry becoming a net GHG sink of -2.85 Mt CO_2eq. In Austria, changing between the two metrics resulted in a 49% reduction in the global warming impact of milk production (Hörtenhuber et al., 2022). However, it is important to note that this is a new methodology and may not be the most suitable in every application. Further research is required to assess which metrics are best for different situations. Additionally, transparency towards policy makers with regards to the implications of the different values will aid in prevention of flawed conclusions.

5.5 Conclusions

An understanding of the sensitivity of a model to different parameters can clearly aid in indicating focus for model development. This is true for both assumptions underpinning the model (e.g., GWP and EF values) and for farm activity data (e.g., animal weight and numbers). This study found that the model is particularly sensitive to forage digestibility, which remains a parameter with a lack of data availability in particular for extensive pastures. The results also suggest that better understanding of how Ym changes across pasture types and the effect of underlying uncertainty behind experimental data is necessary to improve future estimates. The recommendation that footprints should be as site-specific as possible, based on the available data, is further reinforced by the results seen here. The choice of GHG metric changed the footprint results and therefore should also be considered, and any developments in the numbers recommended implemented into lamb production carbon (C) calculators. The lamb production industry is understudied despite being integral to UK and Welsh agriculture - further study related to quantifying uncertainty behind model parameters and improving the level of data available for use, in particular forage characteristics, will aid in achieving long term sustainability for the sector.

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Chapter 6 Mitigation options and opportunities for land sparing in stratified lamb production systems

6.1 Introduction

Given the known impact of ruminant systems on climate change, it is imperative that scenarios to mitigate that impact are considered to allow agriculture to reach net zero (Grossi et al., 2019). Small ruminants, such as sheep and goats, comprise 56% of the global ruminant population and have an important socio-economic impact, particularly in developing countries in terms of landscape management and ecosystem services. There are estimated to be over 1.2 billion sheep worldwide (FAO, 2019). The most recent UK agricultural survey states that as of June 2021, there were 32.8 million sheep across UK agricultural holdings (DEFRA, 2022). Additionally, they provide a variety of products and sustenance for communities worldwide. Therefore, understanding their contribution to climate change and role in mitigation are imperative for considering future food and farming systems (Batalla et al., 2015).

There are several important aspects to consider when assessing future reduction of greenhouse gases (GHGs) from livestock systems: the accuracy of the underlying modelling, the cost and efficacy of the mitigation options available and how these factors interact. Additionally, how mitigation options may interact with wider environmental or development goals should be considered (Gerber et al., 2013). In terms of modelling, carbon footprints (CF) and life cycle assessment (LCA) is a common and established method now used for the broader assessment of products both within and out-with agriculture (Guinée et al., 2011). LCA is commonly used to assess the carbon (C) impact of a product throughout its lifecycle, but can also be used to assess wider environmental impacts such as eutrophication and acidification (de Vries and de Boer, 2010). However, for agricultural systems, the method has inherent uncertainty that can be largely attributed to the high variability of farm management and environmental characteristics seen within these systems. This creates uncertainty within model input data as well as model methodologies (Notarnicola et al., 2017). This uncertainty then represents issues for selection of mitigation options, in terms of their efficacy across different systems. Improving emissions calculations remains key in improving understanding of mitigation options and in facilitating adoption of robust and evidence-based policy decisions. Additionally, the ability for farm businesses to adopt long term sustainability strategies will be improved (Cederberg et al., 2013).

Mitigation options can work in a variety of ways - by improving on-farm efficiency, targeting reduction of biological emissions e.g., enteric methane (CH_4) and excretal nitrous oxide (N_2O), increasing soil C sequestration, and reducing fossil-derived greenhouse gas outputs (Hyland et al., 2016; Wang et al., 2015). When recommending potential mitigation options themselves, consideration of a variety of factors is necessary. Options must give reasonable reductions to greenhouse gas outputs but also be technically feasible and practicable for farmers to implement with no significant unintended consequences. The cost of implementing a mitigation option should be proportionate to the emissions reduction potential of the strategy (Moran et al., 2011). Additionally, there is potential for interactions between different mitigation options to result in unintended emissions increases in other parts of the farm (Vellinga et al., 2011). For example, a decrease in crude protein (CP) present in an animal's diet has been shown to reduce N_2O emissions by 30 ± 38% but may result in an increase in enteric CH_4 by 71 ± 131%, dependent on the diet chosen (Sajeev et al., 2018).

For lamb production systems, the key GHG to be mitigated is enteric CH_4. This is produced by the rumen during the animals' digestive processes and can account for over 55% of lamb system footprints with some studies reporting contributions of 64% and above (Eldesouky et al., 2018; Marino et al., 2016). Other contributors to the footprint include direct and indirect N_2O emitted due to excreta deposition to pasture and nitrate leaching, emissions related to forage, fertiliser and feed production and related fertiliser application and energy use – although the contribution of these are dependent on their use at a farm level. It therefore follows that any implemented strategies that mitigate enteric CH_4 and minimise use of farm inputs would have a greater effect on lamb production systems.

It has been recommended that the multifaceted relationships between system variables be effectively accounted for to fully assess mitigation options and prevent negative trade-offs (Grossi et al., 2019). To reach dependable conclusions from CF in terms of mitigation potentials, fully understanding the above uncertainty and the relative impact of inputs on resulting CF is essential. Full assessment of emission drivers across farms aids in identifying emission hotspots and further supports decision making. Studies have shown that despite farms producing similar products, inherent differences in size, farm management and location caused notable differences in the resulting product footprint (Adewale et al., 2019). Within CF of lamb production assessment of different farms (varying by region and level of intensity) found a variety of results, with CF values ranging from 3.5 – 25 kg CO_2eq kg LW^{-1}(Bhatt and Abbassi, 2021). An assessment of management options to improve sustainability of different sheep farms found that different impact categories (e.g., GHG emissions, eutrophication, and land use) varied across different farm types including intensive and extensive farms. LCA results and suggestions to improve sustainability of the systems were therefore strongly linked to the system being discussed (Geß et al., 2022). For example, the GHG emissions impact of the product was increased for more extensive systems, whereas there were less local environmental impacts (eutrophication, acidification, and land use). Similar conclusions were drawn from an assessment of mitigation of the Scottish lamb production industry, where the use of Scottish-specific information was recommended in order to capture the variety of breeds, farming system and grazing (particularly rough grazing) that occurs and improve accuracy of future suggested farm strategy conclusions (Moxey and Thomson, 2021).These findings further support the view that site-specific model inputs and emission factors (EFs) are important for model development. It may therefore be helpful to consider improvement of CF in terms of data inputs alongside selection of mitigation scenarios as a holistic approach to improving total system assessment.

Many lamb production systems in the UK follow a stratified production system (that exploits lowland, upland and hill grazing to optimise production) (Morris, 2017), although not all systems follow stratified production and many sheep graze upland and hill areas only (Wolf et al., 2014). The frequent moving of animals in lamb production systems results in natural variations in spatial and temporal factors involved as animals are moved between different pastures. These factors include weather, topography and other related factors including soil type and plant species. Given the variation between farms, it can be assumed that variation in footprints will be apparent and therefore mitigation option efficacy may also differ (Jones et al., 2014a). There is a lack of consideration for this approach when studying lamb production systems and assessing the efficacy of mitigation options across the altitudinal gradient. This stratified structure of production could result in mitigation options commonly presented for cattle and lowland sheep farms not being as effective as stated within current studies or even appropriate for the extensive grazing areas. Additionally, mitigation strategies may be ranked differently in terms of effectiveness depending on the site-specific factors involved.

Sustainable intensification is a strategy that has been explored that can potentially decrease the GHG intensity of a system by increasing productivity. It can reduce demand for additional land by increasing production on land already being used (García et al., 2020). Addition of inputs within regulatory limits to improve grass growth and therefore animal production were seen to improve Irish sheep farming environmental impacts and decrease the product footprint, although increasing concentrates had lower resource efficiency and was less effective than improving grazing (O'Brien et al., 2016). Sustainable intensification has also been (partially) demonstrated in beef systems with an increase in stocking rate resulting in a decrease in CF value (Foley et al., 2011). However, impacts on species decline and biodiversity have been linked to intensification of agriculture (Storkey et al., 2012), therefore consideration of wider impacts is necessary when implementing such a strategy.

The introduction of forestry is often discussed within the land conflict issues surrounding lamb production and it has been determined that afforestation could be an effective option for the sector in terms of improving economic viability and mitigating climate change (O'Neill et al., 2020). In the context of stratified lamb production, questions persist regarding how these options might be applied within the sector. For example, should animals be moved to lowland pastures and fed concentrates to improve finishing times or is it more effective to increase the animals fed from poorer quality pasture and utilise better quality land for other purposes? Efforts to improve productivity using a smaller area of land may also allow land to become available for afforestation. Mitigation options that effectively consider these questions for lamb production specifically have not yet been studied.

This chapter aims to assess these options while considering altitudinal changes in emission outputs. Mitigation scenarios have been selected that allow assessment of animal movements between altitudes on farm and whether GHG savings can be made. Furthermore, the effect of longer-term genetic improvements on animal CH_4 output and feed efficiency on the CF are evaluated. The results are presented using both disaggregated (using IPCC default EFs and forage characteristics) and site-specific footprints as described in Chapter 4 and compared. Finally, where land sparing might occur, an assessment of the potential C sequestration if the area were to be converted to forest has been completed.

6.2 Methodology

6.2.1 Model and data background

The mitigation scenarios in this chapter were completed in the disaggregated lamb production C calculator developed as part of this thesis (see Chapter 4), with the boundary of the analysis being the same case study farm. The CF was generated with input data from Bangor University's Centre for Hill and Upland Management (CHUM), and calculated using Tier 2 IPCC methodologies (IPCC, 2019, 2006).

Scenario analyses were conducted within the lamb production system model that is disaggregated by altitude, as described in Chapter 4 (built in Microsoft Excel). For this analysis, the effect of the different scenarios was assessed using the model with two different categories of input data: IPCC 2019 default inputs (using IPCC default EFs, digestibility ranges but with animal numbers and weight disaggregated by altitude) and the site-specific inputs using experimental data collected within this thesis at CHUM (see Chapter 3). These inputs were selected to show how addition of site-specific information may change the relative effect of the chosen mitigation scenario. Detailed information regarding the farm characteristics and model development can be seen in Chapter 4. Results were then presented in terms of the percentage change in product footprint and annual farm GHGs in comparison to the baseline of each chosen methodology (Table 6.1).

Table 6.1 Model baselines following disaggregated and site-specific footprinting

	IPCC disaggregated footprint	**Site-specific footprint**
CF (kg CO_2eq kg LW^{-1})	20.4	13.8
Annual GHGs (tCO_2)	590.4	397.3

There is uncertainty present in the site-specific data used due to a lack of long-term data collection across the altitudinal gradient at the study site. However, analysis was still undertaken with the limited data collected to show the potential differences in footprint and importance of considering altitudinal variances to inform future land use choices.

Additionally, it further provides insight into the added functionality of the disaggregated model.

6.2.2 Selection of mitigation options

There are a variety of options that have been studied to reduce GHG emissions from ruminant systems (Garnett, 2009). However, limited studies have been performed for sheep systems specifically due to their extensive nature and structure (Jones et al., 2014; Moxey and Thomson, 2021). Mitigation options here were selected for assessment based on their technical feasibility for systems represented by the case study used here, and with consultation of expert opinion.

Previous assessments of UK-wide mitigation options did not consider animal management options for sheep farms due to the structure of the UK industry (Moran et al., 2011). This study attempts to fill this gap by assessing the effect of animal management changes but only at a farm level. It must be highlighted that the final list of selected mitigation options selected are not exhaustive and there may be other options available for use within lamb production systems. However, the updated UK agricultural MACCs (Eory et al., 2015) and expert opinion were consulted when choosing mitigation scenarios for use in analysis.

6.2.3 Description of mitigation scenarios

6.2.3.1 Animal benchmark scenarios

These scenarios focus on the effect of changing specific animal criteria, with a focus on those that may reduce the enteric fermentation emissions of the farm. These scenarios may require a longer time frame to be effective.

Scenario 1 – Breeding for low CH_4 output

Breeding for low CH_4 is a potential option for lamb production systems to improve the genetics of the overall flock. This would mean animals that naturally produce lower CH_4 would be selected for breeding. Long-term experiments conducted in New Zealand have found substantial changes when breeding is selected for low CH_4 output (Rowe et al., 2019). This work found that over a period of 10 years, selective breeding could reduce CH_4 output by 10-12% (g CH_4 kg dry matter intake (DMI)$^{-1}$). There were no negative effects on other traits as a result of selecting for low CH_4. This scenario therefore applies a 10%

decrease in CH_4 output across animals in all altitudes. The lower end of the range was chosen in order to provide a more conservative estimate of potential savings, as breeding programmes with a high impact would take several years to implement. No other changes are made to the model.

Scenario 2 – Breeding for improved feed efficiency

Improving the efficiency by which animals convert feed into product is another opportunity to reduce the GHG burdens of lamb production systems. To simulate this, the gross energy (GE) requirements of the animals are decreased 10% while keeping productivity the same. The value of 10% was chosen in keeping with previous studies of this value being a reasonable decrease possible within a selective breeding programme (Alcock et al., 2015). This also allows a direct comparison with breeding for low CH_4 output. No other changes are made to the model.

Scenario 3 – Increasing the number of lambs per ewe

At the case study farm, the ratio of lambs to ewe is regarded as low (0.78 lambs/ewe). This means that each ewe across the flock produces less than one lamb per year, on average. As CHUM farms the Welsh Mountain sheep, twin lambs are not encouraged to reduce lamb mortality. However, this scenario assumes that the number of lambs per ewe is increased to 1 (from 0.78). No other changes are made to the model.

6.2.3.2 Farm management and animal movement scenarios

These scenarios involve movement of animals and changes to farm management to assess if GHG mitigation can occur across the altitudes. These scenarios potentially result in land sparing to occur, with the possibility of using spared land for other purposes such as afforestation, therefore promoting C sequestration.

Scenario 4 – Increased number of ewes and ewe lambs in hill pasture

Lamb produced in hill pastures have thus far been assessed as having a higher CF than lamb from lowland sites (EBLEX, 2012; Jones et al., 2014a). However, with evidence suggesting the potential for decreased emissions from these pastures (see Chapter 3), it may be that emissions could be reduced by maximising grazing of pastures that cannot be used for other purposes – although there may be a trade-off between GHG savings and product output.

CHUM has grazing rights for approximately 1800 sheep on the hill common grazing land. However, the flock is currently less than that number so there is a potential to increase the number of ewes and ewe lambs in the hill pasture (rams and ram lambs are not grazed on the hill). This scenario therefore increases the number of ewes and ewe lambs grazing the common land from the months April to October. Numbers are increased by 10, 25 and 50% where possible, recognising that numbers may be limited by participation in agri-environment schemes. The additional animals will be subtracted from lowland and upland sites proportionately and dependent on the number of animals in each pasture during each month.

Increasing numbers +25% and +50% were not investigated for October, because there were fewer ewes on site at this time of year overall and most are already in the hill pasture, therefore it is not possible to increase numbers by this magnitude. There were no ewes in the upland in October, therefore the 10% extra is taken from the lowland. No other changes to the model are made, other than changes to the ewe numbers as described above.

Scenario 5 – Moving lambs to lowland pasture and creep feeding to finish quicker

This scenario explores if intensifying production by creep feeding animals so that they are finished and sold quicker would result in reduced GHG burdens on farm. All animals that are sold are fed to boost growth rate, allowing all animals to leave the farm by end of September. Sold animals that are assumed to be grazing in upland and hill pastures are moved to lowland pasture within the model to enable creep feeding. A growth rate of 150g per day was chosen, in keeping with a realistic target for the Welsh Mountain breed as provided by expert opinion. The growth that occurs already due to consumption of grass was derived from calculations within the model. To calculate the required level of concentrate needed to obtain the higher growth rate above what was already provided by the grass, a feed conversion ratio (FCR) of 5:1 was assumed based on industry data (Hybu Cig Cymru, 2018). This means that for every 5 kg of feed, the lamb could gain 1 kg in weight. In summary, changes were made within the model to the number of animals in the lowland, the month some animals left the farm and the level of concentrates consumed by the lambs sold. No other changes were made.

Scenario 6 – Increasing grazing capacity and land sparing in upland pastures

This scenario assesses the effects of improving the upland pastures to allow increased sheep numbers in some areas, sparing land elsewhere for afforestation. It is assumed that lime and fertiliser are applied in the upland area to increase grass growth, and then the additional animals that the grass growth could provide are moved from other farm areas.

Experimental work conducted within the same site found a grass growth of 12.09 kg DM ha^{-1} day^{-1} (increased from 8.22 kg DM ha^{-1} day^{-1}) after modest lime and fertiliser inputs (Williams et al., 2021). Data from this study were used to estimate the increased grass availability across the upland pasture based on the above numbers. DMI requirements of the flock were estimated using animal numbers, which then allowed the ratio of grass requirements to grass available at the control and improved growth rates to be determined. The difference between these allowed the fraction of spared land to be calculated. The required fertiliser inputs for the area were then calculated, using the fertiliser module within the model. It is important to note that it is possible that the fertiliser treatment could affect grass quality and therefore emissions. The results of Williams et al, 2021 indicate a small improvement to grass quality in terms of metabolisable energy (following fertiliser and lime application only) over the control treatment, although no significant block effect was determined between the treatments. However, as no relationship between the treatment and digestibility was shown, no changes to the digestibility within the model were made for this analysis.

6.2.4 Assessing potential for woodland C sequestration

The final aim of this chapter was to assess the potential for C sequestration if the assessed mitigation scenario resulted in potential for land to be spared. To calculate this, a literature value of the annual C sequestration rate of one hectare under woodland regeneration was multiplied by the estimated area (hectares) of land spared in each scenario. The chosen value was 3.6 t C ha^{-1} $year^{-1}$, derived from an average of a variety of studies assessing forest C sequestration (Searchinger et al., 2018). This global average was used due to a lack of available recent UK data for soils of this type. This value may therefore over or underestimate the sequestration that occurs. C sequestration was then converted to CO_2 removal via a molecular weight conversion calculation (44/12 multiplied by the mass of C sequestered). The modelling completed here did not account for soil carbon sequestration, as the model itself was not built to include measures of this at this time. This analysis therefore estimates the carbon sequestered by woodland creation only.

The estimate of land spared was calculated by taking the average annual stocking rate (livestock ha^{-1}) across the farm to determine the potential hectares saved when moving animals to other areas of the farm. This does not result in a precise estimate as the stocking rate may change monthly in each pasture, however it does provide an indicative level of sequestration that could be explored and refined further with more precise data in a further study. The calculated amount of CO_2 sequestered was then added to the emissions saved in each scenario to estimate potential for further emissions savings via woodland creation.

This analysis was applied to scenario 4, 5 and 6 (although land spared in scenario 6 was calculated differently as described in the previous section). In scenario 4, moving more ewes to the hill pasture may result in land sparing in the upland and lowland areas that could then be forested. In scenario 5, finishing animals quicker may reduce the land required on-farm for grazing, allowing other uses to be considered. Another aspect is the importance of animal health, where improvements to this could alter grazing requirements on farm and therefore consideration of these mitigation scenarios. Animal health is not assessed in this analysis but is likely to have an effect in reality.

6.3 Results

6.3.1 Breeding for low methane

Breeding for low CH_4 output resulted in a percentage decrease in both annual farm GHGs and product CF (Table 6.2).

Table 6.2 Breeding for low CH_4 results on CF (kg CO_2eq kg LW^{-1}) and annual GHGs (t CO_2eq)

	IPCC disaggregated	Site-specific
Effect on CF (% change)	-8.9	-9.3
Effect on annual farm GHGs (% change)	-8.9	-9.3

The effect on CF and annual GHG emissions was the same for both footprint, with site-specific footprints seeing a slightly bigger decrease at 9.3%.

6.3.2 Breeding for improved feed efficiency

Breeding for improved feed efficiency resulted in a percentage decrease in both annual farm GHGs and CF (Table 6.3).

Table 6.3 Breeding for improved feed efficiency results on CF (kg CO_2eq kg LW^{-1}) and annual GHGs (t CO_2eq)

	IPCC disaggregated	Site-specific
Effect on CF (%)	-8.9	-9.3
Effect on annual farm GHGs (%)	-8.9	-9.3

As with breeding for reduced CH_4 output, the effect on CF and annual GHGs was the same for both footprints, with site-specific footprints seeing a slightly bigger decrease at 9.3%. The scenarios of breeding for reduced CH_4 and for improved feed efficiency had the same magnitude of effect on both the CF and annual farm GHGs, indicating that both breeding strategies represent similar potential for reducing on-farm GHGs. An additional implication to consider is that if feed efficiency is improved, less land would be required for grazing and feed production. This may in turn spare land for other purposes such as woodland creation on farm - though that is not included in the calculation.

6.3.3 Increasing the number of lambs produced per ewe

Increasing the number of lambs per ewe from 0.78 to 1 resulted in changes to both the CF and annual GHGs on farm (Table 6.4).

Table 6.4 Effect of increasing the number of lambs produced per ewe on CF (kg CO_2eq kg LW^{-1}) and annual farm GHGs (t CO_2eq)

	IPCC disaggregated	Site-specific
Effect on CF (%)	-11.4	-12.1
Effect on annual GHGs (%)	+6.2	+5.4

There was a moderate decrease in the CF, at just over 10%, for both methodologies, although the decrease in the site-specific footprints was slightly larger (0.7% more). However, annual farm GHGs increased in both cases at the farm level with the IPCC disaggregated footprint showing a greater increase. Additionally, the product output increased from 28.9 t LW leaving the farm to 34.6 t LW leaving the farm: a change of 20%. The increase in product output is due to the increase in number of lambs being born and therefore sold within this scenario. This increase in animal numbers explains the increase in annual farm GHGs. However, the increase in output also results in the ratio of emissions to product output decreasing, which is reflected in the lower CF value.

6.3.4 Increased number of ewe and ewe lambs in hill pasture

Increasing the number of ewe and ewe lambs had a differing effect dependent on what footprint results are considered (Table 6.5).

Table 6.5 Increasing number of ewes in hill pasture effect on CF (kg CO_2eq kg LW^{-1}) and annual GHGs (t CO_2eq)

10% increase in ewe numbers		
	IPCC disaggregated	**Site-specific**
Effect on CF (%)	+0.8	-0.5
Effect on annual GHGs (%)	+0.8	-0.5
25% increase in ewe numbers		
Effect on CF (%)	+0.8	-2.3
Effect on annual GHGs (%)	+0.8	-2.3
50% increase in ewe numbers		
Effect on CF (%)	+3.1	-3.5
Effect on annual GHGs (%)	+1.3	-5.1

In all increases of animal numbers, the IPCC disaggregated footprint resulted in an increase of both CF and annual GHGs. In contrast, using the site-specific footprint resulted in a decrease in all cases with the 50% increase in ewe numbers in the hill site having the greatest decrease on both CF and annual GHGs. While the changes were modest, this scenario represents the differences that may occur when using different data inputs.

6.3.5 Moving lambs to the lowland to creep feed and finish quicker

This scenario resulted in an overall decrease in farm CF, yet an increase in annual GHGs (Table 6.6).

Table 6.6 Effect of creep feeding and finishing lambs quicker on the CF (kg CO_2eq kg LW^{-1} and annual GHGs (t CO_2eq)

	IPCC disaggregated	Site-specific
Effect on CF (%)	-7.3	-6.8
Effect on annual GHGs (%)	+1.5	+2.2

The CF had a reasonable decrease close to 7% for both methodologies, with the IPCC footprint having a slightly bigger increase then the site-specific. The annual GHGs saw a small increase at approximately 2%, although the site-specific footprint saw a bigger increase than the IPCC. In addition, this scenario resulted in a 9.6% product output increase from 28.9t LW to 31.7 t LW.

6.3.6 Comparison of mitigation scenario ranking between footprints

Comparing the results of the different scenarios on the footprint showed similarity in effect of some scenarios (Figure 6.1).

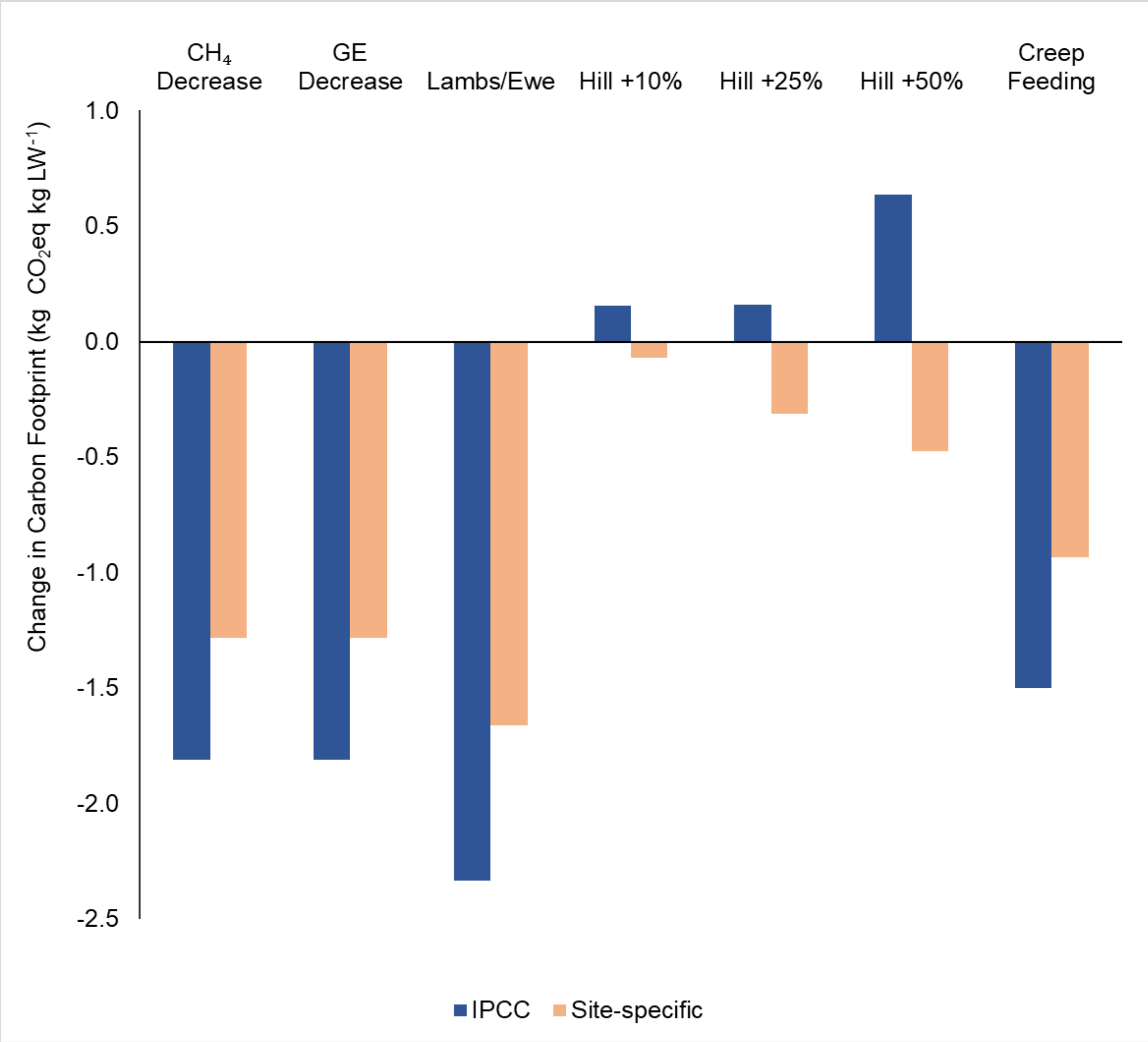

Figure 6.1 Comparison of change in footprint values (kg CO_2eq kg LW^{-1}) for each scenario using IPCC disaggregated and site-specific footprints

Increasing the number of lambs/ewe resulted in a clear decrease in footprint value of between 1.5 and 2.5 kg (CO_2eq kg LW^{-1}). In general, the IPCC disaggregated, and site-specific footprints resulted in a similar direction of movement (decrease) in footprint, with the IPCC disaggregated footprint showing a greater decrease. A key difference are the scenarios involving moving an increased number of ewes to the hill. These scenarios had the smallest change in footprint overall, but it can also be seen that these scenarios increased the IPCC disaggregated footprint, whilst the site-specific footprint decreased.

Similar general trends were seen when comparing change in annual farm GHGs (Figure 6.2).

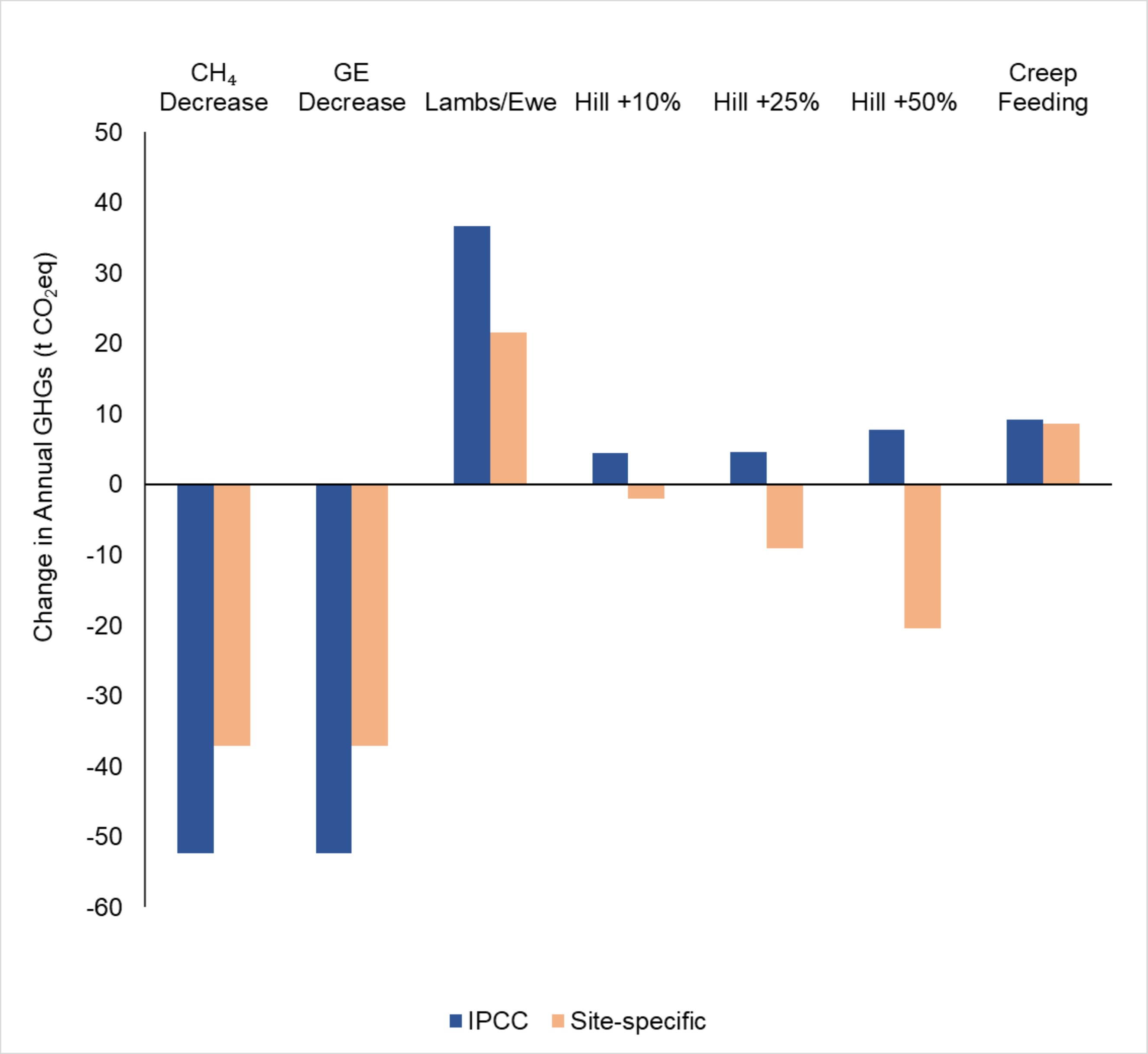

Figure 6.2 Comparison of change in annual farm GHGs (t CO_2eq) for each scenario using IPCC default and site-specific footprints

The strategies aimed at reducing CH_4 output had the greatest effect in reducing overall farm emissions. Both the increasing lambs per ewe and creep feeding scenarios resulted in an increase in annual farm GHGs, whilst simultaneously decreasing the product CF. A similar effect was seen on annual farm GHGs as was seen on the CF value in the increasing the number of ewes in the hill pasture scenario. The IPCC disaggregated footprint resulted in an increase in farm GHGs, whereas the site-specific footprint resulted in a decrease in farm GHGs. This is likely due to the lower EF values in the hill site in the site-specific footprint in comparison to the IPCC default values and further highlights the differences that may occur when using different sources of input data.

6.3.7 Increasing carrying capacity and land sparing in the uplands

The calculated kg dry matter (DM) available in the upland pasture as well as the estimated total DMI requirements of the grazing animals (daily average) are shown in Table 6.7.

Table 6.7 Total grass DMI requirements (daily average, kg DM) of animals grazing the upland pasture and estimated grass availability (kg DM day^{-1}) in control and improved pasture

	kg DM
Total animal requirements (kg DM day^{-1})	849
Control DM production (kg DM day^{-1})	1220
Improved DM production (kg DM day^{-1})	1800

The difference in ratio of total DMI requirements and grass availability between the control and improved pasture was then used to calculate the fraction of area spared after improvement. This was determined as 33.4 ha being spared from an area of 148.8 ha, with 115.4 ha being improved. The potential C sequestered from woodland creation on the 33.4 ha spared was then calculated. This resulted in the total annual farm GHGs decreasing in both the IPCC disaggregated and site-specific footprints, with a decrease of 50% and 74%

respectively (Figure 6.3). This was despite an increase in fertiliser emissions from improving grass availability.

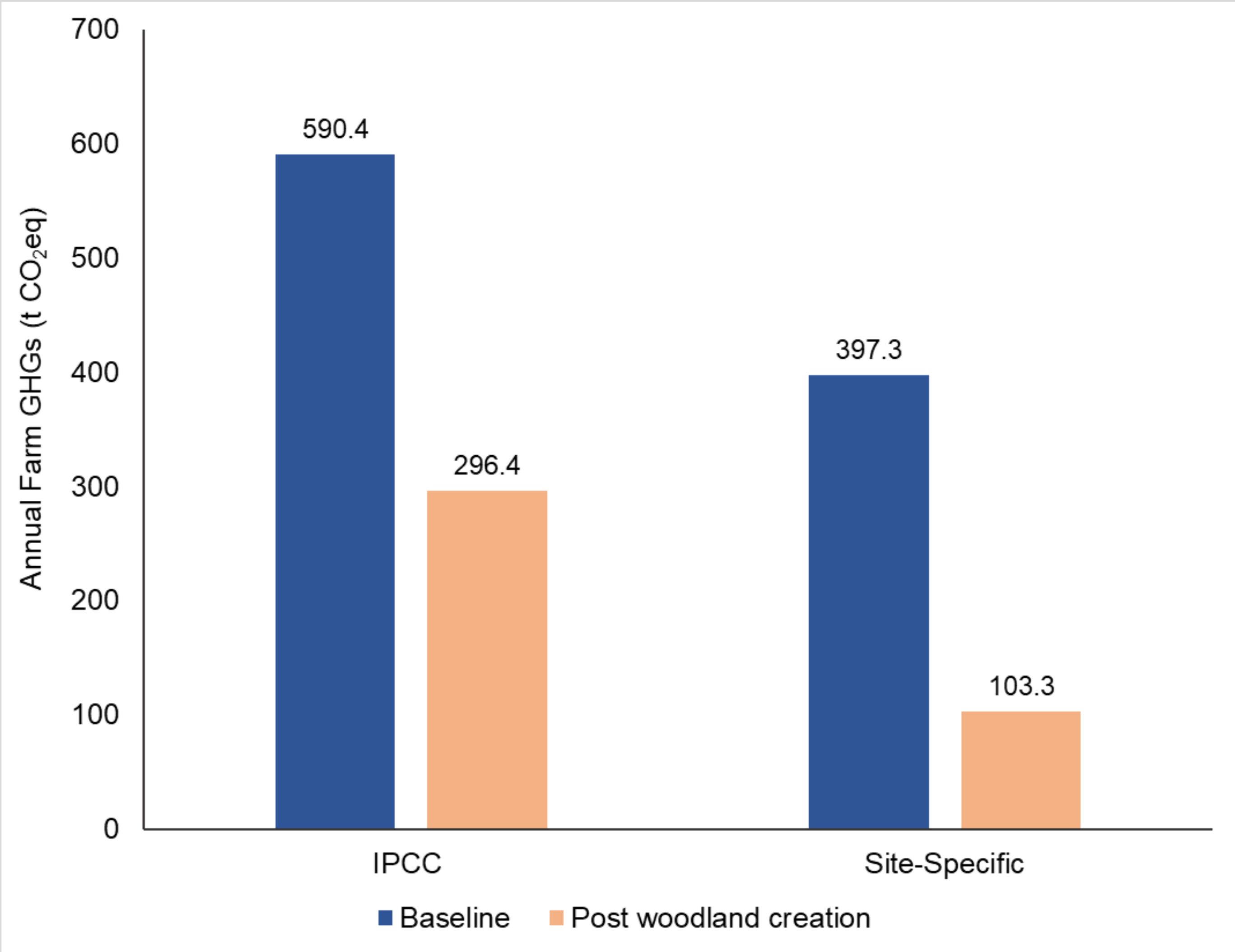

Figure 6.3 A comparison of IPCC and site-specific calculated annual farm GHGs (t CO_2eq) in the baseline footprint and after woodland creation

6.3.8 Other potential for woodland creation

Assessment of the two other animal movement scenarios found potential for afforestation and reduction of annual farm GHGs using the IPCC default footprint (Figure 6.4).

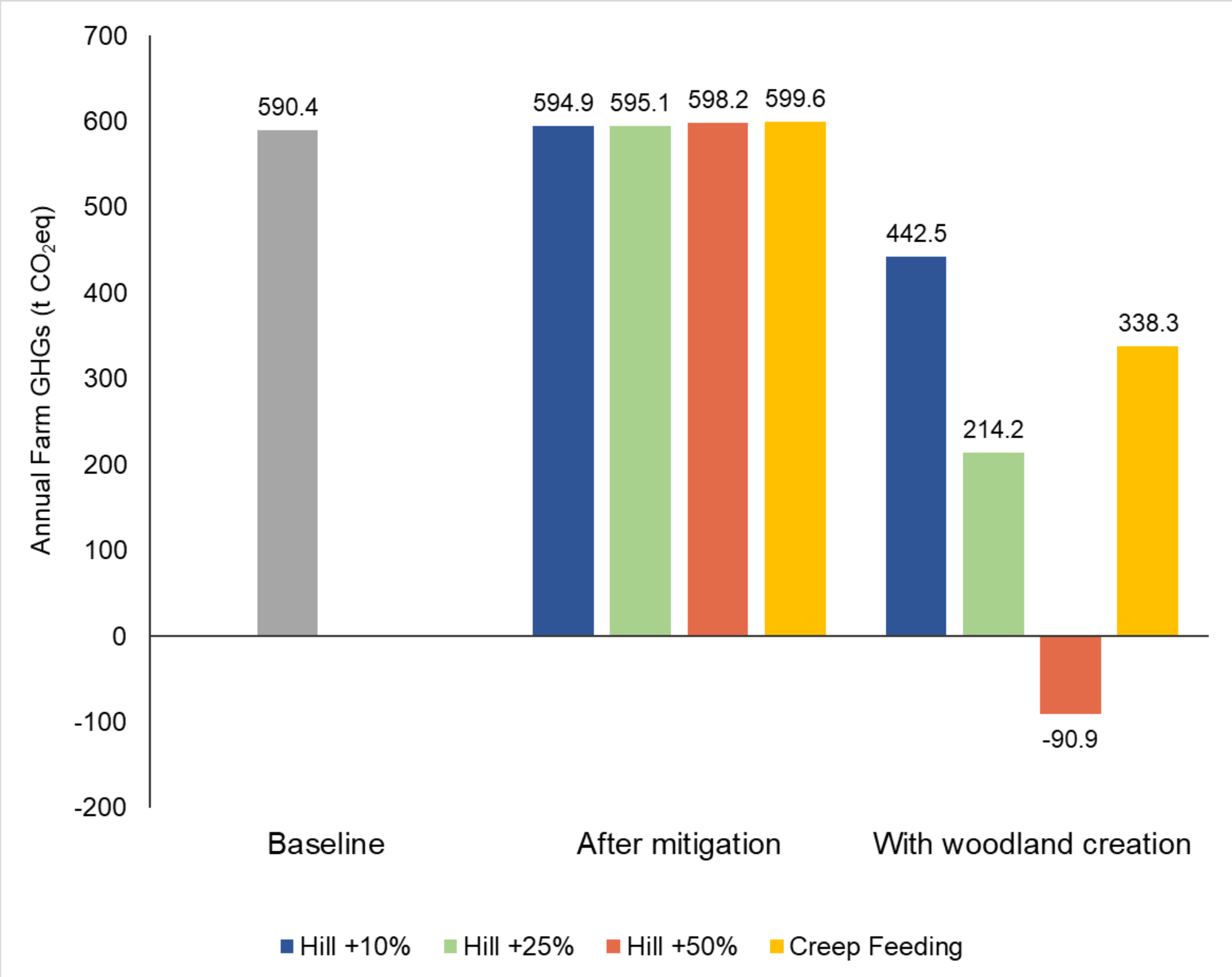

Figure 6.4 Comparison of annual GHGs (t CO_2eq) (baseline, after mitigation strategy is applied and with woodland creation) across four mitigation scenarios using IPCC default footprint

In all scenarios, a decrease in annual farm GHGs was found. The scenario involving increasing ewe numbers in the hill allowed a greater effect of sequestration as the number of animals moved increased. Increasing the number of ewes in the hill by 50% resulted in a net overall sink, as did the release of land from grazing by creep feeding lambs to finish earlier.

The same pattern was seen when applying sequestration rates to the site-specific footprint, although with greater magnitude (Figure 6.5).

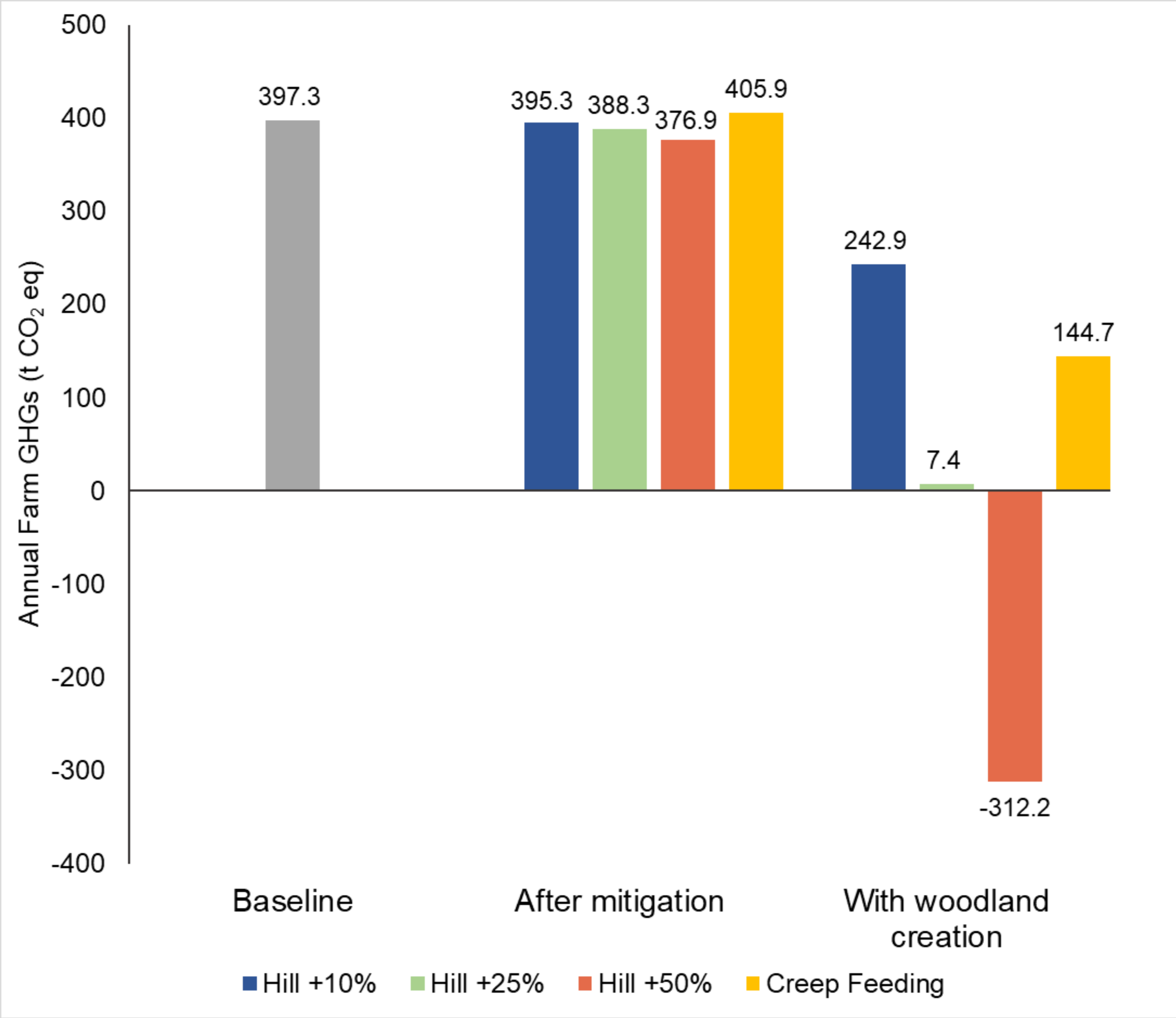

Figure 6.5 Comparison of annual GHGs (t CO_2eq) (baseline, after mitigation strategy is applied and with woodland creation) across four mitigation scenarios using site-specific footprint

As discussed above in section 6.3.4, the site-specific footprint resulted in a decrease in emissions when the mitigation strategy of increasing ewe numbers in the hill was applied. The baseline value was therefore lower, resulting in a bigger net sink for the 50% increase of ewes in the hill than in the IPCC disaggregated comparison. As before, this is likely due to the lower EFs used in the hill pasture. A considerable decrease in annual farm GHGs was also seen in the creep feeding scenario. Overall, despite the footprint methodology selected, the analysis suggests a potential for woodland creation to occur on farm by manipulating animal movements.

6.4 Discussion

6.4.1 Mitigation options in comparison to wider literature

The mitigation scenario here that appeared to have the greatest influence on farm footprint was increasing the number of lambs per ewe. A previous study showed similar findings, in that improving productivity of the ewe and survival of the lambs showed a positive effect in both reducing the CF as well as being cost effective to the farmer(Jones et al., 2014a). The effect of creep feeding to finish lambs quicker was also assessed in the aforementioned study, with a small reduction in CF. However, it came with a high cost to the farmer. This matches results of the current study as creep finding resulted in a small CF decrease (6.6 – 7.3%) but the cost was not assessed, so it is therefore possible that it would not be a cost-effective mitigation option given this small decrease. A different study found that larger CFs were correlated with farms that have an increased time in sending lambs to slaughter but also those with increased concentrate use, which differs to the lower CF from creep feeding found in this thesis (Hyland et al., 2016). Overall, an increase in production efficiency, in terms of utilising inputs more effectively, was the most effective mitigation option assessed. Collectively, these results indicate that there is potential to strike a balance in finishing lambs quick enough to not increase the CF significantly, whilst utilising inputs as effectively as possible.

Both breeding for lower CH_4 output and increased feed efficiency had moderate reductions in the farm footprint, as well as annual GHGs. As discussed previously, it is known that the CH_4 that arises from enteric fermentation is a considerable contributor to the CF, particularly for low-input lamb production systems. It has therefore been found that strategies that aim to improve feed efficiency as well as manipulate enteric CH_4 have been identified as effective options for the reduction of sheep farm GHG emissions (Escribano et al., 2020). An additional option is the improvement of forage digestibility, which is known to have a significant effect on enteric CH_4 output (Eugène et al., 2021). While not assessed as a mitigation option here, forage digestibility was found to have notable sensitivity to the footprint in Chapter 4 and 5 of this thesis.

A factor not assessed here that may have implications for the results is possible interaction across different mitigation options and the changes that may have to be made in farm management to achieve the strategy. For example, increasing the number of lambs per ewe improved the product CF of the farm, however it was not assessed here which specific actions would need to be implemented to achieve that aim. Increased LW of the ewe has been linked to a greater number of weaned lambs per ewe across a variety of breeds (Thompson et al., 2021). Increasing the number of lambs per ewe at CHUM therefore could be achieved by increasing ewe LW, which may be achieved by increasing concentrate feeding. The efficacy of this mitigation strategy would potentially be inhibited by an increase in inputs and therefore GHG emissions associated with said inputs. For the creep feeding scenario and improving upland scenario, effort was taken to account for the increased concentrate and fertiliser use involved.

6.4.2 Improving site-specific information can aid mitigation scenario analysis

While there are a variety of mitigation options studied and suggested as a potential for reducing GHGs, it is known that these are not always applicable to every farm and specific systems should be individually assessed for effectiveness of different options (Smith, 2012). In Chapter 4 of this thesis, the effect of using site-specific inputs on the footprint was shown. This results of this chapter further represent those findings, in that differences were seen across different mitigation options when comparing default footprinting against footprinting that used site-specific parameters. In some of the scenarios here, the same effect (e.g., increase or decrease) was seen between the footprint types, but with a different magnitude of effect. However, in the case of increasing the number of ewes grazing in hill areas, the IPCC footprint saw an increase yet the site-specific saw a decrease in CF. This difference is likely due to there being a smaller difference between forage digestibility values in the lowland and hill sites in the site-specific footprint vs. the IPCC disaggregated footprint. The measured hill pasture forage digestibility utilised in the site-specific footprint is also higher, resulting in lower enteric CH_4 being calculated. The site-specific value being higher than the IPCC disaggregated footprint is due to choosing to apply the IPCC guidelines digestibility value for animals consuming low quality forage for upland and hill pastures. A conclusion from this work may consequently be that the chosen IPCC value is not appropriate for the upland and hill pastures assessed here. Differences in EFs may also contribute.

Site-specific model parameters, both in terms of EFs and farm data, may be more difficult to generate, but application of more farm-specific data within footprinting may aid evaluating effectiveness of mitigation options and prevent unintended negative trade-offs. Global analyses of beef systems have found that effectiveness of different mitigation options varied across different regions, resulting in different recommendations (Cusack et al., 2021). For example, improving on-farm efficiency and improving farm management for C sequestration was a dual approach to reducing C intensity of Brazilian beef production. Whereas in the United States, focus on facilitating C sequestration would have greatest effect. While farm-level is much smaller scale, the necessity to consider spatial variation persists.

While this study attempted to utilise site-specific data, limitations to this data exist that resulted in key areas of uncertainty. When moving ewes to the hill, it is assumed that there is land spared elsewhere on farm as a result of increasing the ewe numbers on the hill. However, the ewes can only be grazed on the hill from April to October and this scenario does not account for ewe management out with the time periods they can graze on the hill. While that is a source of uncertainty, it indicates that there is a potential to spare land by manipulating animal movements around farm provided the full year can be accounted for.

The area of land spared calculated in Scenario 4, 5 and 6 may be overestimated due to limitations in the site-specific information collected. For example, average number of animals are used to determine DMI requirements in Scenario 6, when in reality animal numbers vary throughout the year. Additionally, uncertainty in data surrounding precise animal numbers and animal weights affect calculation of the DMI requirements, the stocking rate, and the grass utilisation; all of which are used to derive area spared across the three scenarios. Other sources of uncertainty are present in the analysis, across all scenarios. When estimating the CH_4 reduction possible through breeding, data from NZ experiments with different breeds were used, therefore UK-specific data would provide better estimates of the emissions reductions possible via this route for the sheep breeds available in this country. This is true also for breeding for improved feed efficiency, as the data underpinning this scenario is from an Australian study. Furthermore, Scenario 5 involved use of a feed conversion ratio to estimate the weight gain possible from creep feeding. Increased understanding of this value for Welsh Mountain sheep would give greater accuracy for this scenario. Overall, improving understanding of these different model parameters, particularly for extensive lamb production systems would aid in increasing robustness of sequestration estimates and indeed mitigation scenarios as a whole. While research assessing site-specific emissions and input data variations alongside mitigation options is lacking, the increasing use of CF and LCA will aid in facilitating selection of the mitigation scenarios with greatest effect.

6.4.3 Wider methodological implications for mitigation scenario analyses

It is also important to consider other ecosystem services derived from extensive livestock systems, particularly in comparison to intensive systems with higher productivity. On a primarily GHG basis, intensive systems often have a lower CF in comparison to extensive systems (Escribano et al., 2020). The benefits of grazing management, such as biodiversity improvements and control of wildfire or flood risk are not captured. It has been evidenced that including wider outputs of lamb production systems as well as market income can result in extensive grazing having a lower CF relative to the more intensive systems (Ripoll-Bosch et al., 2013). This was partly due to the soil C sequestration that may be present within extensive grazing systems. An assessment of national inventories found that increased soil C sequestration under extensive grazing outweighed the C produced, resulting in an offset of these emissions (Viglizzo et al., 2019). However, uncertainties related to the knowledge around soil C saturation were acknowledged. Not fully assessing these wider benefits may lead to system changes that reduce GHGs but have negative effects on other valuable farm outputs. Conversely, assessment of wider negative environmental impacts that arise as a result of lamb production, such as ammonia emissions and eutrophication, are also not covered by CF but are equally important in achieving long term sustainability.

The presentation of results and desired outcomes is an additional important consideration. For example, in the creep feeding scenario and increasing lambs per ewe, there was a decrease in footprint yet an increase in annual farm emissions. In these cases, the farm improved its efficiency but in the context of net zero targets, absolute emissions must also be considered. This links additionally to the point above regarding intensive vs. extensive systems. Intensive systems may appear more productive, but in terms of absolute emissions smaller mixed grazing systems may have lower total emissions output than large intensive livestock systems (Garnett, 2011). It has been suggested within a global review that focus purely on the productivity measurements, such as unit of product (kg or litre) may not be the most effective approach for fully assessing the wide variety of livestock systems present worldwide (Rivera-Ferre et al., 2016).

Additionally, there are wider environmental impacts, including water supply and quality, soil quality and ammonia emissions that must be mitigated for long-term sustainability. Excessive focus on reducing the C impact may result in undesired effects in other environmental impacts (Röös et al., 2013). A recent study valued agricultural ecosystem services as worth £1,434.02 ha^{-1} $year^{-1}$ and forestry as worth £1,261.09 ha^{-1} $year^{-1}$

(Hardaker et al., 2020). Agriculture mostly delivered provisioning services whereas forestry provided greater amounts of public ecosystem service benefits than agriculture. This highlights that each have different societal positives to bring, and effective integration could allow advantage to be taken from both. It is therefore important to consider the desired outcome of any assessments, particularly when designing strategies for policy making.

6.4.4 Positives and negatives of on-farm land use change

Increasing tree cover on agricultural land is discussed regularly when considering offsetting agricultural GHG emissions, and so was assessed here for its potential within a Welsh farm. Selected scenarios in this analysis were considered as having potential to spare land for woodland creation. For example, increasing grazing of the hill pastures could allow other areas of the farm to be released for woodland creation. Additionally, creep feeding animals to finish them quicker reduces their time on the farm and the grazing land required. Furthermore, there is potential for improving feed efficiency of animals to also result in decreased grazing land required as their feed requirements would reduce. However, the potential for this was not assessed in the current study. Sustainable intensification, by increasing fertiliser inputs in upland areas resulted in land being released for woodland creation. The C impact of introducing woodland has been discussed and presented in the results here, with all scenarios resulting in reduced annual farm GHG output.

Changing land use has the potential to bring many wider benefits to farms and the environment. For example, It has been found that sharing land between agriculture and forestry in the Welsh uplands provides the most significant ecosystem service delivery, in comparison to fully sparing land for afforestation (Hardaker et al., 2021). Afforestation also offers a diversification of income for farmers, particularly for smaller-scale farms that may struggle with income (Duesberg et al., 2014). However, there are costs involved in implementing woodland in terms of equipment and set up costs. Animal welfare can be improved from introducing trees, as sheep that were provided access to trees had less stress indicators than those that were grazing in open areas. Certain tree species, such as willow, comprise anti-inflammatory constituents that give animals grazing on them natural medicinal benefits (Muhammad et al., 2022).

The drawbacks that are present with introducing forestry must also be considered before implementing a blanket land use change. Inherently, there may be a reduction in the key output of the farm, which hampers farmer uptake but may also impact food security depending on where and how it is implemented (Sagastuy and Krause, 2019). Issues of practicality exist. The farm used as a case study here is a mixed altitude farm and the introduction of forest may be hampered by local environmental limitations (e.g., land slope affecting tree growth) that cannot be controlled. Socio-economic impacts may occur, with potential negative consequences for lower-income and marginalised groups as access to land, knowledge and labour for the introduction of agroforestry is inequitably distributed

(Hastings et al., 2021). Consideration of the forestry included is another aspect, as planting of monoculture plantations increases forest cover and forest sequestration but may not be optimal for wider ecosystem services (Mylliemngap, 2021).

A global review found a substantial potential for climate change mitigation using afforestation. Approximately 4.9 $GtCO_2$ $year^{-1}$ abatement was calculated, and implementation of afforestation was found to potentially reduce the costs of climate change mitigation. However it is recognised that it would involve large amounts of land and issues with implementation and permanence of storage represent major risks (Doelman et al., 2020). Overall, understanding around C balance changes during land use change requires further research to fully assess afforestation potentials. The ability of grassland itself to effectively sequester C is of importance, as improving grassland management has sequestration potential equivalent to that of afforestation as well as often resulting in increased farm productivity (FAO, 2010). Generally, the benefits of implementing forestry on agricultural land is dependent on the land that was originally in place and while it can result in improved ecosystem services and land management, local context should be considered before the strategy is used (Torralba et al., 2016).

6.5 Conclusions

Assessment of different mitigation options is essential in reaching net zero within the agriculture sector and reducing its impacts on the environment. Strategies to improve production efficiency as well as reduce the CH_4 output of a system appear particularly effective in reducing the CF of lamb production systems. To further develop CF methodologies and improve assessment of mitigation strategies concurrently, improving and developing understanding of site-specific factors could aid in increasing accuracy of results in the future. For lamb production and extensive systems in particular, a better understanding of emissions burdens and animal data will contribute to this accuracy as a lack of representative data introduced uncertainty within this study that could be reduced. Alongside assessing C reduction of agricultural systems, developing understanding of wider ecosystem services can help to fully assess gains and losses from a particular system. Understanding the desired use of results, for example for farm level mitigation or for higher level policy making, will aid in correctly interpreting results. There is a potential for mitigation options to work alongside increasing afforestation levels on farm, possibly allowing partial or complete offsetting of farm emissions. However, a better understanding of C sequestration from land use change to forestry across different land types is essential in fully assessing this potential. Afforestation comes with positives and negatives depending on the farm and local context is important to consider when assessing implementation.

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Chapter 7 Improving lamb production carbon footprinting: A summary of generated results and discussion of the wider implications

7.1 Summary and discussion of the key findings

The key aim of this thesis was to assess greenhouse gas (GHG) emissions changes across altitudes present in lamb production systems and incorporate these changes into a purpose-built carbon footprint (CF) tool (see Chapter 1). Through assessing the available literature for lamb production and wider livestock systems (see Chapter 2) it became evident that, while significant progress has been made in understanding livestock derived GHGs, various limitations exist. Lamb production systems remain a key part of the United Kingdom (UK) and Welsh agricultural sectors, with a large amount of grazing occurring in higher altitude pastures for example upland and hill sites (Williams, 2012).

The literature review summarised the key sources of GHGs from these systems as well as limited works that indicate a potential for differences in emissions across altitudes. The application of CF and life cycle assessment (LCA) to lamb production systems was described and through critical evaluation, an understanding of the key data gaps that were limiting development of methodologies was achieved. It became clear there was a lack of empirical data quantifying emissions changes across different altitudes as well as other data generally used for CF inputs being outdated. For example, there were limited data available that categorised forage digestibility; an input with big implications for methane (CH_4) outputs within modelling. This allowed research gaps to be identified and the work within this thesis to be designed towards filling those gaps.

Experimental work focussed on quantifying emissions changes across altitudes and modelling work focussed on development of an altitudinally disaggregated CF tool. These two pieces of work combined to enable improved evaluation of mitigation strategies at a farm level. The works presented in each chapter, alongside the thesis objectives and wider research, are further summarised and discussed below.

7.1.1 Assessment of changes in enteric methane and excretal nitrous oxide across altitudes in a lamb production system

Measurement of GHG burdens from livestock systems, in particular enteric CH_4 and excretal N_2O, has been the subject of research for some time. However studies where animals are grazing naturally are limited, which has caused concern regarding the applicability of collected data for different grazing systems (Richmond et al., 2015). For lamb production systems in the UK where grazing largely occurs on upland and hill areas, this issue is of greater concern. The different environmental and soil factors, forage species and grazing behaviour in these sites all have implications for the level of emissions produced by the animals. A key objective of this study was to assess those changes across lowland, upland and hill sites within a lamb production system (Chapter 3).

Regarding enteric CH_4 emissions, this study found changes in emissions between upland, lowland, and hill sites in terms of average CH_4 output across the animals and derived CH_4 conversion factor (Ym). This is the first study of its kind to measure lamb production enteric CH_4 using the GreenFeed measurement system in this way, but other studies using different methodology also point to differences in output related to various environmental factors (Zhao et al., 2016; Pinares-Patino et al., 2003; Zhao et al., 2017; Archimède et al., 2018). The IPCC recommends a default of 6.7% Ym to be used, which matches closely to the upland value derived in this study but is lower than the lowland value and higher than the hill value (IPCC, 2019). The use of Ym itself has been critiqued related to its inability to capture changes in animal diet and rumen characteristics (Moraes et al., 2014). It is therefore reinforced that assessment of lamb production enteric CH_4 requires refining, either by improving Ym estimates or developing wider models that may better capture the variability of emissions (either due to diet changes or other factors such as breed). The UK inventory has gone some way in attempting to improve lamb production enteric CH_4 estimations by creation of a country-specific calculation (Brown et al., 2022). Further improvement could still be sought by additional collection of model inputs, such as better understanding of liveweight (LW) changes of animals grazing upland and hill pastures.

Regarding N_2O emissions, this study found that the combined excretal emission factor (EF_{3prp}) between lowland and upland sites also differed, with a higher emission factor (EF) in the lowland site in comparison to the upland site. This matches the findings of other studies conducted with sheep excreta at differing altitudes, mostly conducted within Ireland, the UK and New Zealand (NZ) (Luo et al., 2013; Mancia et al., 2022; Marsden et al., 2018). These results are positive in that they indicate upland and hill sheep have a reduced environmental burden in the context of N_2O emissions. The NZ GHG inventory report now disaggregates direct N_2O emissions from sheep urine by slope class: flat (less than 12° gradient) and medium and steep (greater than 12° gradient) (Ministry for the Environment, 2022). However, this change is not yet captured within IPCC and UK inventory methodologies where there is no disaggregation by altitude for lamb production systems. Between the 2006 and 2019 refinement to the IPCC methodologies, EF_{3prp} for grazing sheep was lowered from 1% to 0.3% (IPCC, 2006, 2019). The UK inventory currently scales the cattle derived EF by 50% for sheep, giving a value close to the IPCC 2019 refinement (Brown et al., 2022). While the reduction is appropriate given the evidence and close to the value generated for the lowland in this study, both of these values are still higher than the upland value here. This strengthens the necessity to further assess these EFs both in terms of values in different pastures and the potential for disaggregation, which could be by soil type or altitude as two examples. A better understanding of emissions changes, both in relation to enteric CH_4 and excretal N_2O, can aid in better targeting emissions reductions towards Net Zero.

7.1.2 Development of an altitudinally disaggregated model for lamb production systems, accounting for variation in selected inputs

LCA and CF are key approaches to aid in understanding the environmental burdens of a system, as well as identify opportunities for improving production efficiency (Geß et al., 2020). LCA of lamb production systems should take into account the pastural variation of the system (Williams, 2012), although thus far specific considerations of pasture-based systems has been limited within LCA (Bhatt and Abbassi, 2021). A key objective was therefore to develop a model that could disaggregate between pasture type (in this case lowland, upland, and hill). The model developed is an excel-based spreadsheet that is structured to utilise monthly animal activity data (numbers and weight) at each altitude, based on the IPCC Tier 2 approach. Additionally, EFs are disaggregated by altitude, with the ability to use either IPCC default EFs or EFs (including EF_{3prp} and Ym) generated from experimental data collected in this thesis (Chapter 3). Following model development, the results found that CF values differed markedly depending on whether IPCC 2006, 2019 or site-specific information were utilised. Moving from default methodologies to use of site-specific information resulted in a decrease in the footprint after disaggregation. However, all footprint results fell within the range found by a recent global sheep LCA review (Bhatt and Abbassi, 2021).

Many Welsh lamb footprints calculated until now have followed the IPCC 2006 Tier 1 approach, making direct comparison of results difficult (Edwards-Jones et al., 2009; Jones et al., 2014; Taylor et al., 2010). The IPCC 2019 refinement includes updated EFs based on a wider breadth of evidence, therefore using footprints generated with the 2006 methodologies will result in different conclusions. Extending this, use of IPCC 2019 Tier 2 methodology with disaggregation of parameters further resulted in different conclusions as mentioned above. The work here therefore indicates that default methodologies may cause inaccurate lamb production footprints, potentially overestimating them dependent on the system assessed. It also highlights that comparing CF with other published material requires caution as the methodologies followed may differ.

Following the UK leaving the European Union (EU), agricultural subsidy systems are undergoing reform. The UK Government and Devolved Administrations are now moving to focus the new schemes on improving environmental outcomes, linking payments to farmers with the provision of public goods (DAERA, 2022; DEFRA, 2022; Scottish Government, 2023; Welsh Government, 2022). It therefore ensues that farmers may wish to better understand their environmental impact and increase use of carbon (C) calculators. The potential to inform consumers and label food with its C intensity is also being discussed (De-loyde et al., 2022). It is therefore imperative that calculators are continuously improved to quantify farm GHGs in the most robust way. This research highlights the data gaps still currently present in lamb production systems and shows the variability that occurs when using different methodologies. The data collected in Chapter 3 and the modelling conducted in Chapter 4 showing differences in footprint after disaggregation further support the necessity to begin disaggregating EFs and model inputs where possible for lamb systems. Failure to not effectively capture the uncertainty and variability in emissions in relation to system characteristics, such as altitude of sheep pastures, could lead to decisions being made on incomplete evidence and ineffective policy.

7.1.3 Assessment of the sensitivity of the carbon footprint results to different input parameters

Sensitivity analysis is an important process to aid in understanding of model outputs as well as increase confidence in outputs (Lacirignola et al., 2017). It can aid in understanding the contribution of different model inputs to the results, therefore identifying the key areas of uncertainty within a model (Chen and Corson, 2014). In this context, the assessment of the effect of different parameters on the model output in Chapter 5 showed that a focus on improving forage digestibility factors across different pastures is important to improve estimations of lamb production footprints. Additionally other animal parameters such as growth rate and weight are important. This finding is useful in considering how to improve future agricultural surveys, for example increasing farm monitoring and collecting further data pertaining to the above factors could improve the UK National GHG Inventory, as well as CFs.

Furthermore, the results provide some insight into areas of research focus in the coming years. Given the greater effect of Ym on the footprint results in comparison to the N_2O EF_{3prp}, it is particularly important to increase understanding of this factor. Quantifying the N_2O EF_{3prp} is still of importance to better improve robustness of estimates as well as understand seasonal variation. However, very little work has been completed on enteric CH_4 under extensive grazing conditions. While the UK Inventory goes some way in disaggregating between lowland, upland and hill sites, estimation of enteric CH_4 remains the same across altitudes within the National GHG Inventory (Brown et al., 2022). Further information on this could therefore aid inventory reporting in addition to the recommendation above.

A key conclusion of the Chapter 5 results is that what the data are being used for and the desired outcomes for results must be considered when presenting them. This distinction is important for policymakers to consider when assessing different sources of evidence to support policy development. These results provide an explanation of this phenomenon and quantify the potential differences between different metrics (CF results and annual farm GHGs). This is further shown with the assessment of different global warming potential (GWP) methodologies as the footprint varied up to approximately 10% between different IPCC assessment report methodologies. Consideration therefore must be given to what methodology a study followed when comparing it to other evidence as results may differ.

This increases with scale as GWP values used within country inventories are likely to have bigger effects.

7.1.4 Mitigation strategy potential and opportunities for introducing woodland on-farm

In the most recent inventory, agriculture accounted for 15% of Wales's GHG emissions (Welsh Government, 2022). Decarbonisation of this sector is therefore imperative in reaching Industry and policy Net Zero targets, although is inherently difficult given the dispersed nature of emissions and varying production efficiencies across farms. Therefore, a key aim in this study was to assess different mitigation strategies in the context of the disaggregated methodology followed throughout the thesis. Increasing productivity on farm is often an effective way of reducing the product footprint (Ghosh et al., 2020). It therefore follows that the results of Chapter 6 showed the most effective mitigation option to reduce the CF was one that improved efficiency (increasing number of lambs per ewe). This aligns with other work assessing mitigation strategies for lamb production systems, where increasing the number of lambs per ewe and improving production efficiencies had the most significant effect on the CF (Hyland et al., 2016; Jones et al., 2014). However, this scenario alongside the scenario involving creep feeding animals resulted in an increase in annual farm GHGs despite the reduction in footprint. This shows that the farm productivity improved, however may conflict with Net Zero goals and delivery. These results build upon the results of Chapter 5 and show that it is important to consider the goal of implementing a strategy and fully understand the purpose of different metrics, otherwise strategies could be implemented that improve efficiency yet increase emissions.

This work provides further evidence of mitigation options that may be appropriate for lamb production systems specifically; an area that has had less research conducted than other livestock systems. Strategies that target enteric CH_4 (breeding for reduced CH_4 and improved feed efficiency) were also effective at reducing farm annual GHGs. Improving genetics to reduce feed intake or enteric CH_4 output has likewise been identified as a strategy to reduce the emissions intensity of lamb products (Alcock and Hegarty, 2011). However, the work required to develop industry breeding programmes was emphasised. Additionally, these results highlight the importance of considering characteristics specific to the farm in question. As with Chapter 3 and 4, the assessment of mitigation options here had different results depending on use of default methodologies or site-specific footprinting. In some scenarios, the overall effect was the same (e.g., increase or decrease) but with differing magnitude. In the case of increasing the number of ewes grazing the hill site, there was a divergence in effect indicating that for extensive sites in particular the default methodologies may not be fully capturing the specific context of the area. An overall key message of work completed throughout this thesis is therefore that modelling should be conducted with as much site-specific data as is available. These results can be used to aid in further informing upland land use decision making in the future.

The role of nature-based greenhouse gas removals (such as afforestation and peatland restoration) in meeting Net Zero targets has been highlighted by the Climate Change Committee (CCC) in their recommendation reports to the UK government (Climate Change Committee, 2020). It is estimated that agriculture as a sector will have a high level of residual emissions in 2050 that will need be offset by greenhouse gas removals. In addition to this, the Welsh Government currently has tree planting targets of 43000 ha by 2030. The Wales Low Carbon Delivery Plan states a target of 10% of agricultural land will be converted to tree planting by 2050 (Welsh Government, 2021). The results of Chapter 6 indicate that there is a potential to share the land with woodland creation whilst maintaining agricultural production and considering this on a farm basis could aid in ensuring trees are planted in the correct places without impacting sectors currently operating. Combining lamb production systems with afforestation has previously been shown as having potential to reduce environmental impacts while maintaining system productivity (Beckert et al., 2016). In Chapter 6, improving upland pasture and grass growth to decrease the land required for grazing was assessed and potentially allowed farm emissions to be entirely offset, depending on the methodology followed. A review of different studies assessing the potential for emissions offsetting with trees in ruminant systems likewise determined that at least some portion of emissions could be offset (Jordon et al., 2020). Improving grass growth has additionally been highlighted as an effective option to reduce the emissions intensity of lamb production systems (O'Brien et al., 2016), coinciding with the conclusions of this thesis. The work described here can therefore be used as a basis of informing further research in this area, to better quantify the area of land that can be used for woodland creation and further assess impacts to farmers of land use change.

7.2 Limitations of the works

Throughout the works, effort was taken to reduce limitations as far as possible and design representative experiments. However, key limitations remained. There is a lack of repetition across different pastures, seasons and with different animal groups. Additionally, within the N_2O section, there was a lack of soil sampling for measurement of wider soil characteristics occurring throughout the experimental period, to help explain the observed N2O emissions and aid in future modelling of emissions. Additionally, measurements were not taken for the recommended one-year period (IPCC, 2006), although emissions were measured for 6 months, which aligns with the recommended minimum period of 60-180 days (Vangeli et al., 2022). These limitations arose partly due to the inherent time-restrictive nature of a PhD studentship and the resources involved. However, in addition, university facilities including laboratories were closed for all but essential activities from March 2020 until August 2020 as a result of the COVID-19 pandemic. Due to grazing rights and weather conditions, some pastures were only available from April until October therefore this had a significant effect on ability to capture annual emissions variations. When experimental work resumed, it was subject to further time constraints and safety mitigations needing to be considered for example lack of out of hours access and Personal Protective Equipment (PPE) use. This reduced the time available to access the laboratories and increased planning time. Technical issues related to the GreenFeed unit as well as inclement weather causing damage to equipment resulted in further delays.

Throughout the duration of this project various technical problems occurred with different parts of the GreenFeed unit, including power supply, the air filter, the gas regulator, battery capacity, and internet connection. This limited the number of trials that could be completed across each pasture. Some of the issues were fixed over time with experience of use of the unit, therefore further development of using this system in more remote locations would allow further data collection and more confidence in results. It is recognised that additional data are required to make definitive conclusions regarding lamb production systems and the variation in GHG outputs across different pastures. Results from the hill experiment are considered to be less robust as the grazing area used was smaller in size than what animals would naturally roam. Effort was taken to decrease the number of animals involved in the trial due to this, however a minimum number of animals had to be used to generate enough measurements from the GreenFeed unit. It is recognised that the Ym values here are highly indicative and cannot be taken as absolute values, however they are included to show that there is a potential for differing emissions outputs from this type of pasture and this should be explored further.

The focus of the model development chapter was to explore how addition of disaggregation to the model would affect final farm footprints. While this was achieved, limitations do exist in the approach taken. A key limitation of the comparison between default methodologies and the site-specific modelling is the limited input data available for upland and hill areas in terms of both animal characteristics, forage characteristics and EFs. The site-specific information generated in this thesis was collected in one area, therefore may not be representative of wider upland and hill pastures. Additionally, factors that are known to affect final LCA results e.g., allocation methodology were not explored within this study (de Vries and de Boer, 2010). Combining disaggregation with improvements to allocation of impacts would further benefit accuracy of lamb production footprints. Efforts to increase transparency of this limitation were summarised in the sensitivity analysis section of the thesis (Chapter 5).

As with the model development chapter, a key limitation of assessing mitigation options is related to the lack of representative data available for these systems. There is limited information available for site-specific EFs, pasture characteristics (forage digestibility and crude protein content) and animal characteristics (animal weights and growth rates) making site-specific conclusions more uncertain. Assumptions were therefore made on links between mitigation options, input factors and the related effect on the model and resulting CF. An additional limitation is that the mitigation scenarios were completed using only one case study farm. However, given Henfaes exploits pastures at all three production altitudes within its system (lowland, upland, and hill), it is representative of many UK lamb production systems and focussing on one farm prevents deviation in any bias of on-farm data collection as all input data refers to the same farm, collected by the same staff.

The assessment of afforestation and its potential for C sequestration is clearly limited and should only be taken as an indicative first estimate. A default value of C sequestration per ha of introduced forestry was used, but this is a global estimate and will not fully account for local factors that may affect the C balance within the soil (Searchinger et al., 2018). It is necessary to consider the type of land present and its suitability for forest conversion before drawing more concrete conclusions. Additionally, the approach used to estimate the land saved from improving the upland has limitations as it relies on an accurate estimate of the number of animals in the upland to derive the grass use efficiency. This number has associated uncertainty as it is based on staff estimations rather than detailed stock takings.

A key area of uncertainty is that full assessment of wider secondary effects (both co — benefits and disbenefits) were not considered when applying mitigation scenarios, for example increasing the number of lambs per ewe could occur by increasing inputs to improve ewe health. This decision was taken due to the lack of empirical data that quantitively defines the relationship between desired outcomes and input parameters. Instead, focus was on showcasing how lamb production CF could be improved and the underlying effect that altitudinal differences could have when considering mitigation options. Broadening the model to account for other environmental burdens, i.e., taking a more holistic approach using product environmental footprinting to determine wider system impacts (Famiglietti et al., 2019) or farm-scale modelling that can better account for interacting factors within a system (including genetic factors, farm management and site-specific factors) (del Prado et al., 2010; Del Prado et al., 2011) would be useful to identify and quantify potential co-benefits and unintended consequences. This is particularly important as agriculture also impacts on water and air quality, and some GHG mitigation measures may act on other environmental burdens.

7.3 Recommendations for future study

A key recommendation for future research that has been made clear throughout all chapters of this work, is further data generation for site-specific model inputs. This includes EFs, digestibility data and collection of different animal characteristics at a farm level. Due to the increased effect of digestibility on the CF in comparison to EFs, a clearer understanding of how sward composition varies throughout year in contrasting pastures (lowland, upland, and hill) would improve footprinting to a higher degree. Further field experiments similar to those conducted here across different UK pastures using a wider variety of breeds would allow greater robustness in generating site-specific EFs across the year. This would include further urine and dung excretal N_2O emissions quantification in upland and hill pastures to increase robustness of EFs. Additionally further work to measure ruminant CH_4 at different times in the grazing season would aid in understanding variation. Seasonal differences could then be identified, allowing them to be used in country specific GHG inventory calculations and CFs. This could then also improve assessment of mitigation scenarios.

Further work on modelling would also provide greater understanding of the system environmental impacts. Improving model structure to better capture improved data inputs and further allow the disaggregation that was introduced in this study would be beneficial. There is potential to utilise programming languages to facilitate integration and analysis of large amounts of data, to improve both model functioning and parameterisation. As mentioned, reassessment of calculations and use of different model simulations may aid in improving CH_4 and N_2O estimations, although further data is required to be measured in the field to inform said models (for example, use of predictive modelling instead of Ym equations). A predictive model for UK lamb production systems across the different altitudes is not currently available. A further step to build upon this work, would be to begin assessment of wider environmental impacts, including water quality, soil quality and ammonia emissions, and not focus only on GHG emissions.

The mitigation options discussed here would be well supported by an additional economic analysis, allowing farm-level mitigation options to be selected that are effective in terms of GHG reduction but also for farmers costs. A multiple-pollutant MACC approach (Eory et al., 2013) that links GHG mitigation options and their cost to wider environmental impacts would allow the trade-offs and interactions between impacts to be assessed. Field experiments to better understand C sequestration changes when converting UK and Welsh land to forestry, particularly for the upland areas, will aid in understanding it's potential for future farm sustainability and policy options. This data could then be used within the mitigation scenarios analysis to provide C sequestration estimations with greater robustness. Additionally, the work could be supplemented by understanding how different mitigation options will be achieved and if there are any emissions trade-offs at a farm level as a result.

7.4 Wider implications of the findings

This research has the potential for wide-reaching implications. At a high-level, agriculture reaching Net Zero requires a thorough understanding of emissions outputs and the abatement required for residual emissions. This research aids in further understanding this for the lamb production sector, across the different pastures utilised within UK and Welsh lamb production. The model created within this work allowed disaggregation of model inputs and to determine if this approach differed from default methodologies. Overall, the results supported disaggregation of modelling and model inputs; a conclusion that can be taken forward into IPCC guidelines and improved inventory reporting. The model can also be used to recommend farm management strategies and to optimise animal management across lowland, upland and hill pastures for decreased CF. From a research perspective, it has highlighted where further research is required and the priorities for improving LCA of lamb production systems.

Within the policy sphere, particularly in Wales, it is important to understand how land may be used in the future. A clearer understanding of emission burdens in extensive pastures allows targeted mitigation options to be considered that will provide greater emission reductions than a "one-size fits all" approach. This can then aid understanding with regards to optimal land use in a future world of land conflicts, particularly in the context of sheep being used as "conservation grazers" to encourage biodiversity within current environmental stewardship schemes (DEFRA, 2022; Welsh Government, 2022). Afforestation targets are promoted for both the contribution of trees in provision of greenhouse gas removals as well as wider environmental implications such as biodiversity. This research demonstrates the potential for different environmental aims to work in tandem with the necessity of food production and offer farmers a way to offset emissions while keeping production the same. While changes must be made across the lamb production sector to reduce its environmental impact, it is also essential to prevent local economic harm and relocation of production away from the UK. This research and the data collected can be used alongside other literature to inform Welsh agricultural policy of Welsh-specific impacts, enabling better informed and more sustainable choices to be made.

7.5 Reference list

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Appendices

Appendix 3.1 – Supplementary material related to Chapter 3

After the hill experiment had been completed, the GF manufacturers estimated a potential 10% underestimation of CH_4 output and provided two workbooks: one removing potential outliers and one including all data points. The 10% discrepancy arose due to animals removing their heads frequently from the unit while measurements were ongoing. While head proximity is not used to calculate CH_4 output, it is used as a filtering criterion to remove visits with lower head proximity as outliers. This is due to the potential for CH_4 output to not be accurately measured as a result. The initial recommended workbook increased the head proximity threshold to include more measurements. An additional workbook was then supplied that reduced the head proximity but therefore also reduced the available measurements. Choice was left with the researcher as to which workbook to select. The analysis preceding this section used the initial supplied workbook as this was the recommended workbook by the manufacturers. However, the data in both workbooks is presented here to identify if there are any significant differences (Table A).

Table A Differences between the initial supplied workbook and reduced head proximity threshold workbook in terms of number of measurements, range, standard deviation, mean and CV%

Measured values		
Year	Initial Workbook	Reduced Head Proximity Threshold
Number of measurements	364	202
Range (g day^{-1})	1.06-32.34	1.07-33.65
Standard deviation	3.54	4.02
Mean (g day^{-1})	7.9	8
CV%	44.9	50.1

There was no significant difference determined between the two datasets (p=0.81), therefore it was concluded that completing analysis using the initial workbook was justified (and allowed use of an increased number of measurements).