Bangor University

DOCTOR OF PHILOSOPHY

The environmental cost-benefits of improving pasture productivity on upland cattle systems

Williams, Non

Award date:
2020

Awarding institution:
Bangor University

DOCTOR OF PHILOSOPHY

The environmental cost-benefits of improving pasture productivity on upland cattle systems

Williams, Non

Award date:
2020

The environmental cost-benefits of improving pasture productivity on upland cattle systems

Non Gwenllian Williams

School of Natural Sciences

Prifysgol Bangor University

A thesis submitted for the degree of Doctor of Philosophy

2020

Title	The environmental cost-benefits of improving pasture productivity on upland cattle systems
Principal supervisor	Doctor A. Prysor Williams School of Natural Sciences, Bangor University
Co-supervisor	Doctor James M. Gibbons School of Natural Sciences, Bangor University
Assessment Committee	Doctor Eifiona Thomas-Lane (chair) School of Natural Sciences, Bangor University Professor Davy I. McCracken (external examiner) Scotland's Rural College Doctor Philip A. Hollington (internal examiner) School of Natural Sciences, Bangor University
Financed by	The research was funded by the KESS 2 scheme in collaboration with Hybu Cig Cymru
Date of registration	01 October 2016
Date of first submission	10 July 2020
Date of final submission	20 November 2020

Executive Summary

A large proportion of the United Kingdom's agricultural land is classified as uplands. The majority of this land is used for livestock production, but pasture productivity is often low. This, together with market forces and changes in support payments for agriculture, mean that cattle numbers in the uplands are declining. A continuation of this trend could compromise beef production and lead to other undesirable impacts such as the spread of invasive species. The red meat sector is also under considerable pressure to reduce its environmental burden. This may bring additional pressures on upland livestock systems.

Optimised management of pasture is fundamentally important in maintaining grass quality and quantity, providing sufficient nutrition to ruminants. Improving pasture productivity and grass utilisation could provide numerous benefits such as increasing the length of the grazing season, thereby reducing supplementary feed requirements and decreasing greenhouse gas emissions from agriculture. While many studies have evaluated the effectiveness of pasture improvement methods on increasing pasture productivity, few have focussed on the environmental outcomes in relation to the uplands. An improved understanding of this is important for identifying opportunities to increase production efficiencies on-farm as well as reduce greenhouse gas emissions from upland cattle systems.

The overall aim of this thesis was to determine the effect of alternative pasture and grazing management options in upland cattle systems on production efficiencies, and their respective environmental trade-offs. Firstly, a literature review (Chapter 2) was conducted whereby I discussed existing studies, as well as identified knowledge gaps in the research. Chapter 3, the first experimental chapter, assessed the current and potential pasture productivity on a typical upland farm in north Wales over a three-year period in order to investigate the environmental cost of various options to increase pasture productivity in the uplands. Pasture productivity was significantly higher following reseeding with a grass variety for marginal land than from current upland permanent pasture. Furthermore, higher nutrient (lime and fertiliser) application in accordance with the Nutrient Management Guide (RB209) resulted in an increase in pasture production. Field operations i.e. reseeding and nutrient application led to increased nitrogen loss via nitrous oxide emissions from the soil. This was particularly evident from rotovated land.

In Chapter 4, the efficiency of upland grazing systems for cattle was evaluated by investigating cattle liveweight gain when grazing improved (pasture that received lime and fertiliser application) and unimproved uplands, and nitrous oxide emissions produced from the soil as a trade-off. This was assessed by conducting a field experiment, once again on a typical upland farm in north Wales. In contrast to existing literature, cattle liveweight gain did not differ between grazing treatments, possibly due to not utilising the improved pasture to its full potential. Urine excretion led to significantly higher nitrous oxide emissions from the soil than dung excretion and fertiliser application. In Chapter 5, the findings from previous experimental chapters as well as data from other sources were used to investigate the potential of upland beef production and pasture productivity at the regional level. Furthermore, land use competition for livestock production and afforestation in the future was examined. Many upland areas in Wales identified as potential sites for intensifying beef production overlapped with areas suitable for afforestation, indicating a challenge to accomplish both in the future.

To summarise, this work shows that increasing the efficiency of upland beef production via improved pasture management and sustainable intensification is achievable for upland cattle systems in the future. However, the environmental impact of such mechanisms should be carefully considered to work towards reducing greenhouse gas emissions from agriculture. Further work is required to quantify unmeasured environmental burdens from upland cattle systems.

Acknowledgements

I would like to acknowledge the following:

- My supervisors: Dr. Prysor Williams and Dr. James Gibbons of Bangor University. Their expertise, time, guidance and support has been invaluable. Without this, it would have been an impossible task to produce the work included within the thesis.
- Knowledge Economy Skills Scholarships (KESS 2) for funding this PhD project. Knowledge Economy Skills Scholarships (KESS 2) is a pan-Wales higher level skills initiative led by Bangor University on behalf of the HE sector in Wales. It is part funded by the Welsh Government's European Social Fund (ESF) convergence programme for West Wales and the Valleys.
- Hybu Cig Cymru for their financial support and the opportunity to collaborate on this project. I hope that they find the outcomes relevant and useful. Also, Gwawr Parry and Nia Davies for their assistance as company supervisors.
- Prof. Dave Chadwick and Dr. Kara Marsden for their expertise.
- Ian Harris for his guidance, advice and support throughout the project.
- Llinos Hughes, Mark Hughes and Bleddyn Jones at Henfaes Research Centre for their guidance, assistance and patience with the field work.
- Emily Cooledge and Sarah Chesworth for their general technical support.
- NERC - Centre for Ecology and Hydrology for providing meteorological data for the field work.
- My parents, Lois, Jack and friends for their encouragement, patience and support.
- Dr. Amy 'Spike' Lewis and Tara Garraty for their guidance and support prior to the Viva.
- Finally, my colleagues within the School of Natural Sciences for their interest and companionship.

Table of Contents

List of Figures

List of Tables

Authorship

All data chapters (Chapter 3-5) in this thesis have been prepared as manuscripts for future journal publication. The references can be found at the end of each chapter, and the supplementary material in the Appendix at the end of the thesis.

Chapter 3: Williams, N.G., Gibbons, J.M., Marsden, K.A., Chadwick, D.R., Williams, A.P. (Manuscript submitted for publication in Agriculture, Ecosystems and Environment). The environmental cost of increasing upland pasture productivity.

1. Introduction

1.1 Research rationale

The global population is projected to exceed 9 billion by 2050 (United Nations, 2019). This population growth, along with changes in diet and food consumption patterns requires an increase in food production in the future. However, a rise in urbanisation and greenhouse gas (GHG) emissions produced via land use change by agriculture pose a challenge to addressing this (United Nations Environment Programme, 2014). Food production methods must therefore expand sustainably in order to adapt to climate change (FAO, 2009).

Agriculture is the dominant land use in Wales' uplands (National Assembly for Wales, 2016), with the majority of this land managed for livestock production. Historically, this sector has been largely driven by changes in agricultural policy. Many upland livestock farming systems have been very dependent on agricultural subsidies, notably direct payments via 'Pillar 1' of the Common Agricultural Policy (CAP) to remain economically viable in the past (Hardaker, 2018). In recent years, there has been a shift in payments from Pillar 1 to Pillar 2 of the CAP, i.e., more funding for agri-environment schemes, with a greater emphasis on the delivery of environmental benefits. Whilst we know that a large proportion of these upland livestock systems are managed as extensive sheep systems (Fraser, 2008), many upland farms also keep cattle for beef production and/or to sell offspring as 'stores' for finishing elsewhere, typically in lowland areas. However, the economic viability of cattle in the uplands has been under challenge for many years, primarily due to market returns not increasing in line with costs (Mansfield, 2011). This has led to a reduction in cattle numbers in the uplands; which is at odds with some agri-environment schemes that incentivise cattle grazing as it can help enhance biodiversity (Critchley et al., 2008).

It is widely recognised that agriculture, and the livestock sector in particular, is a large contributor to GHG emissions. With ambitious targets to reduce GHG production in order to tackle climate change, the agricultural sector must also address this. Agriculture accounts for a considerable proportion of nitrous oxide (N_2O) emissions, a highly potent GHG, mainly by means of livestock excreta on pastures and the use of synthetic fertilisers. Hence, there are calls to reduce N_2O emissions produced from pasture-based livestock systems and the quantification of emissions has received recent research attention (Chadwick et al., 2018; McAuliffe et al., 2018; 2020). However, these emission estimates were based on lowland grazed pastures. As a significant proportion of Wales' land area is classified as uplands, there is a need to quantify N_2O emissions from upland pasture-based cattle systems, both via excretion and fertiliser application.

With increasing debate about future land use in Wales' uplands following withdrawal from the European Union (Hubbard et al., 2018), along with net-zero GHG emission targets for Wales by 2050, there is great uncertainty about the direction of upland livestock production in the future. A better understanding of the pasture management options and environmental impacts of upland cattle production systems is fundamental in identifying opportunities to increase production efficiencies on-farm, thereby improving profitability of cattle systems, whilst simultaneously reducing GHG emissions without compromising production.

1.2 Aims and objectives

The overarching aim of the work contained in this thesis is to determine the effect of alternative pasture and grazing management options in upland cattle systems on production efficiencies, and their respective environmental trade-offs.

The objectives of the research are as follows:

- To report the findings of existing research on upland cattle systems (Chapter 2)
- To investigate pasture productivity and quality in the uplands under different management options, along with the associated environmental impact (Chapter 3)
- To assess cattle liveweight gain from upland pasture, and to investigate the environmental impact of cattle grazing systems (Chapter 4)
- To examine upland land use in the future and competition for land between agriculture and afforestation (Chapter 5)
- To identify opportunities for further work within the research area (Chapter 6)

1.3 Thesis synopsis

Chapter 2 is a review of existing literature on upland cattle systems. This chapter analyses what is already known about upland beef systems in the UK, and their economic and environmental significance. Upland land use is discussed, as well as the management options for increasing the productivity of upland grazed pasture, and consequently beef production. Such options that are examined include varying the grazing system implemented, reseeding, fertiliser use, cattle breed effect and the inclusion of supplementary feeding in upland beef systems. Finally, the future of cattle in the uplands is discussed.

Chapter 3 is an experimental chapter, where the current and potential pasture productivity in the uplands is examined. Different pasture management scenarios were explored at Bangor University's research farm, namely ploughing and rotovating, followed by grass reseed, as well as targeted lime and fertiliser application. The results of this randomised block design simulated grazing experiment on a typical upland farm were used to assess the productivity of unimproved upland pasture, along with the effect of field operations on grass growth, quality and N_2O emissions released from the soil.

Chapter 4 details the findings of a second field experiment on a typical upland farm that investigated the effect of pasture management on cattle liveweight gain. This experiment measured cattle liveweight gain when grazing improved (targeted fertiliser and lime application) and unimproved upland pasture. The environmental impact (soil N_2O emissions) was also quantified, with the results used to produce an environmental footprint of each grazing treatment by means of a life cycle assessment model.

The last data chapter, Chapter 5, aims to assess the potential degree of land use in competition for agriculture and forestry in Wales' uplands in the future. The results from Chapters 3 and 4, along with matching variables to the experimental sites in the previous two chapters were used to identify potential areas for the expansion of cattle production in Welsh uplands in the future. Spatial analyses were performed to determine locations for potential new woodland creation in the future. Intensifying agricultural production in some regions in order to convert other agricultural land areas to new woodland, as well as the sustainable intensification concept, are discussed.

Finally, Chapter 6 consists of a general discussion whereby the key findings of the each chapter are summarised, as well as the strengths and limitations of the research methods. Furthermore, gaps for further work in the research area are identified, along with the policy implications of the research.

1.4 References

Chadwick, D.R., Cardenas, L.M., Dhanoa, M.S., Donovan, N., Misselbrook, T., Williams, J.R., Thorman, R.E., McGeough, K.L., Watson, C.J., Bell, M., Anthony, S.G., Rees, R.M., 2018. The contribution of cattle urine and dung to nitrous oxide emissions: Quantification of country specific emission factors and implications for national inventories. *Science of the Total Environment 635,* 607-617.

Critchley, C.N.R., Adamson, H.F., McLean, B.M.L., Davies, O.D., 2008. Vegetation dynamics and livestock performance in system-scale studies of sheep and cattle grazing on degraded upland wet heath. *Agriculture, Ecosystems and Environment 128*, 59–67.

FAO, 2009. Global Agriculture Towards 2050. *FAO, Rome.* [Online]. Available at: http://www.fao.org/fileadmin/templates/wsfs/docs/Issues_papers/HLEF2050_Global_Agriculture.pdf. Accessed June 22nd, 2020.

Fraser, M., 2008. Managing Change in the Uplands. [Online]. Available at: https://www.aber.ac.uk/en/media/departmental/ibers/pdf/ibersinnovationsen08/chap_2.pdf. Accessed June 18th, 2020.

Hardaker, A., 2018. Is forestry really more profitable than upland farming? A historic and present day farm level economic comparison of upland sheep farming and forestry in the UK. *Land Use Policy 71*, 98-120.

Hubbard, C., Davis, J., Feng, S., Harvey, D., Liddon, A., Moxey, A., Ojo, M., Patton, M., Philippidis, G., Scott, C., Shrestha, S., Wallace, M., 2018. Brexit: how will UK agriculture fare? *EuroChoices 17*, 19–26.

Mansfield, L., 2011. Upland Agriculture and the Environment. Badger Press, Bowness on Windermere.

McAuliffe, G.A., López-Aizpún, M., Blackwell, M.S.A., Castellano-Hinojosa, A., Darch, T., Evans, J., Horrocks, C., Le Cocq, K., Takahashi, T., Harris, P., Lee, M.R.F., 2020. Elucidating three-way interactions between soil, pasture and animals that regulate nitrous oxide emissions from temperate grazing systems. *Agriculture, Ecosystems and Environment 300*, 106978.

McAuliffe, G.A., Takahashi, T., Orr, R.J., Harris, P., Lee, M.R.F., 2018. Distributions of emissions intensity for individual beef cattle reared on pasture-based production systems. *Journal of Cleaner Production 171*, 1672-1680.

National Assembly for Wales, 2016. The Farming Sector in Wales. [Online]. Available at: https://senedd.wales/research%20documents/16-053-farming-sector-in-wales/16-053-web-english2.pdf. Accessed June 18th, 2020.

United Nations, 2019. World Population Prospects 2019 Highlights. [Online]. Available at: https://population.un.org/wpp/Publications/Files/WPP2019_Highlights.pdf. Accessed June 22nd, 2020.

United Nations Environment Programme, 2014. Assessing Global Land Use: Balancing Consumption with Sustainable Supply. A Report of the Working Group on Land and Soils of the International Resource Panel. Bringezu, S., Schütz, H., Pengue, W., O´Brien, M., Garcia, F., Sims, R., Howarth, R., Kauppi, L., Swilling, M., Herrick, J.

2. An analysis of upland beef systems in the UK: their environmental and economic significance – a review of the literature

2.1 Introduction

The agricultural sector is faced with the dual challenge of feeding a rapidly increasing population whilst reducing its environmental impact, as the projected effect of climate change is likely to become increasingly severe in future decades (Nelson et al., 2010). A significant proportion of global anthropogenic greenhouse gas (GHG) emissions derive from livestock production systems; particularly via enteric methane (CH_4) emissions from ruminants, and nitrous oxide (N_2O) emissions from grasslands due to fertiliser and urine deposition (Cardenas et al., 2019; Johnson et al., 2007). Cattle are responsible for an estimated 65% of the GHG emissions produced from meat production globally (Gerber et al., 2013). In the UK, approximately a third of the agricultural land area is classified as 'uplands' and is primarily managed for the production of sheep and cattle (Reed et al., 2009a), indicating the significance of the uplands as a source of food production. Despite this, cattle numbers in the uplands are declining due to many factors, including changes in policy and economic pressures. In addition, the uplands deliver benefits apart from food production, including the provision of multiple ecosystem services and public benefits, from local to national level (Godfray and Garnett, 2014). However, some argue that the livestock production systems seen in the uplands is to the detriment of the provision of these services (Bullock and Kay, 1997; Evans et al., 2015).

Optimal management of pasture is fundamentally important to maintain grass quality and quantity, providing sufficient nutrition to ruminants. Livestock grazing habits and the use of cattle as 'conservation grazers' are well studied (Limb et al., 2011; Rook et al., 2004). However, grass is often undervalued as a resource for ruminant production, with anecdotal evidence suggesting that as much as 50% of grass is wasted due to poor management (Hunt, 2011). Strategies to increase grassland use efficiency include altering grazing systems and more targeted fertiliser applications. These could provide numerous benefits, including an extended grazing system, and reductions in supplementary feeding and GHG emissions.

The following review is based on UK uplands and outlines the main historic land uses and management that have led to their current state, as well as the way that cattle have influenced these areas in terms of grassland management and biodiversity; with Wales' uplands examined as a case study. Whilst most studies to date have focussed on mixed beef and sheep systems, this review will primarily consider grazing solely by cattle. Finally, the potential use of grazing cattle to implement sustainable intensification by enhancing both food production and supply of ecosystem services in the uplands is reviewed.

2.2 Upland land use and management

2.2.1 Defining the uplands

The definition of 'uplands' varies significantly, depending on the landscape and its properties. Features include climate, geology, topography, soils, vegetation and land use, which interact collectively and individually to shape the uplands. Each upland district is unique to its region and surroundings (Usher and Thompson, 1988). Table 2.1 below summarises the competing definition used to describe the uplands. Typically, they are defined by their physical or ecological parameters, often using elevation, land use management, soils or vegetation criteria (Reed et al., 2009b; Morgan-Davies et al., 2006).

Table 2.1. Varying definitions of the uplands, in chronological order.

Definition	Source
'…hill land as that part of the country which stands above the 700 ft [213 m] contour. …in this country (UK), however, farming practices, soil conditions, vegetation and the habitations of man all begin to show differentiating characteristics'	Stapledon (1937)
'…Between this altitude (600/800 ft) [183/244 m] and 1400/1500 ft [427/457 m], where arable cropping and intensive livestock management become uneconomic, are transitional zones known as 'uplands'	Committee on Hill Sheep Farming in England and Wales, De La Warr (1944)
'The land classes which have a combination of high rainfall, low evapotranspiration, low insolation and generally poor soils, provide the basic upland area'	Merlewood land classification system, Bunce et al. (1983)
'Land predominantly above 240 metres (800 feet)…The 240 metre contour corresponded closely to the boundaries of the Less Favoured Areas (LFA) of England and Wales as defined and approved under EC Directive 75/268, prior to the recent application to include marginal land. Consequently we have regarded the uplands as being synonymous with the LFA...'	Countryside Commission for England and Wales (1984)
'Land associated with the upland meat regime and farming systems, both beef and sheep, i.e. moorland, allotment and in-bye land…Overall the best	English Nature (1996)

definition of upland for targeting policy is described by the administrative boundary of Less Favoured Areas (LFAs)’	
‘All land which lies above or beyond enclosed farmland or croftland, and which is covered by vegetation which is at least semi-natural in character’	Ratcliffe and Thompson (1988), Scottish Natural Heritage (2002)
‘Less distinct (than mountains) elevated landscape with less extreme environments’	Smithson et al. (2002)
‘There is no statutory definition of upland or lowland Britain. However, there are statutory maps of Less Favoured Areas for the United Kingdom, prepared under Council Directive 75/268/eec. These maps, which are predominantly upland, also include other farming areas which suffer disadvantage and are used for the purpose of targeting certain CAP support payments’	Whitty, UK Government (2003)
‘Land > 250 m above sea level; in northern Scotland upland can be almost down to sea level though’	UK National Ecosystem Assessment (2011)

Publications about the uplands often do not use an explicit definition, but rather describe the typical environmental and geographical attributes that constrain production in such areas. Defra, similarly to the Countryside Commission mentioned in the table above, proposed the LFA boundaries as a definition of the uplands (Great Britain Parliament, 2011). Consequently, the definition proposed by the Countryside Commission in Table 2.1 will be the working definition accepted for the thesis.

UK uplands are set within a variety of landscapes and provide an abundance of valuable ecosystems and public benefits. LFA constraints occur in various forms e.g. harsh climates, extreme rainfall patterns, shorter growing seasons, steep slopes and high altitudes (Reed et al., 2009b). Along with anthropogenic influence, a series of climatic changes over the last two million years have been responsible for the current condition of UK uplands. Over the last few centuries, livestock and fodder production (e.g. grass, hay, silage and root crops) are the main land uses in UK uplands (Mansfield, 2011). Despite this, pasture-based livestock production systems in the uplands are facing increasing competition from other land uses and diversification options such as cropland, forestry and land for wind turbines and solar arrays (Donnison and Fraser, 2016).

2.2.2 Drivers of change in the uplands: past and present farming systems

Numerous factors have, and continue, to shape and influence the upland agricultural landscape; the main factors being political changes and economic pressures, such as from upland cattle systems' perspective, the costs of keeping cattle relative to the returns. These factors contribute to the ability of upland livestock systems to remain financially viable and are mainly responsible for the changes in cattle numbers in the uplands over time (Reed et al., 2009b).

Livestock systems have played an important role in upland management for many centuries. Following on from the Bronze Age, field systems were adopted, leading to intensification and the uplands being utilised for beef and sheep production. The introduction of cultivation to the land led to cattle and sheep production being the principal upland use from the 11th century AD onwards (Mansfield, 2011). Developing practices such as liming, drainage and selective breeding from the 11th to the 14th century improved productivity sufficiently to make farms viable. The Black Death outbreak in the 14th century was responsible for the general depopulation and abandonment of the uplands and the extensification of production. Despite this, of the land that wasn't abandoned, the uplands remained to be predominantly used for livestock production – cattle, dairy and pigs, with farms mainly operated as landlord and tenant businesses rather than subsistence systems (Mansfield, 2011).

The Industrial Revolution of 1700-1850 influenced both the upland landscape and meat consumption, in that it was a period of advancements and increased livestock production, lamb in particular. Despite this, many perceived cattle as the fundamental livestock during this period as they produced a higher proportion of the nation's food supply in the form of meat and milk. Consequently, this adversely affected sheep numbers (Garner, 1944). However, this phase of development was rapidly followed by the Agricultural Depression in the 1880s and 1890s, with a dramatic reduction in cattle, sheep, wool, and land prices. Some would claim that upland farmers were at a disadvantage during this period due to the harsh physical conditions experienced, along with lack of easy access to markets (Perry, 1974). Increased demand for products during the First World War led to a renewed emphasis on livestock production in the uplands. Wool continued to increase in value and limited labour and traditional practices meant that sheep grazing was at its most intense in certain regions. A number of schemes were instigated post-war in an attempt to create employment and maintain the livestock population in such areas, whilst aiming to increase national self-sufficiency of livestock foods (Collins, 1978).

The Second World War proved to be a critical factor in driving change in agricultural practices in the uplands. Agricultural land use shifted towards crop production at the expense of grassland normally used for livestock grazing. This led to a halt in cattle numbers, despite subsidies to alleviate loss of animals during the war (Bowers, 1985). This continued until the early 1950s due to food shortages, which compromised beef production. Following the war, marginal land was used for livestock grazing, increasing the total area designated for upland farming systems, therefore fuelling the development of upland agriculture.

Regardless of the fact that guaranteed prices for sheep were removed in 1959, the financial encouragement for beef production was increased. Higher headage payments under the Hill Livestock Compensatory Allowances (HLCA) in 1965 meant that farmers were paid a subsidy per head of stock; regardless of reduced market prices and herd size (up to a maximum of 0.9 LSU ha^{-1}) during this period (Mansfield, 2011; Mowle and Bell, 1988). This proved to be a significant incentive for farmers to maintain beef cattle in the uplands, with the HLCA subsidy forming over 17% of upland farm support between 1973 and 1984 (Winter et al., 1998). This, along with the Beef Annual Premium led to a growth in numbers (Bowers, 1985). Sheep systems receiving payments in the form of the Annual Ewe Premium and the Sheepmeat Variable Premium following HLCA outweighed the support provided to beef farmers, resulting in a rapid increase in sheep numbers and a notable decline in cattle.

The Common Agricultural Policy (CAP) underwent a major reform in 2003, largely driven by World Trade Organisation pressures (Swinbank and Daugbjerg, 2006). Subsidies were decoupled from agricultural production; with a greater emphasis on the rural development measures, known as 'Pillar 2' of the CAP. Headage payments were removed and agri-environment schemes introduced, many of which had upper limits on stocking densities and restrictions on grazing periods. Such factors, together with the development and alterations to agri-environment schemes since 2003, have had a notable influence on upland livestock production in the UK.

In more recent times, society's perception and understanding of the uplands has led to greater calls to protect and enhance the value of the uplands for contrasting uses e.g. biodiversity, recreation, carbon storage and water supply. Some claim that new and evolving policies such as the EU Water Framework Directive and the Kyoto Protocol (requiring the UK to reduce GHG emissions to 12.5% below 1990 levels by 2012) led to changes in farming practices in order to meet the required conditions (Reed et al., 2009a). Although one could debate how many landowners changed their practices directly due to such policies, they may have been precursors to more stringent and ambitious environmental targets recently introduced. For instance, the designations of some upland as Nitrate Vulnerable Zones through the EU Nitrates Directive mean that there are binding requirements on nitrogen fertiliser use and the volume and management of cattle slurries (Defra, 2018). In terms of GHG emissions, forward targets set by the UK Government include reducing GHG emissions by 57% from 1990 to 2030, along with an amended target of 100% reduction in emissions by 2050, compared to 1990 levels (Committee on Climate Change, 2019). Such targets will almost certainly have implications for cattle systems (Rojas-Downing et al., 2017).

As seen in Figure 2.1 below, different policies have resulted in variations in cattle numbers, particularly in the uplands.

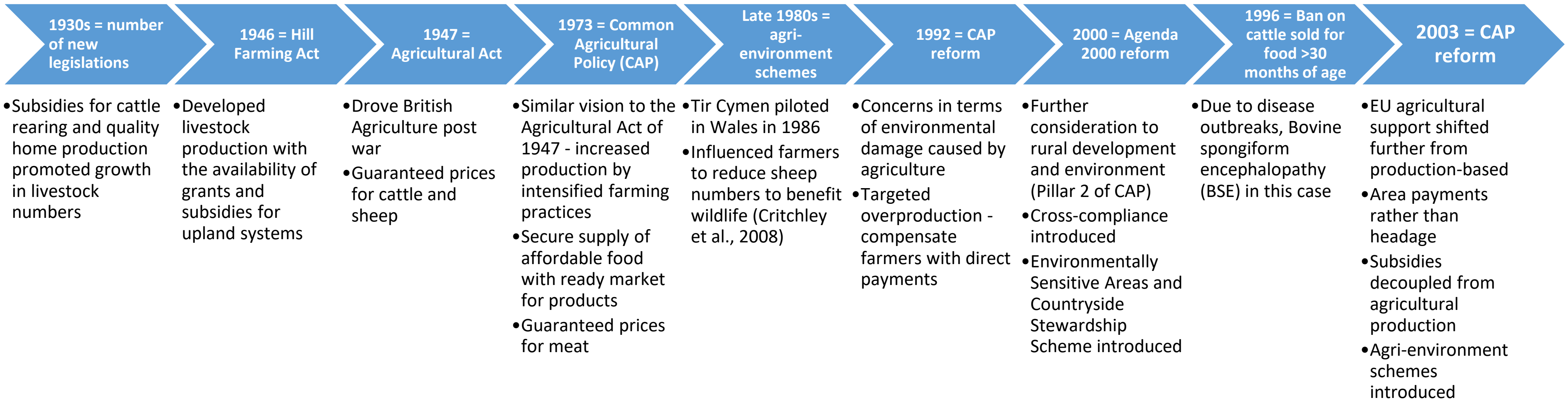

Figure 2.1. Timeline denoting the main UK agricultural policies from 1930-2003 inducing change in upland livestock systems.

2.2.3 Cattle production in the uplands: influencing factors

2.2.3.1 Consumer demand, behaviour and preferences

As previously mentioned, with the global population estimated to reach 9 billion by 2050 (FAO, 2009), there is increased pressure on the agricultural sector to safeguard food security. Figure 2.2 below highlights the main trends seen in consumption patterns dating back to 1700. Supply and demand is undoubtedly a prominent influence on cattle production in UK uplands and will continue to be so due to the rapidly increasing global population.

1700 - 1850 = increased consumption

- Period of Agricultural Revolution - improvement
- Rise in living standards increased urban demand for meat and dairy products

1860 - 1918 = consumption increased dramatically

- Increase in consumption despite period of Agricultural Depression from 1875
- General rise in UK population standard of living
- Developments in global transports increased imports
- Therefore increased consumption of imported rather than home-produced meat

1918 - 1939 = increased demand for products, but shift in beef demand

- Following the First World War, shift in demand towards small, mean and tender joints of beef, containing a small proportion of bone
- 1938 = 45% meat consumed in the UK was beef and veal; 1/3 UK citizen's weekly food expenditure spent on cattle products (Garner, 1944)
- Despite this, meat prices proceeded to decline (Mansfield, 2011)

1939 - 1950 = steady meat consumption

- Following the Second World War, rationing of meat products introduced to ensure consistent availability in period of austerity - restricted beef market

1950 - 2000 = increased meat consumption

- More urbanised and wealthier population e.g. in Asia and Latin America
- Wealthier households more likely to purchase higher volume of beef (Berry and Hastry, 1982; Mathijs, 2015)

2000 onwards = shift in consumption patterns

- Change in consumer demand, particularly in developed countries e.g. United States (Resurreccion, 2004)
- Belief that white meat was healthier (Webster, 2011)
- Pork the most consumed red meat in developed countries as a whole (FAO, 2009)
- However, beef remains most popular meat type in certain developed countries e.g. Australia and Canada (Daniel et al., 2011; Baghurst et al., 2000)
- Dietary changes - shift towards more meat-free diets in some regions in recent years, with the vegan movement developing

Figure 2.2. Timeline of beef meat consumption patterns which has driven change in upland livestock systems over time.

Studies have shown that grass-fed cattle produce different nutritional quality meat to cattle fed on concentrates, have a stronger flavour and possess a healthier fatty acid profile (Daley et al., 2010; Scollan et al., 2006). On selection, some studies have indicated that these attributes are favoured over dark cutting meat (Acebrón and Dopico, 2000). Most upland cattle production systems implement an extensive, pasture-fed regime, usually followed by finishing on concentrates (often in lowland regions). Issues have also been raised regarding the welfare of concentrated cattle production systems, and research in the UK and further afield concluded that consumers are willing to pay for beef meat closely associated with animal welfare and environmentally-friendly production (Ashworth, 2000; Bignal et al., 1999; Kuit and van der Meulen, 1999; Sonoda et al., 2018). Advances in meat-labelling systems has led to wholly grass-fed beef being certified and identified with a certification mark at supermarkets. Beef products labelled in this form are believed and expected to be healthier, therefore often preferred by consumers over concentrate-based beef systems (Font-i-Furnols and Guerrero, 2014). However, upland beef production systems are often from suckler herds; and several studies have recorded that the environmental impacts of beef meat production are in fact greater from suckler herds than dairy-derived beef (de Vries et al., 2015; Nguyen et al., 2010; Soteriades et al., 2019). In turn, this may have implications on the demand from environmentally-aware consumers.

A reduction in the average price of beef meat is an indication of changes to supply and demand, but can also reflect the need to compete with other sectors such as cheaper white meat. Communication between producer and consumer is key in increasing transparency and satisfying demand. The majority of beef consumers base their buying preferences on meat eating quality, nutrition and method of production (Issanchou, 1996; Verbeke et al., 2010). Current research as part of the 'Beef Monitor' and 'BeefQ' projects looks at the use of 3D cameras to measure conformation, fat and yield of a live animal (BeefQ, 2019; International Committee for Animal Recording, 2015). Providing carcass data on such traits to upland beef producers could be useful in broadening their understanding of consumer demands.

Research dating back to the 1980s emphasised then that the older population is more inclined to purchase beef meat than the younger generation (Berry and Hastry, 1982). Despite changes in consumption patterns since this period, there has been a continuation of this trend over recent decades (Kantar Worldpanel, 2017). Kantar Worldpanel figures (2018) have indicated that 91% of British households purchase red meat. Despite this, a consumer insight study (AHDB, 2018a) revealed that over half of UK vegans are under 35 years of age. Measures intended to target this age category have been instigated, with a collaborative campaign between the red meat levy boards Hybu Cig Cymru, Agriculture and Horticulture Development Board, and Quality Meat Scotland appealing to a young audience aimed at highlighting the role of red meat in a healthy diet (AHDB, 2018b).

Furthermore, a review of the Beef Sector in Wales in 2014 highlighted a growth in demand for minced beef (SAC Consulting, 2014). This was as a result of a change in consumer preference due to value and convenience. Therefore, the sector is faced with the challenge of increasing beef consumption by the younger generation as well as building on consumer preferences to maintain the viability of beef production systems. In addition to this, evidence has shown that the environmental sustainability of livestock systems in particular influences consumption trends, posing a risk to demand in the days of sustainability-driven consumers (Henchion et al., 2014), as mentioned earlier. Recently, there has been an increased awareness of local food sources due to the population's understanding of food being transported over large distances (Garnett, 2011), which is supportive of upland beef systems.

2.2.3.2 Biodiversity effects

Agriculture is deemed as both a threat to biodiversity, and key to its survival. Conservation of biodiversity is now regarded as a priority across many levels of society around the world (Hodge et al., 2015; Zachos and Habel, 2011), with it being increasingly prominent with every CAP reform. Despite this, there is a significant decrease in biodiversity in agricultural landscapes within Europe (EEA, 2015; EEA, 2017), with agricultural intensification and abandonment being two major processes contributing towards this (Henle et al., 2008). Appropriate grazing by cattle can enhance species richness and reduce the spread of over-dominating, undesirable and invasive species in the uplands (Critchley et al., 2008). Their grazing and trampling habits create favourable conditions for the restoration of habitats that have become degraded (Mitchell et al., 2008). Often, the conditions of upland landscapes used as grazing land does not allow for reseeding the land. Therefore alternative means of encouraging native grass species is vital.

A large proportion of the UK upland habitats are listed as Biodiversity Action Plan (BAP) habitats, and support a number of upland bird species listed on the BAP, e.g. curlew *(Numenius arquata)*, chough *(Pyrrhocorax pyrrhocorax)* and golden plover *(Pluvialis apricaria)* (Chambers and Daniell, 2011; Johnstone et al., 2008). Many of these protected birds require grassland areas of varying sward height. Cattle as grazers will leave areas of tussocky vegetation which is of benefit to some small mammals and invertebrates (Donnison and Fraser, 2016). The uplands are important in supporting wildlife and agriculture in these areas, and can play a significant role in the management and restoration of these habitats.

Heather *(Calluna vulgaris)* is a BAP species in upland habitats and was traditionally deemed an important food source for livestock in the winter (Grant et al., 1976). Farm management practices, mainly over-grazing, coupled with atmospheric nitrogen deposition, have resulted in replacement by grasses in much of the UK uplands (Anderson and Yalden, 1981; Hartley and Mitchell, 2005; Welch, 1986). Despite this, cattle trampling at low intensity result in bare patches on the surface, which encourages the germination of *C. vulgaris* seeds (Critchley et al., 2008; Mitchell et al., 2008). Studies have shown that cattle trampling is more favourable than sheep trampling due to their higher body mass. It is also deemed that cattle grazing alone is more beneficial than mixed grazing regimes at low intensity for promoting *C. vulgaris* establishment through the promotion and increase of the length of the pioneer phase of its growth (Grant and Hunter, 1966; Mitchell et al., 2008). Consequently, this may promote the diversification of invertebrates in uplands. Grasses that replace *C. vulgaris* include purple moor-grass *(Molinia caerulea)* and matgrass *(Nardus stricta)*, largely dominant grass species that restrict sward heterogeneity (Fraser et al., 2011). Furthermore, other native, undesirable species such as bracken *(Pteridium aquilinum)* are responsible for decreasing upland species composition. Studies have shown that cattle grazing is more effective for the control of these species than sheep grazing (Critchley et al., 2008; Rook et al., 2004). Some regard the reason for the domination of species like *M. caerulea* as the imbalance in the sheep:cattle ratio, therefore increased cattle grazing could increase sward diversity (Fraser et al., 2011; Mowle and Bell, 1988).

Studies have shown that age and sex are additional factors that may affect the dietary choice of grazing livestock. This is due to dental and digestive system development. When seven-months old calves were compared to eighteen-months old heifers, it was found that they were more selective in their grazing habits when grazing cocksfoot swards (Ferrer-Cazcarra and Petit, 1995). Similarly, female cattle tended to be more selective than males. However, there is little data available regarding differences between beef cattle breeds. In addition to the direct effect of the contrasting grazing management routines mentioned above on sward composition, cattle grazing can provide a wide variety of secondary benefits, including altering the availability of breeding sites and supply of food plants for invertebrates (Rook et al., 2004). One must acknowledge the potential role of appropriate cattle grazing for enhancing biodiversity in the uplands in the future.

2.3 Management options for increasing productivity of upland grazing

Grass is the cheapest form of readily available feed for grazing livestock. Efficient grazing leads to better land use efficiency of livestock systems in terms of output per hectare (van Zanten et al., 2016). Numerous factors contribute to the productivity of the grass including grazing management, sward composition in terms of productive herbage species and weeds, fertiliser application and soil status (Fuller, 1988). However, measures that improve production (e.g. agrochemical inputs) are often not possible in the uplands due to their physical features. Therefore agricultural output can only be altered through manipulating stocking rate, stock type or timing of stocking (Fraser et al., 2014a).

When effectively managed, grass can provide sufficient nutrient supply in the form of metabolisable energy and protein to maintain optimum growth rates in beef cattle (Webster, 2011). Efficient management of the land is a balance between grass supply and animal diet demand. Pastures may contain enough energy and protein to support cattle at all growth stages, depending on factors such as grass species, soil type, pH, drainage, and fertiliser use (Laser, 2007). Finishing cattle on grass is increasing in popularity for a number of reasons, including reducing reliance on costly supplementary feeds (Donnison and Fraser, 2016).

2.3.1 Grazing management

Grass growth can be divided into the following five main stages: (i) germination, (ii) vegetative, (iii) elongation, (iv) reproductive, (v) seed ripening (Moore et al., 1991). Grass plants have the ability to produce tillers, which will form until individual grass plants come in contact with each other (IGER Grassland Development Centre, not dated). Tillers age and die whilst new tillers are formed. Following the sowing of seeds, germination occurs when the grass plant grows from the seed. At this point, soil moisture is required in order for germination to occur. Following this, the vegetative stage is the point at which the first leaf appears. The elongation stage is simply the stage at which the grass elongates and nodes are formed. This leads on to the fourth stage of growth, when the plant head emerges. The seed ripening stage completes the cycle of grass growth and is the stage that occurs following seed forming and prior to germination (Moore et al., 1991).

Anecdotal evidence estimates that approximately 50% of grass grown in upland systems is wasted due to poor matching of grazing to periods of the grass growth, indicating poor grassland use efficiency (Farming Connect, 2013). This has been an ongoing issue for hundreds of years, with insufficient grass to satisfy the animal's need during periods of low or no growth, and an over-abundance during periods of peak growth (Garner, 1944). In order to sustain grassland production, a large proportion of the leaves produced must remain in the sward to contribute to photosynthesise (Parsons and Johnson, 1986). Grass should not be overgrazed in the spring, as it will lead to failure of subsequent grass growth. Increasing grass utilisation is largely dependent on the amount the grass has grown. Grazing intensity has a profound effect on the balance between grass' gross production and death, and forage maturity affects CH_4 emission from ruminants (Pinares-Patiño et al., 2007). Maintaining sward at a suitable height for grazing is important for maximum efficiency (Fuller, 1988; Parsons and Johnson, 1986). This implies that the grass is approximately 3-6 cm when grazed, not too short, nor too tall. Tall swards (e.g. over 9 cm) will have limited tillers and elongated stems, leading to a reduced leaf area (Parsons and Johnson, 1986). In beef suckler systems, early calving in January and February can be beneficial in that by the time cattle are turned out to graze, the calves are competent enough to be able to utilise the grass themselves, as well as benefitting from the additional milk produced by the cows due to the spring flush of grass (Garner, 1944).

Grassland use efficiency is largely dependent on its quality and intended use. Cow demand will vary depending on bodyweight and the stage of production cycle (Fuller, 1988). Therefore, it may be argued that the grassland's ability to supply sufficient nutrients to the animal is dependent on cow demand. Although store cattle and bulling heifers utilise coarse grasses, fattening cattle, cows and calves require grass of better quality to graze (Garner, 1944). This provides an opportunity to utilise different grasses for different classes of stock. In terms of livestock preferences, a study by Rutter et al. (2000) showed that both cattle and sheep, when unrestricted, will choose a diet of approximately 70% clover. Results have shown that dietary preferences change over the day, e.g. with increased clover consumed in the morning (Rutter et al., 2004a; 2004b).

2.3.1.1 Mixed grazing relative to solely grazing cattle

The differences between stock selectivity is important when deciding which herbivore to graze in order to manage biodiverse swards (Rook et al., 2004). The grazing habits of cattle are different to that of sheep due to the morphology differences of their teeth and jaws. Cattle are not as selective; their method of biting, tearing and pulling at the sward to a minimum of about 6 cm above ground level is different to a sheep's grazing habits. This results in a tussocky sward that is of different heights throughout; a structurally diverse sward. A sheep's morphology consists of a narrow mouth and curved incisor arcades, allowing for selective grazing, biting the sward to produce an even layer that can be grazed to 3 cm above ground level (Mansfield, 2011; Rook et al., 2004). It is believed that mixed grazing systems are beneficial for both cattle and sheep and increases grass productivity (Abaye et al., 1994; Brelin, 1979; Garner, 1944; Nolan and Connolly, 1977; Wright et al., 2006). Cattle will graze both the long and rough vegetation, allowing the improved sward to come through for sheep. Due to this, cattle are often portrayed as a supplementary conservation aid in upland regions (Critchley et al., 2008).

2.3.1.2 Grazing systems

Set-stocking and rotational grazing are the two most popular grazing systems. Set-stocking is the most common form of grazing management implemented, however, rotational grazing is increasingly operated within sheep systems, though notably less so than in other ruminant production systems such as dairy. The type of system operated on-farm is dependent on several factors including stock type, farm and field size and layout, and labour availability. Often, farms will implement a combination of grazing strategies in their system. The specifications of both grazing systems and their environmental impacts are summarised in the table below.

Table 2.2. Specifications of both set-stocking and rotational grazing systems implemented in livestock systems.

	Set-stocking	Rotational grazing
Description	• Livestock have access to whole field	• Field split into a number of plots using temporary fencing • Livestock graze plots for 0.5-7 days depending on when target measurement (grass height) is reached. Livestock then moved onto next plot
Benefits	• Cheap – no additional fencing and water supply required • Less labour required • Less poaching (stock spread out) • Less variation in seasonal grass growth	• Easier to view changes in grass quality and quantity (eye view and through measuring, e.g. with a plate meter) • Livestock dung spread evenly across plots • Smaller grazing area - easier to check livestock • Areas can be brought in/out of rotation depending on grass growth and demand
Disadvantages	• Can lead to stock grazing and re-grazing grass before allowing re-growth • Larger field so checking on stock is more time consuming • Sward less evenly grazed – stock will preferentially graze certain areas of field	• Expense – requires more fencing and water troughs in each paddock • More labour required (e.g. erecting fencing, transferring stock between plots) • Increased poaching – stock more confined to area/compacting the land

	• Measurements needed to assess changes in sward quality and quantity (not easy to quantify visually)	• Some livestock classes are less obedient of electric fencing systems • Shade required for each plot • Sward deteriorates more rapidly if rest periods don't match plants' needs • Seasonal distribution of grass growth more variable
Environmental impacts	• Dung not spread across whole field – likely to block out sunlight to sward, preventing grass growth and consequently leading to bare ground patches (Wilkins and Garwood, 1986) • Cattle will avoid grazing areas around dung patches, so ultimately, a large proportion of the pasture is wasted across the field	• Soil compaction leading to changes in soil structure and damage to plant roots and growing points (Wilkins and Garwood, 1986). Caused by increased treading and poaching due to increased pressure on the land (Simek et al., 2006) • Dung deposition more uniform across the grazing area provide form of nutrients (Wilkins and Garwood, 1986) – encourages sward heterogeneity by promoting introduction of grasses, e.g. sweet vernal-grass *(Anthoxanthum odoratum)* and meadow grass (*Poa annua)* • Livestock rotate around subplots, nutrients excreted from dung, absorbed into the soil therefore increasing its nutrient content at a period where subplot is rested from grazing (Gillet et al., 2010)

Despite the general benefits and disadvantages of both the above systems, there is a lack of studies available which assesses their respective economic and environmental benefits for beef production systems. Furthermore, the effectiveness of grazing systems is largely dependent on the efficiency of management applied. A higher grazing intensity often restricts the dietary preferences of cattle, and could potentially have a stocking rate capacity of 2-6 times higher than that on continuously grazed areas, whilst also resulting in improved pasture utilisation (Eaton et al., 2011). Thorough care must be given when deciding on stocking rates as it may affect carrying capacity and livestock performance (Fales et al., 1995; Parsons and Johnson, 1986).

It is accepted that implementing a rotational grazing system will result in increased grass production in the autumn. This provides numerous benefits, such as extending the grazing system for the stock and enabling grass to be carried over winter, thus allowing for early turnout the following year (Peyraud et al., 2004). However, in order to achieve the maximum benefits possible, familiarity with the stock as well as the land in terms of the potential of the available pasture is required. A set-stocked grazing regime is similar to how a herd of cows would graze in the wild, by eating the preferred, tall vegetation and moving on to the next section of land, leaving behind dung, urine and trampled vegetation.

A growing research base on the effectiveness of rotational grazing regimes globally have uncovered multiple benefits of the system's implementation. In areas of wetlands such as regions of west Brazil, e.g. the Pantanal, implementing rotational grazing practices has highlighted benefits beyond those stated above. These include increased cattle weight (15% higher than continuous grazing cattle) and pregnancy rates (22% higher than continuously grazing cattle) (Eaton et al., 2011). Furthermore, improved water quality and a possible reduction in disease transmission between livestock and wildlife was witnessed in rotational grazing systems. This was due to a more even distribution of dung and urine, resulting in reduced concentration of nutrient runoff to watercourses (Freitas et al., 2010). Similar studies include the ongoing strategy in northern Utah of altering their management of public grazing land to benefit wildlife and the environment by adopting a rotational grazing system (Vallentine, 2000). Findings highlighted more cattle per unit area being supported, increased pasture productivity and gains to soil quality, thus increasing bacterial activity in the soil whilst reducing the demand for artificial fertilisers (McGinty et al., 2009). This is expected from the rotational system as it provides the opportunity for grass roots to regrow, reducing the likelihood of severe soil erosion and runoff, and increasing infiltration, provided there is no added compaction. Some studies argue that the grazing system implemented does not have a significant effect on animal performance (Stejskalova et al., 2013). In New Zealand however, rotational grazing is commonly used within beef, dairy and sheep systems, with 'TechnoGrazing' being a grazing concept adopted across the country for over two decades. This grazing system uses rotational grazing practices, as rotating onto fresh pasture is believed to provide a healthier and consistent diet for the stock as well as enabling better control of grazing, rest, and animal husbandry (Charlton and Wier, 2001). It is inevitable that these benefits should substantially increase farm profitability. It is therefore questionable why, despite studies and published literature promoting the efficiency of adopting rotational grazing systems where appropriate, the popularity of adoption in the UK beef sector is so low. However, farmers are often apprehensive with regard to adopting a rotational grazing system on their farms due to the labour commitment required, despite research confirming improved incomes when the system is adopted (Kim et al., 2008).

2.3.2 Environmental impacts of grazing

Numerous studies in the UK have assessed the effectiveness of co-grazing cattle and sheep in providing environmental and biodiversity benefits without compromising livestock production. Such benefits include a reduction in g CH_4 per kg liveweight gain per hectare, and a positive effect on breeding birds (Fraser et al., 2014a). Mixed grazing offers the opportunity to maintain, or indeed, enhance livestock production output per ha, whilst providing secondary benefits (Fraser et al. 2014a). These include cattle's ability to meet habitat management prescriptions that are favourable for breeding birds and butterflies; thus positively impacting on the numbers of these species. However, enteric CH_4 emissions from grazing livestock are increased when poorer quality native grassland is consumed, thus increasing the environmental burden involved (Fraser et al., 2014b).

Results from Morgan-Davies et al. (2014) indicated that survey respondents largely believed that cattle grazing under careful management had a positive effect on hill environments. However, the responses to the extent of grazing impacts and breed types were varied within the survey. Grant et al. (1996) concluded that cattle grazing in the Scottish uplands for a five-year period decreased *N. stricta* cover from 55% to 30%. In contrast, sheep grazing increased the cover to 80% (Rook et al., 2004). This highlights the importance of cattle on this particular land in controlling the widespread species. Similarly, cattle are more willing to consume *M. carulea* than sheep (Grant et al., 1985), and grazing cattle at low intensities are preferred to sheep grazing for promoting *C. vulgaris* growth. There is a tendency for sheep to graze as a flock in areas that have recently been burned, thus grazing new shoots and causing a setback in *C. vulgaris* growth (Grant, 1968). This is a common problem of excess sheep grazing, which consequently negatively impacts on arthropods and breeding birds (Dennis et al., 2008; Evans et al., 2006; Evans et al., 2005). Higher grazing intensities, such as those associated with rotational systems, may reduce biodiversity (Milne and Osoro, 1997).

Various studies have been conducted into comparing sheep grazing systems with mixed grazing. However, studies into the effect of grazing cattle alone, on production and biodiversity are scarce. Discussions have included incorporating horses into upland farming systems due to their diverse grazing habits. Horses are hind-gut fermenters, and therefore readily graze tall, fibrous grasses that are of low quality in their diets and are successfully used in the Netherlands and in regions of the French uplands to control tall grass vegetation (Vulink, 2001). Despite this, claims have been made that horses are an ineffective conservation management tool in the UK due to regular cases of overstocking, leading to overgrazing (Bullock and Armstrong, 2000). Given that farm systems inherently seek to increase output per hectare and that there is no real market for horses in the UK, this compromises production and income. However, sheep and cattle as co-grazers can provide both farmers and conservationists with 'win-wins', and has been encouraged under agri-environment payment schemes (Evans et al., 2006). From a production perspective, careful measures must be undertaken to ensure grazing of both herbivores remain complementary, and not at the expense of the other.

2.3.3 Reseeding

Reseeding can be an effective method of increasing grass yield, thus promoting farm productivity. It is a common method of renewing pastures whereby the species mix within the sward is changed over time by sowing with more favourable (productive) species and varieties. The main explanatory factors driving farmers to reseed their pastures include aspirations to improve yield and quality, incorporate new grass varieties with improved genetics to tackle diseases and weeds, increase pasture response to fertiliser, and to mitigate soil compaction issues (AHDB, 2017). However, correct timing is essential to ensure weeds are controlled and the greatest likelihood of successful establishment. The cost-benefits of reseeding are summarised in Table 2.3 (AHDB, 2014).

Table 2.3. Outline of the main cost-benefits of reseeding pastures.

Benefits	Management costs	Environmental costs
Increased pasture productivity	Very costly operation – may also require ploughing beforehand to avoid competition from old grasses	Carbon release from soil disturbance, e.g. the ploughing of permanent pasture
Increased nitrogen use efficiency (NUE) of productive swards compared to weed grasses	New ley may be colonised by other grasses unless suitably fertilised and managed	NUE compromised as weed grasses re-establish; leading to N losses through leaching (Cuttle and James, 1995)
Improved drought resistance of new grass varieties	Level of success will be affected by weather conditions, e.g. drought would lead to lack of soil moisture to encourage germination	
Potential to increase the grazing season due to extended Spring and Autumn growth	Grazing too soon after reseeding could damage establishment	
Increased grass quality (e.g. protein content) – reduces requirement for supplementary feed of deficiencies in livestock diet		
Decreased potential of sward disease		
	Temporary loss of grazing - land required elsewhere to counteract this	

It is estimated that reseeding a five-year-old ley can increase pasture productivity by an additional £1,235 ha^{-1}. Furthermore, reseeding permanent pasture could lead to £2,000 ha^{-1} of supplementary pasture over five years (British Grassland Society, 2011). Despite this, reseeding is not always a practical operation to carry out on uplands due to the steep terrain preventing accessibility for machinery, and due to the costs outweighing the profits on the majority of reseeding operations carried out in the uplands (Mansfield, 2011). Where possible, perennial ryegrass (*Lolium perenne*) is the target dominant grass species for reseeding in pasture systems due to its rapid establishment, high yielding and palatability qualities (Farming Connect, 2013), along with its greater response to N fertiliser application than other grass species (Davies and Munro, 1974). Other suitable grass species for upland pastures include fescues (*Festuca*) and timothy (*Phleum pratense*).

The development of new grass varieties can increase nutrient use efficiency in pastures (Edwards et al., 2007). The Institute of Biological, Environmental and Rural Sciences (IBERS), Aberystwyth University, have developed a number of new grass varieties that are high in sugar content, named AberHSG. The first permanent high sugar grass variety, AberDart was produced in 2000, but since then, a number of high sugar grass varieties have been developed globally; with the aim of providing a better balance of carbohydrates to proteins in grass, thereby improving feed conversion efficiency and reducing CH_4 emissions (Edwards et al., 2007; Ellis et al., 2012). With enteric fermentation by ruminants responsible for 39% of total CH_4 emissions from global livestock (FAO, 2019), and between 2% and 12% of gross energy intake by ruminants believed to be lost as CH_4 (Johnson and Johnson, 1995), improving ruminants' feed conversion efficiency without compromising meat production is vital (Martin et al., 2010). In addition to this, more recent new grass varieties such as AberMagic, a high-sugar ryegrass, have the ability to reduce nitrogen emissions from livestock (Staerfl et al., 2012). This is due to the high sugar content allowing for a higher digestibility value, thus improved utilisation of protein by livestock so less nitrogen is emitted in faeces and urine (Edwards et al., 2007).

Recently developed grasses include clover varieties such as AberGuard which has an enhanced resistance to clover eelworm, which is problematic in warm, wet winters (Frame and Laidlow, 2011). Varying grass composition by altering forage mixtures, e.g. incorporating nitrogen-fixing legumes is deemed effective in improving diet quality as well as improving grass utilisation, leading to some environmental gains (Garnett, 2011; Hyland et al., 2016b; Soteriades et al., 2018). The adoption of clover into upland swards reduces the demand for fertiliser and increased feed to the rumen, whilst not compromising livestock systems' stock carrying capacity either (Phelan et al., 2015).

Despite the studies mentioned above proving the efficiency of incorporating newer grass varieties into agricultural systems, the majority of the research concentrates on dairy systems, such as the study by Staerfl et al. (2012). However, there is a possibility of decreasing the cost of producing livestock in general in the uplands by increasing the efficiency of the grass by incorporating high sugar grasses, provided that the grazing is managed for optimal quality. This would lead to reduced demand for concentrate feed, improved growth rates and consequently, decreased GHG emissions from livestock.

Furthermore, plant breeders are continuously developing mixtures specifically designed for different farming systems, e.g. upland and hill sites. Information on establishment, production and utilisation of the particular seed mixtures on-farm is often unavailable. Raising farmers' awareness of the environmental impact of their agricultural practices and the benefits of incorporating new grass varieties into the sward is required to increase the likelihood of implementation (Dietz et al., 2007) as Hyland et al. (2016a) confirmed the adoption of environmental measurements by farmers is largely influenced by the practicality of the strategy and its cost. With the UK aiming for net zero GHG emissions by 2050, reducing emissions from livestock systems is vital. Incorporating the correct seed mixture when reseeding could be part of the solution to this.

2.3.4 Fertiliser use

A suitable fertiliser application programme can increase pasture yields. Prior to fertiliser application, soil testing is required to target applications for better nutrient use efficiency. Nutrient application guidelines such as the RB209 Nutrient Management Guide (AHDB, 2019) can then be used as a guideline so the correct nutrients are applied at the right amounts. It is also important to consider the nutrient input from manures, slurry and any other organic resource inputs (e.g. sewage sludge) when calculating nutrient inputs required from fertiliser, which also provide the benefits of increasing soil organic matter levels.

Maintaining that the correct stocking rate is operated ensures maximum grass utilisation, whilst fertiliser application is dependent on stocking intensity; and whilst a higher stocking rate generally requires greater fertiliser application, optimising stocking densities can increase pasture utilisation without necessarily requiring greater fertiliser use. This is of note, given that there are several potentially negative impacts of applying fertilisers. For instance, they are costly to purchase, can increase soil emissions of N_2O (Cardenas et al., 2019), and regular application can also reduce soil organic matter in the long-term (Kotschi, 2013). Cuttle and James (1995) reported greater concentrations of organic-N, potassium, phosphorus and calcium in drainage water as a result of lime and fertiliser applications to reseeded upland pasture in previous years. Table 2.4 highlights the main environmental impacts of applying fertiliser, and how they can be mitigated.

Table 2.4. Summary of main environmental impacts deriving from fertiliser use.

Environmental impact	Cause	Mitigation
Nutrient loss – N_2O emissions	Excess fertiliser applied onto land at incorrect time, soils inadequately absorb the nutrient/insufficient plant uptake	Target time of grazing and fertiliser application
Nutrient run-off and eutrophication in watercourses	Secondary effects of poor fertiliser application, increased surface water	Targeted fertiliser application in appropriate weather conditions

Traditionally, hill land received low production inputs. The England and Wales Hill Farming Committee highlighted several recommendations in 1944, including reseeding in-bye land, draining, lime and potash application as well as bracken control and heather burning in order to "maintain the land in good working order" (Balfour, 1942; De La Warr, 1944). However otherwise, any mention of fertiliser application on hill land is scarce. Similarly, fertiliser use in the uplands is generally low (Acs et al., 2010). Agri-environment schemes can also reward the maintenance of land under low intensity management e.g. reduced stocking rates and fertiliser applications, particularly in the hills and uplands (Critchley et al., 2008; Welsh Government, 2018).

2.3.5 Breed effect

Native cattle such as Highland and Welsh Black breeds have adapted to cold, harsh winter environments and are willing to graze the poor-quality swards on marginal land (Mansfield, 2011; Rook et al., 2004). Due to this, they are often considered hardier (Morgan-Davies et al., 2014), and upland (especially hill) livestock are often free to roam around a large area, establishing home ranges, and generally spend an increased amount of time on a certain area of land (Lawrence and Wood-Gush, 1988). This provides an opportunity for the utilisation of poor-quality grasses by beef suckler systems in the uplands. These attributes of native breeds allow increased reproductive success on marginal, mature pastures when compared with continental breeds (D'hour et al., 1998).

Despite the benefits of native cattle breeds, in recent times there has been a tendency for UK farmers to breed continental cattle such as Limousin and Belgian Blue beef cattle, originally from France and Belgium. Generally, the native breeds are considered to be slow maturing and to gain weight due to their low meat-to-carcass ratio in comparison to the continental breeds (Fraser et al., 2014b), adding cost to the farmer that may not always be recovered through selling at a premium. There is increased pressure on livestock systems to increase efficiency by finishing cattle by 16-18 months rather than 20 months and older, and continental cattle breeds have shown to outperform traditional breeds from this perspective (Yarwood and Evans, 1999). Stabiliser cattle are also increasing in popularity in the UK. Traditionally, the breed was adopted as suckler cows in beef systems due to the low costs of production involved. However, the realisation of the quality of meat achieved from finishing Stabiliser cattle has led to an increase in their population in the UK. As previously mentioned, farmers alter their systems to meet consumer demand and with the pressure of producing high output and quality animals, other traits such as hardiness are not deemed a priority. However, a crossbreeding programme is considered to be advantageous, leading to an improved performance from the herd, as well as adopting the beneficial traits of both native and continental cattle breeds.

Hoffman (2010) suggested there may be more value in operating native cattle systems due to their ability to cope with disturbances such as extreme weather conditions in the form of drought, floods and diseases related to climate change that may affect their feed supply. Following the CAP reform and a shift away from production-based agriculture to environmental management, this has become increasingly more important as land may be reserved for the provision of ecosystem services, thus limiting land for grazing. In summary, the general perception is that livestock systems based on traditional breeds of cattle may suffer economically in the form of reduced production efficiency and economic output (Rook et al., 2004).

Regardless of the general assumption that all cattle breeds follow the same grazing habits, it has been noted that differences in body size is a primary factor affecting biodiversity (Rook et al., 2004). The belief is that continental cattle breeds may have poorly developed features including behavioural responses to biodiverse grasses (Tolhurst and Oates, 2001); therefore, native breeds are generally favoured over the continentals for biodiversity purposes. However, there is a lack of current data on whether this can be overcome. In fact, Isselstein et al. (2007) concluded that traditional cattle breeds did not perform better than commercial breeds from a production perspective within extensive pasture systems. Cattle with a smaller genotype graze more selectively due to their reduced body intake capacity, thus leading to grassland communities of increased heterogeneity, which is beneficial for biodiversity purposes. A within-breed study by Cid et al. (1997) stated that the degree of selective grazing may be influenced by the individual’s genotype. Despite this, there is insufficient scientific research directly comparing breeds to support this. Of the available research, one study found very few differences in diet selection of grassland vegetation, with the traditional cattle breeds only slightly less selective than commercial breeds (Dumont et al., 2007).

Studies that have been undertaken on the environmental impacts, such as Fraser et al. (2014), have concluded that breed type does not significantly affect enteric CH_4 emissions from cattle grazing. Berry et al. (2003) recorded the nitrogen intakes of Brown Swiss Dairy Cows and Highland suckler cows on improved Alpine pastures. The results showed that the Highland, native cattle had excessive nitrogen intakes relative to their growth, leading to increased urine being discharged into the vegetation, therefore increased nitrogen lost through leaching and as GHG emissions (Boon et al., 2014). Furthermore, the study indicated the same trend on unimproved pastures, with high nitrogen losses to the atmosphere.

2.3.6 Supplementary feeding

Several factors contribute to determining a cow's demand for food. These factors include stocking rate, sward composition, the animal's physiology, and the interaction between these three factors. Several forms of feed supplementation are available, with varying nutrient supply. As previously mentioned, there are numerous factors that may be responsible for the variation in energy and protein needs of cattle. In the event of their unavailability, supplementation may be required.

Feed intake in beef cattle fed on forage is limited by the rate at which the food can be digested in the rumen (Mcdonald et al., 2002). Different forages promote different intakes, for instance a forage with a high cell wall content will promote low intake as it is of low digestibility. Some believe that grazed grass alone is insufficient to meet the nutritional requirements of cattle and sheep due to its low carbon:nitrogen ratio (Rook et al., 2004). This is largely dependent on the condition and composition of the grass produced, which may vary from season to season due to factors such as fluctuating rainfall and temperature. The demand for supplementation is influenced by a number of factors, including livestock physiology, grazing habits and stage of grass growth.

A study by Rutter et al. (2000) looked at the grazing habits of dairy cows and sheep, and observed an increase of clover in the diet during the day and contrastingly, increased grass consumption at night. This may be due to an increase in grass sugar levels at night, thus increasing digestibility (Orr et al., 2001). It is argued that improving diet quality in the form of nutrient dense supplements is often more favourable than agronomic inputs in terms of environmental effect, by reducing CH_4 emissions released to the atmosphere (Donnison and Fraser, 2016; Martin et al., 2010). Results reflect that upland beef farms had a higher carbon footprint than farms at lower elevations (Hyland et al., 2016). This is presumably due to slower growth rates, thus increasing time taken to finish upland cattle for slaughter, or the need for an increased number of cattle to counteract the lower liveweights of the animals. Feeding supplements or feed crops reduce the land area required for a given quantity of nutrition, which may lead to decreased GHG emissions from the production system as a whole, as well as an increase in stock growth rates and therefore reduced CH_4 emissions (Garnett, 2011; Jones et al., 2013). Although not always practical, supplementary feeding may be particularly useful on upland farms where animals graze poor quality vegetation. Contradictorily, other studies have shown that upland beef systems are deemed to emit less GHG emissions, particularly N_2O, than other beef systems (Webb et al., 2014). This is due to none or low levels of supplementary feeding, and therefore relying on pasture as forage in the early stages of life. Youngstock are then sold on to be finished on lowlands in more favourable conditions, therefore a reduced environmental impact during the growth period in the uplands.

The required level of supplementary feeding depends on the livestock's nutritional needs, condition and health. They are used to provide the trace elements required by cattle. Vitamin and mineral feed supplements in the form of mineral blocks, slow release boluses and minerals dissolved in drinking water are increasingly used in beef systems to balance deficiencies such as phosphorus, magnesium and copper (Donovan and Weigel, 1988; Habib, 2004). The diversity of options available provide hill and upland systems with practical and low-input options, e.g. placing mineral blocks on areas of land that need to be heavily grazed. Under-nutrition can lead to several other problems including poor growth rates, failure to conceive, poor offspring survival rate and failure to finish stock to meet market requirements (Hinton, 2007). However, when considering supplementary feed, the liveweight gain of the cattle must be sufficient to justify the cost of the supplements in order to maintain a viable system. Of the literature that is available, supplementary feeding resulted in increased price returns on cattle (Tronstad and Teegerstrom, 2001); although this is clearly dependent on the degree of fluctuation in price (AHDB, 2020).

2.4 Uplands significance: Wales as a case study

Approximately 58% of Wales' land area is classified as uplands, compromising of 800,000 - 1,000,000 ha of land (Johnstone et al., 2008). Most of this land is designated as LFA, due to the recognition that they are areas of economic disadvantage and poor land quality, from a production sense (Reed et al., 2009b). Seventy seven percent of Wales' agricultural land area is classified as LFA as a result of poor soil quality and wet climate (Simmons, 2002), with 56% of Wales' land area classified as Severely Disadvantaged Areas (Welsh Government, 2019a). Due to this, the land is mainly managed for livestock grazing, with timber production another fundamental land use (Welsh Government, 2016). Statistics from the Welsh June Agricultural Survey in 2016 demonstrated that approximately 46% of the country's agricultural land was classified as cattle and sheep farms in the LFA, indicating the extent of these livestock systems in Wales (Welsh Government, 2016). Within these LFA, there are approximately 28,000 agricultural holdings, directly and indirectly employing 42,000 farmers (Johnstone et al., 2008). More recent figures on LFA holdings and employment were not available.

Figure 2.3 below highlights the distribution of Wales' uplands (as mountains, moorlands and heaths) according to the UK National Ecosystem Assessment.

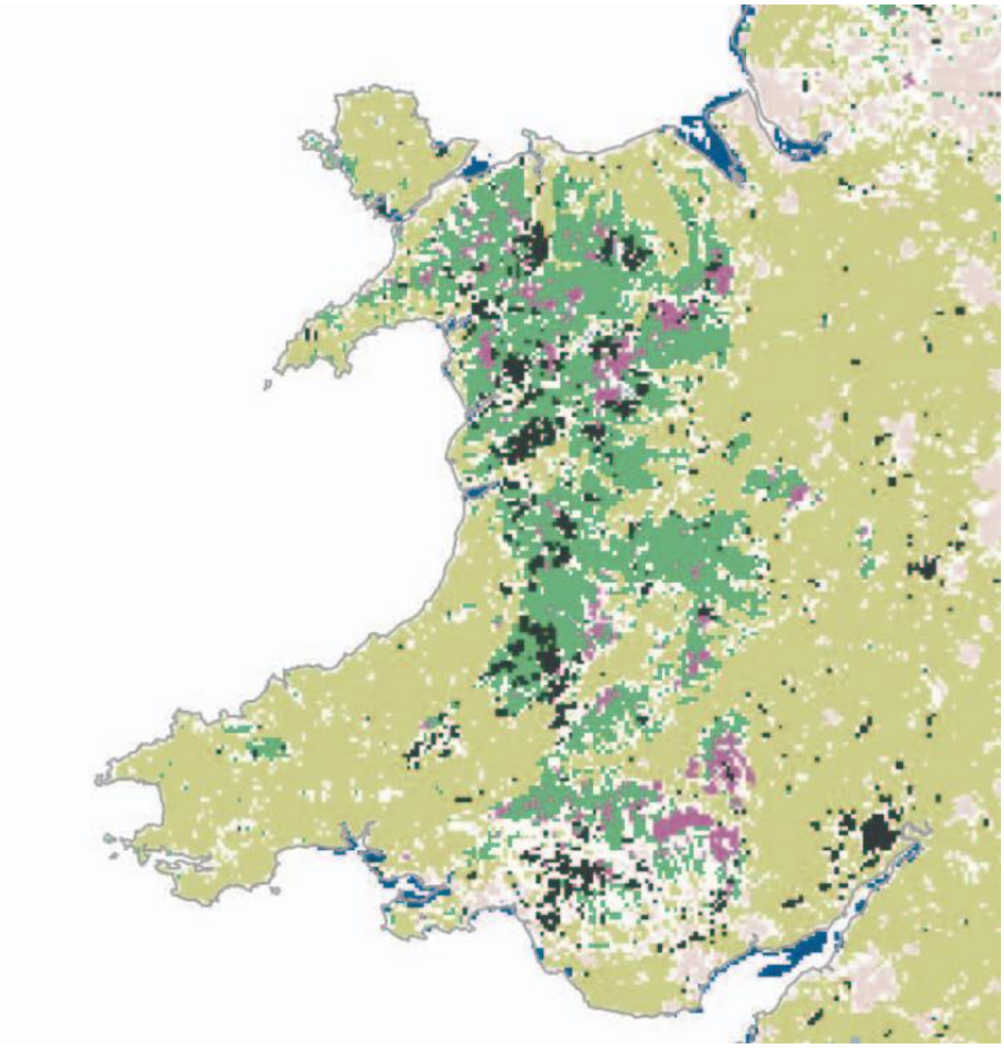

Figure 2.3. Distribution (%) of the UK National Ecosystem Assessment Broad Habitat types in Wales by area at 1 × 1 km resolution (UK National Ecosystem Assessment, 2011). Purple areas represent Mountains, Moorlands and Heaths. (For interpretation of all the references to colour in this figure legend, the web version of this figure can be accessed from the reference list).

Figures show that the number of people employed on Welsh farms has decreased over recent years (Welsh Government, 2016), with off-farm employment being encouraged (Morris et al., 2017). However, it cannot go unnoticed that the uplands were recognised for their importance in supporting rural livelihoods locally, nationally and internationally in the Millennium Ecosystem Assessment (2005).

As can be seen in Figure 2.4, cattle numbers generally reduced in Wales from 2002 to 2015. While these figures are for total Welsh cattle, the uplands is likely to have followed a similar, if not more pronounced decrease.

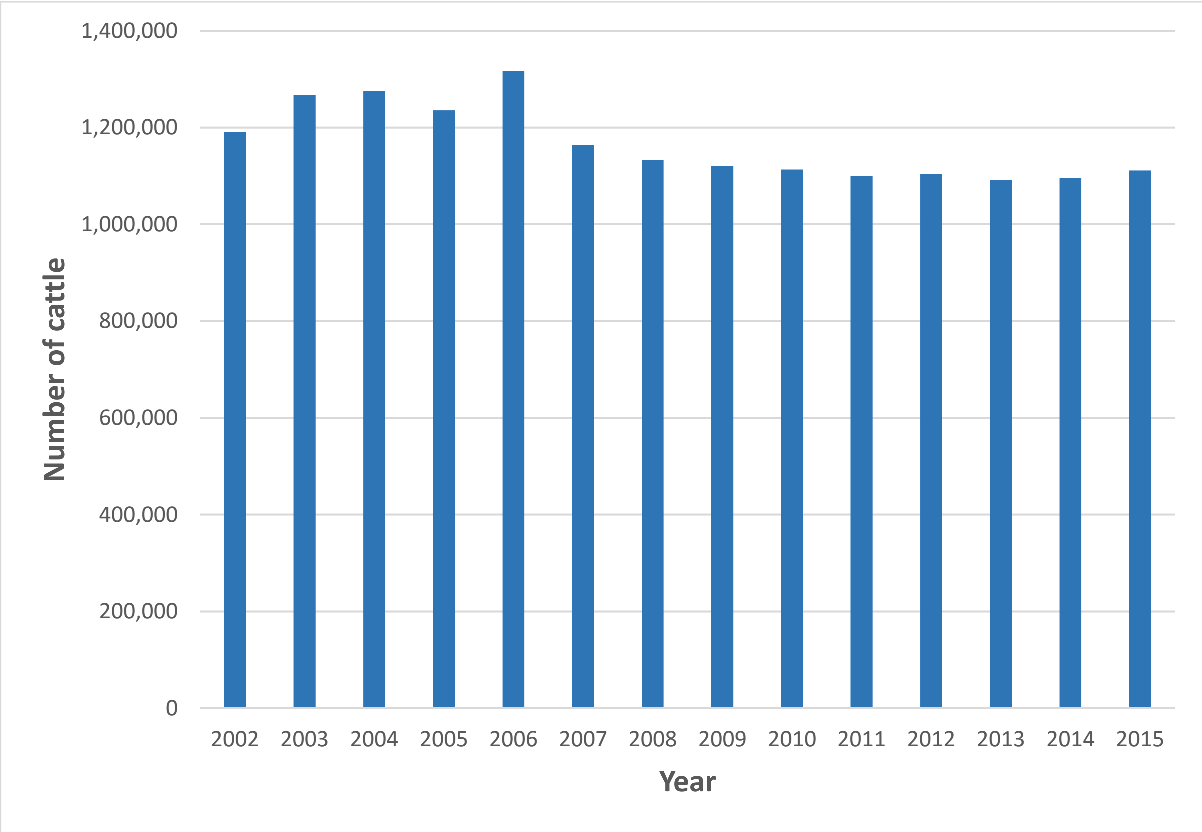

Figure 2.4. Total cattle numbers in Wales annually from 2002-2015. Data collected in annual Welsh June Agricultural Surveys. Information derived from direct survey questions 2002-2006, numbers drawn from the Cattle Tracing System for improved accuracy of figures from 2007 onwards (Welsh Government, 2016).

A large proportion of Wales' uplands highlighted in Figure 2.3 is common land, used for grazing. Upland agriculture in Wales is marginal, and systems often depend on the common land for grazing for their enterprises to remain in viable condition (Johnstone et al., 2008), or on other sources of income. For instance, the upland areas of Snowdonia National Park and the Brecon Beacons attract as much as 17.5 million visitors per year, bringing a significant source of tourism revenue to many farm businesses in the area (Midmore and Moore-Colyer, 2005; Talbot, 2013).

A number of habitats and species listed in the Welsh Government's List of Species and Habitats of Principal Importance for Conservation of Biological Diversity in Wales (under Section 42 of the Natural

Environment and Rural Communities Act 2006) are found in the uplands. These include 5% of the UK upland heath and 5% of the UK's blanket bog (Johnstone et al., 2008). Going forward, managing the uplands for the delivery of agricultural outputs as well as ecosystem services is seen as imperative for the viability of such enterprises (Welsh Government, 2019b).

2.5 The future of cattle in the uplands

As discussed, there are a number of factors influencing upland livestock systems in the UK, particularly suckler beef systems. The future of UK uplands is largely dependent on the factors mentioned as the drivers of change in the past, including political, environmental, socio-cultural and economic drivers; with substantial uncertainties (Welsh Government, 2019b). Upland policy will be contingent upon UK legislation following leaving the European Union. The viability and degree of beef production systems in the uplands in the future is largely unknown. However, its importance as a source of rural jobs and the favouring of cattle for the delivery of other ecosystem services (e.g. for conservation purposes) may mean that future initiatives are implemented to maintain, or even enhance, cattle numbers in the uplands. Of the studies that are available looking at the efficiency of beef cattle systems, the majority are based on finishing cattle. However, research into upland beef systems and grazing management would be useful; considering each phase of the production cycle in detail, thus improving the integration and efficiency of the system. There is therefore a need to critically assess the effectiveness of the different grazing systems available to farmers, from both an economic and environmental perspective. However, alternative benefits provided by the uplands in the form of ecosystem services and public benefits are fundamental and must not be compromised.

The wealth of beneficial ecosystem services and public benefits that are delivered by the upland landscape in the UK cannot go unnoticed. These ecosystem services include food, fibre and timber provision, climate regulation in the form of carbon storage, flood mitigation, recreation, biodiversity, drinking water and landscape beauty to name a few (Bonn et al., 2009a; Dougill et al., 2006; Hubacek et al., 2009; Thompson et al., 1995). Notably since the 1950s, the economy of the uplands has altered to prioritise the provision of secondary services in the form of these public benefits (Reed et al., 2009a). In the UK, over 70% of drinking water is provided by the uplands (Heal, 2003). The use of grazing livestock for alternative benefits such as the provision and maintenance of ecosystem services is deemed possible when well managed (Garnett, 2011). Removing livestock from the uplands and using the poorer quality vegetation for alternative land uses e.g. forestry and biomass would provide a source of renewable fuel, however, the output yielded from the livestock would have to be produced on land elsewhere (Garnett, 2011). In addition to this, previous work has shown that farmers are often hesitant to endorse an alternative land use, e.g. forestry or a form of diversification (Morgan-Davies et al., 2017). The debate is whether, or how, the uplands should provide multiple benefits, from food production and biodiversity benefits, or should they be managed solely for a single purpose.

Some believe further prioritisation should be given to fully valuing the uplands in the future, notably as the general shift from rural to urban societies is causing a change in people's priorities for the land, from production to environmental stewardship (Donnison and Fraser, 2016). The wealth of multiple public benefits delivered by the uplands suggests that competition between stakeholders for the land will increase dramatically in the future (Bonn et al., 2009b), leading to potential conflict that could be avoided by the adoption of multiple land activities to provide use and delight to all (Morgan-Davies et al., 2015).

Sustainable intensification may be an answer to this, in increasing food production whilst enhancing ecosystem services and public benefits. It is a fairly recent but developing concept that lacks a statutory definition. One definition put forward is "producing more output from the same area of land while reducing the negative environmental impacts and at the same time increasing contributions to natural capital and the flow of environmental services" (Conway and Waage, 2010; Godfray et al., 2010; Pretty, 2008; Royal Society, 2009). Sustainable intensification may aid in maintaining and possibly increasing food production while protecting the environment (Petersen and Snapp, 2015). Indeed, some perceive it to be part of the solution to ensuring both long-term food security and provision of ecosystem services. To date though, one may argue that methods of implementing sustainable intensification on a large scale are uncertain (Petersen and Snapp, 2015).

One cannot dispute that sustainability has become an integral part of the agri-food sector (Francis et al., 2003), and in order to satisfy the demand of the 9 billion global population expected in 2050, discovering agricultural systems that benefit both the environment as well as food production is vital. Indeed, beef produced from semi-natural rough grazing may offer health benefits to consumers compared to beef produced on improved permanent pasture, such as significantly more vitamin E (Fraser et al., 2009). Grazing cattle pose an opportunity for sustainable intensification in the UK uplands by enhancing both food production and supply of ecosystem services; though it must differentiate between past intensification practices which negatively impacted on biodiversity (Dawson et al., 2011), amongst other environmental impacts. Further research is needed to discover how to quantify sustainable intensification and the possible contribution of cattle in the uplands to this concept.

2.6 References

Abaye, A.O., Allen, V.G., Fontenot, J.P., 1994. Influence of grazing cattle and sheep together and separately on animal performance and forage quality. *Journal of Animal Science 72*, 1013-1022.

Acebrón, L.B., Dopico, D.C., 2000. The importance of intrinsic and extrinsic cues to expected and experienced quality: an empirical application for beef. *Food Quality and Preference 11*(3), 229-238.

Acs, S., Hanley, N., Dallimer, M., Gaston, K.J., Robertson, P., Wilson, P., Armsworth, P.R., 2010. The effect of decoupling on marginal agricultural systems: implications for farm incomes, land use and upland ecology. *Land Use Policy 27*(2), 550-563.

AHDB, 2020. Feed prices and markets. [Online]. Available at: https://ahdb.org.uk/dairy/feed-prices-and-markets. Accessed 16th January, 2020.

AHDB, 2019. Nutrient Management Guide (RB209). [Online]. Available at: https://ahdb.org.uk/nutrient-management-guide-rb209. Accessed 20th April, 2020.

AHDB, 2018a. Consumer Insights – July 2018. [Online]. Available at: https://media.ahdb.org.uk/media/Default/Consumer%20and%20Retail%20Insight%20Images/PDF%20articles/ConsumerInsights%20WEB_1653_180725.pdf. Accessed 16th January 2020.

AHDB, 2018b. New GB-wide campaign highlights red meat's positive health rose. [Online]. Available at: https://ahdb.org.uk/news/new-gb-wide-campaign-highlights-red-meat-s-positive-health-role. Accessed 16th January 2020.

AHDB, 2017. Grassland reseeding guide. [Online]. Available at: http://beefandlamb.ahdb.org.uk/wp-content/uploads/2018/06/Grassland-reseeding-guide.pdf. Accessed 16th January 2020.

AHDB, 2014. The Two A's of Reseeding. [Online]. Available at: https://dairy.ahdb.org.uk/news/technical-articles/august-2014/the-two-as-of-reseeding/#.WN4Zp1Xyu00. 20th April 2020.

Anderson, P., Yalden, D.W., 1981. Increased sheep numbers and the loss of heather moorland in the Peak District, England. *Biological Conservation 20,* 195-213.

Ashworth, S., 2000. Consumer perceptions of meat quality and their willingness to pay a premium. In: Scottish Agricultural College Economic and Social Science Report 1999, Scottish Agricultural College, Edinburgh.

Baghurst, K., Record, S., Leppard, P., 2000. Red meat consumption in Australia: intakes, nutrient contribution and changes over time. *Australian Journal of Nutrition and Dietetics 57,* S1-S36.

Balfour, A.J., 1942. Report of the Committee on Hill Sheep Farming in Scotland. Department of Agriculture for Scotland: London.

BeefQ, 2019. Research. [Online]. Available at: http://www.beefq.wales/research.html. Accessed 21st August 2019.

Berry, B.W., Hasty, R.W., 1982. Influence of demographic factors on consumer purchasing patterns and preferences for ground beef. *Journal of Consumer Studies and Home Economics 6*, 351-360.

Berry, N.R., Jewell, P.L., Sutter, F., Edwards, P.J., Kreuzer, M., 2003. Selection, intake and excretion of nutrients by Scottish highland suckler beef cows and calves, and brown Swiss dairy cows in contrasting Alpine grazing systems. *Journal of Agricultural Science 139*, 437-453.

Bignal, E.M., McCracken, D.I., MacKay, A., 1999. The economics and ecology of extensively reared Highland Cattle in the Scottish LFA: an example of a self-sustaining livestock system. In: Laker, J.P., Milne, J.A. (Eds.), Livestock Production in Less Favoured Areas: Meeting Future Economic, Environmental and Policy Objectives through Integrated Research. Proceedings of the 2nd Conference of the LSIRD Network. Macaulay Land Use Research Institute, Aberdeen.

Bonn, A., Hubacek, K., Stewart, J., Allott, T., 2009a. Drivers of Environmental Change in Uplands. Routledge, London.

Bonn, A., Rebane, M., Reid, C., 2009b. Ecosystem services: a new rationale for conservation of upland environments. In: Bonn, A., Allott, T., Hubacek, K., Stewart, J. (Eds.), Drivers of Environmental Change in Uplands. Routledge, London, 448–474.

Boon, A., Robinson, J.S., Chadwick, D.R., Cardenas, L.M., 2014. Effect of cattle urine addition on the surface emissions and subsurface concentrations of greenhouse gases in a UK peat grassland. *Agriculture, Ecosystems and Environment 186*, 23-32.

Bowers, J.K., 1985. British agricultural policy since the Second World War. *The Agricultural History Review 33*(1), 66-76.

Brelin, B., 1979. Mixed grazing with sheep and cattle compared with single grazing. *Swedish Journal of Agricultural Research 9*, 113-120.

British Grassland Society, 2011. Making the most of your reseed. [Online]. Available at: http://www.britishgrassland.com/system/files/Reseeding%20Factsheet%202011.pdf. Accessed 18th December 2019.

Bullock, C.H., Kay, J., 1997. Preservation and change in the upland landscape: the public benefits of grazing management. *Journal of Environmental Planning and Management 40*(3), 315-334.

Bullock, D.J., Armstrong, H.M., 2000. Grazing for environmental benefits. In: Grazing management: the principles and practice of grazing, for profit and environmental gain, within temperate grassland systems. Proceedings of the British Grassland Society Conference held at the Cairn Hotel, Harrogate, 29 February-2 March 2000. (191-200). BBSRC Institute of Grassland and Environmental Research.

Bunce, R.G.H., Barr, C.J., Whittaker, H.A., 1983. A stratification system for ecological sampling. In: Ecological mapping from ground, air and space. Edited by R.M. Fuller, 39-46. (ITE symposium no. 10.) Cambridge: Institute of Terrestrial Ecology.

Cardenas, L.M., Bhogal, A., Chadwick, D.R., McGeough, K., Misselbrook, T., Rees, R.M., Thorman, R.E., Watson, C.J., Williams, J.R., Smith, K.A., Calvet, S., 2019. Nitrogen use efficiency and nitrous oxide emissions from five UK fertilised grasslands. *Science of the Total Environment 661,* 696-710.

Chambers, F.M., Daniell, J.R., 2011. Conservation and habitat restoration of moorland and bog in the UK uplands: a regional, paleoecological perspective. *PAGES Newsletter*, *19*(2), 45-47.

Charlton, J.F.L., Wier, J.H., 2001. TechnoGrazing™- a new grazing concept. In: Proceedings of the conference – New Zealand Grassland Association, 33-36.

Cid, M.S., Brizuela, M.A., Aello, M.S., Ferri, C., 1997. Pasture heterogeneity created by grazing of cattle biotypes with different body size. In: Proceedings of the XVIIth International Grassland Congress. Winnipeg, Canada, 29.51-29.52.

Collins, E.J.T., 1978. The economy of upland Britain, 1750-1950: an illustrated review. Centre for Agricultural Strategy, University of Reading, Reading.

Committee on Climate Change, 2019. [Online]. Reducing the UK's emissions. Available at: https://www.theccc.org.uk/our-impact/reducing-the-uks-emissions/. Accessed 18th December 2019.

Conway, G. R., Waage, J., Delaney, S., 2010, Science and Innovation for Development; UK Collaborative on Development Sciences, UKCDS, London.

Countryside Commission, 1984. A Better Future for the Uplands/Upland Landscape Change: a review of statistics. Countryside Commission: Cheltenham.

Critchley, C.N.R., Adamson, H.F., McLean, B.M.L., Davies, O.D., 2008. Vegetation dynamics and livestock performance in system-scale studies of sheep and cattle grazing on degraded upland wet heath. *Agriculture, Ecosystems and Environment 128*(1), 59-67.

Cuttle, S.P., James, A.R., 1995. Leaching of lime and fertilisers from a reseeded upland pasture on a stagnogley soil in mid-Wales. *Agricultural Water Management 28*(2), 95-112.

D'hour, P., Revilla, R., Wright, I.A., 1998. Possible adjustments of suckler herd management to extensive situations. *Animal Research 47*, 453-463.

Daley, C.A., Abbott, A., Doyle, P.S., Nader, G.A., Larson, S., 2010. A review of fatty acid profiles and antioxidant content in grass-fed and grain-fed beef. *Nutrition Journal 9*(1), 10.

Daniel, C.R., Cross, A., Koebnick, C., Sinha, R., 2011. Trends in meat consumption in the United States. *Public Health Nutrition 14*(4), 575-583.

Davies, D.A., Munro, J.M.M., 1974. Potential pasture production in the uplands of Wales: 4. Nitrogen response from sown and natural pastures. *Grass and Forage Science 29*(2), 149-158.

Dawson, L.E.R., O'Kiely, P., Moloney, A.P., Vipond, J.E., Wylie, A.R.G., Carson, A.F., Hyslop, J., 2011. Grassland systems of red meat production: integration between biodiversity, plant nutrient utilisation, greenhouse gas emissions and meat nutritional quality. *Animal 5*(9), 1432-1441.

Defra, 2018. Nitrogen vulnerable zones. [Online]. https://www.gov.uk/government/collections/nitrate-vulnerable-zones#rules-you-must-follow. Accessed 15th January 2020.

De La Warr, E., 1944. Report of the Committee on Hill Sheep Farming in England and Wales, Cmnd 6498, Ministry of Agriculture and Fisheries, London.

De Vries, M.D., Van Middelaar, C.E., De Boer, I.J.M., 2015. Comparing environmental impacts of beef production systems: A review of life cycle assessments. *Livestock Science 178*, 279-288.

Dennis, P., Skartveit, J., McCracken, D.I., Pakeman, R.J., Beaton, K., Kunaver, A., Evans, D.M., 2008. The effects of livestock grazing on foliar arthropods associated with bird diet in upland grasslands of Scotland. *Journal of Applied Ecology 45*, 279-287.

Dietz, T., Dan, A., Shwom, R., 2007. Support for climate change policy: Social psychological and social structural influences. *Rural Sociology 72*(2), 185-214.

Donnison, I.S., Fraser, M.D., 2016. Diversification and use of bioenergy to maintain future grasslands. *Food and Energy Security 5*(2), 67-75.

Donovan, D., Weigel, G.C., 1988. Animal feed blocks containing dietary supplements. U.S. Patent, 4,735,809.

Dougill, A., Fraser, E.D.G., Holden, J., Hubacek, K., Prell, C., Reed, M.S., Stagl, S.T., Stringer, L.C., 2006. Learning from doing participatory rural research: lessons from the Peak District National Park. *Journal of Agricultural Economics 57*, 259–275.

Dumont, B., Rook, A.J., Coran, C., Röver, K.U., 2007. Effects of livestock breed and grazing intensity on biodiversity and production in grazing systems. 2. Diet selection. *Grass and Forage Science 62*(2), 159-171.

Eaton, D.P., Santos, S.A., Santos, M.D.C.A., Lima, J.V.B., Keuroghlian, A., 2011. Rotational grazing of native pasturelands in the Pantanal: an effective conservation tool. *Tropical Conservation Science 4*(1), 39-52.

Edwards, G.R., Parsons, A.J., Rasmussen, S., Bryant, R.H., 2007. High sugar ryegrasses for livestock systems in New Zealand. In: *Proceedings of the New Zealand Grassland Association 69,* 161-171.

EEA, 2017. Environmental indicator report 2017 — In support to the monitoring of the 7th Environment Action Programme, European Environmental Agency, Copenhagen.

EEA, 2015. The European environment — state and outlook 2015: synthesis report. European Environment Agency, Copenhagen.

Ellis, J.L., Dijkstra, J., France, J., Parsons, A.J., Edwards, G.R., Rasmussen, S., Kebreab, E., Bannink, A., 2012. Effect of high-sugar grasses on methane emissions simulated using a dynamic model. *Journal of Dairy Science 95*(1), 272-285.

English Nature, 1996. Developing definitions of natural capital for use within the uplands of England. No. 197 – English Nature Research Reports. [Online]. Available at: http://publications.naturalengland.org.uk/file/6159266016657408. Accessed 20th April 2020.

Evans, D.M., Villar, N., Littlewood, N.A., Pakeman, R.J., Evans, S.A., Dennis, P., Skartveit, J., Redpath, S.M., 2015. The cascading impacts of livestock grazing in upland ecosystems: a 10-year experiment. *Ecosphere 6*(3), 1-15.

Evans, D.M., Redpath, S.M., Evans, S.A., Elston, D.A., Gardner, C.J., Dennis, P., Pakeman, R.J., 2006. Low intensity, mixed livestock grazing improves the breeding abundance of a common insectivorous passerine. *Biology Letters 2*(4), 636-638.

Evans, D.M., Redpath, S.M., Evans, S.A., Elston, D.A., Dennis, P., 2005. Livestock grazing affects the egg size of an insectivorous passerine. *Biology Letters 1*, 322-325.

Fales, S.L., Muller, L.D., Ford, S.A., O'Sullivan, M., Hoover, R.J., Holden, L.A., Lanyon, L.E., Buckmaster, D.R., 1995. Stocking rate affects production and profitability in a rotationally grazed pasture system. *Journal of Production Agriculture 8*(1), 88-96.

FAO, 2019. Key findings and findings. [Online]. Available at: http://www.fao.org/news/story/en/item/197623/icode/. Accessed 16th January 2020.

FAO, 2009. Global agriculture towards 2050. [Online]. Available at http://www.fao.org/fileadmin/templates/wsfs/docs/Issues_papers/HLEF2050_Global_Agriculture.pdf. Accessed 19th August 2019.

Farming Connect, 2013. Grazing Systems – Fact Sheet. [Online]. Available at: http://www.grassdevcentre.co.uk/assets/2013-grazing-systems.pdf. Accessed 20th August 2019.

Ferrer-Cazcarra, R., Petit, M., 1995. The influence of animal age and swards height on the herbage intake and grazing behaviour of Charolais cattle. *Animal Science 61*, 497-506.

Font-i-Furnols, M., Guerrero, L., 2014. Consumer preference, behavior and perception about meat and meat products: An overview. *Meat Science 98*(3), 361-371.

Frame, J., Laidlaw, A.S., 2011. Improved Grassland Management. (2nd Edition). Crowood Press.

Francis, C., Lieblein, G., Gliessman, S., Breland, T.A., Creamer, N., Harwood, R., Salomonsson, L., Helenius, J., Rickerl, D., Salvador, R., Wiedenhoeft, M., 2003. Agroecology: the ecology of food systems. *Journal of Sustainable Agriculture 22*(3), 99-118.

Fraser, M.D., Moorby, J.M., Vale, J.E., Evans, D.M., 2014a. Mixed grazing systems benefit both upland biodiversity and livestock production. *Public Library of Science One 9*(2), e89054.

Fraser, M.D., Fleming, H.R., Moorby, J.M., 2014b. Traditional vs modern: role of breed type in determining enteric methane emissions from cattle grazing as part of contrasting grassland-based systems. *Public Library of Science One 9*(9), e107861.

Fraser, M.D., Theobald, V.J., Dhanoa, M.S., Davies, O.D., 2011. Impact on sward composition and stock performance of grazing Molinia-dominant grassland. *Agriculture, Ecosystems and Environment 144*(1), 102-106.

Fraser, M.D., Theobald, V.J., Griffiths, J.B., Morris, S.M., Moorby, J.M., 2009. Comparative diet selection by cattle and sheep grazing two contrasting heathland communities. *Agriculture, Ecosystems and Environment 129*(1), 182-192.

Freitas, T.P.T., Keuroghlian, A., Eaton, D.P., de Freitas, E.B., Figueiredo, A., Nakazato, L., de Oliveira, J.M., Miranda, F., Paes, R.C.S., Monteiro, L.A.R.C., Lima, J.V.B., Neto, A.A.C., Dutra, V., de Freitas, J.C., 2010. Prevalence of *Leptospira interrogans* antibodies in free-ranging *Tayassu pecari* of the Southern Pantanal, Brazil, an ecosystem where wildlife and cattle interact. *Tropical Animal Health and Production 42,* 1695-1703.

Fuller, R., 1988. Suckled calf production. Chalcombe Publications.

Garner, F.H., 1944. The cattle of Britain. London, New York etc., Longmans, Green and Co.

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

Gerber, P.J., Steinfeld, H., Henderson, B., Mottet, A., Opio, C., Dijkman, J., Falcucci, A., Tempio, G., 2013. Tackling Climate Change through Livestock: A Global Assessment of Emissions and Mitigation Opportunities. Food and Agriculture Organization of the United Nations (FAO), Rome.

Gillet, F., Kohler, F., Vandenberghe, C., Buttler, A., 2010. Effect of dung deposition on small-scale patch structure and seasonal vegetation dynamics in mountain pastures. *Agriculture, Ecosystems and Environment 135*(1), 34-41.

Godfray, H.C.J., Garnett, T., 2014. Food security and sustainable intensification. *Philosophical transactions of the Royal Society of London: Series B, Biological Sciences 369*(1639), 20120273.

Godfray, H.C.J., Beddington, J. R., Crute, I. R., Haddad, L., Lawrence, D., Muir, J. F., Pretty, J., Robinson, S., Thomas, S.M., Toulmin, C., 2010. Food security: the challenge of feeding 9 billion people. *Science 327*, 812–818.

Grant, S.A., Torvell, L., Sim, E.M., Small, J.L., Armstrong, R.H., 1996. Controlled grazing studies on Nardus grassland: effects of between-tussock sward height and species of grazer on Nardus utilization and floristic composition in two fields in Scotland. *Journal of Applied Ecology 33,* 1053-1064.

Grant, S.A., Suckling, D.E., Smith, H.K., Torvell, L., Forbes, T.D.A., Hodgson, J., 1985. Comparative studies of diet selection by sheep and cattle: the hill grasslands. *Journal of Ecology 73,* 987-1004.

Grant, S.A., Lamb, W.I.C., Kerr, C.D., Bolton, G.R., 1976. The utilisation of blanket bog vegetation by grazing sheep. *Journal of Ecology 13,* 857-869.

Grant, S.A., 1968. Heather regeneration following burning: a survey. *Grass and Forage Science 23*(1), 26-32.

Grant, S.A., Hunter, R.F., 1966. The effects of frequency and season of clipping on the morphology, productivity and chemical composition of *Calluna vulgaris* (L.) Hull. *New Phytologist 65*, 125-133.

Great Britain Parliament, 2011. Farming in the Uplands. Third Report of Session 2010-11, Volume I. House of Commons, Environment, Food and Rural Affairs Committee.

Habib, G., 2004. Mulberry-Fruit-Based Feed Blocks: A key supplement for livestock in Mountainous Regions. *Mountain Research and Development 24*(2), 106-109.

Hartley, S.E., Mitchell, R.J., 2005. Manipulation of nutrients and dominance and consequences for community composition. *Journal of Ecology 93*, 990-1004.

Heal, W., 2003. Introduction, context, and seminar conclusions. In: Managing Upland Catchments: Priorities for Water and Habitat Conservation. Joint Nature Conservation Committee, Durham.

Henchion, M., McCarthy, M., Resconi, V.C. and Troy, D., 2014. Meat consumption: Trends and quality matters. *Meat Science 98*(3), 561-568.

Henle, K., Alard, D., Clitherow, J., Cobb, P., Firbank, L., Kull, T., McCracken, D., Moritz, R.F., Niemelä, J., Rebane, M., Wascher, D., 2008. Identifying and managing the conflicts between agriculture and biodiversity conservation in Europe–A review. *Agriculture, Ecosystems and Environment 124*(1-2), 60-71.

Hinton, D.G., 2007. Supplementary feeding of sheep and beef cattle. 2nd Edition. Landlinks Press.

Hodge, I., Hauck, J., Bonn, A., 2015. The alignment of agricultural and nature conservation policies in the European Union. *Conservation Biology 29,* 996-1005.

Hoffman, I., 2010. Climate change and the characterization, breeding and conservation of animal genetic resources. *Animal Genetics 41* (1), 32-46.

Hubacek, K., Dehnen-Schmutz, K., Qasim, M., Termansen, M., 2009. Description of the upland economy: areas of outstanding beauty and marginal economic performance. In: Bonn, A., Allott, T., Hubacek, K., Stewart, J. (Eds.), Drivers of Environmental Change in Uplands. Routledge, Abingdon.

Hunt, J., 2011. Beef and sheep upland producers can boost grassland management. Farmers Weekly, *156*(2), 36.

Hyland, J.J., Jones, D.L., Parkhill, K.A., Barnes, A.P., Williams, A.P., 2016a. Farmers' perceptions of climate change: identifying types. *Agriculture and Human Values 33*(2), 323-339.

Hyland, J.J., Styles, D., Jones, D.L., Williams, A.P., 2016b. Improving livestock production efficiencies presents a major opportunity to reduce sectoral greenhouse gas emissions. *Agricultural Systems 147*, 123-131.

IGER Grassland Development Centre, not dated. Silage Decisions Factsheet – How Grass Grows. [Online]. Available at: http://www.britishgrassland.com/system/files/how%20grass%20grows%20final%202.pdf. Accessed 20th August 2019.

International Committee for Animal Recording, 2015. Conformation Recording Dairy and Beef Cattle. ICAR Conformation Working Group. [Online]. Available at: https://www.icar.org/wp-content/uploads/2015/08/Conformation-Recording-CR-WG.pdf. Accessed 19th April 2020.

Issanchou, S., 1996. Consumer expectations and perceptions of meat and meat product quality. *Meat Science 43,* 5-19.

Isselstein, J., Griffith, B.A., Pradel, P., Venerus, S., 2007. Effects of livestock breed and grazing intensity on biodiversity and production in grazing systems. 1. Nutritive value of herbage and livestock performance. *Grass and Forage Science 62*(2), 145-158.

Johnson, J.M.F., Franzluebbers, A.J., Weyers, S.L. and Reicosky, D.C., 2007. Agricultural opportunities to mitigate greenhouse gas emissions. *Environmental Pollution 150*(1), 107-124.

Johnson, K.A., Johnson, D.E., 1995. Methane emissions from cattle. *Journal of Animal Science 73*(8), 2483-2492.

Johnstone, I.G., Bladwell, S., Noble, D.G., 2008. The State of Birds in Wales 5: a review of 2007. RSPB Cymru, Cardiff. [Online]. Available at: http://www.assembly.wales/NAfW%20Documents/6_rspb_formatted.pdf%20-%2015102009/6_rspb_formatted-English.pdf. Accessed 20th August 2019.

Jones, A., Jones, D., Edwards-Jones, G., Cross, P., 2013. Informing decision making in agricultural greenhouse gas mitigation policy: a best-worst scaling survey of expert and farmer opinion in the sheep industry. *Environmental Science and Policy 29*, 46-56.

Kantar Worldpanel, 2018. Consumer Insights, *Age 18*, 24.

Kantar Worldpanel, 2017. Veganism, flexitarianism, and the power of influencers. [Online]. Available at: https://uk.kantar.com/consumer/green/2017/veganism/. Accessed 16th January 2020.

Kim, S., Gillespie, J.M., Paudel, K.P., 2008. Rotational grazing adoption in cattle production under a cost-share agreement: does uncertainty have a role in conservation technology adoption? *Australian Journal of Agricultural and Resource Economics 52*(3), 235-252.

Kotschi, J., 2013. A soiled reputation: Adverse impacts of mineral fertilizers in tropical agriculture; Fertilization, Conservation of Resources, Food Security; Study. Heinrich-Böll-Stiftung.

Kuit, G., van der Meulen, H.S., 1999. Beef from nature reserves: a market view. In: Laker, J.P., Milne, J.A. (Eds.), Livestock Production in the European LFAs: Meeting Future Economic, Environmental and Policy Objectives Through Integrated Research, Proceedings of the 2nd International Conference of the LSIRD Network, Bray, Dublin, Ireland. Macaulay Land Use Research Institute, Aberdeen, 145-154.

Laser, H., 2007. Effects of liming and nitrogen application on the trace element concentrations of pastures in low mountain range. *Plant Soil and Environment 53*(6), 258.

Lawrence, A.B., Wood-Gush, D.G.M., 1988. Home range behaviour and social organisation of Scottish blackface sheep. *Journal of Applied Ecology 25*, 25-40.

Limb, R.F., Fuhlendorf, S.D., Engle, D.M., Weir, J.R., Elmore, R.D., Bidwell, T.G., 2011. Pyric–herbivory and cattle performance in grassland ecosystems. *Rangeland Ecology and Management 64*(6), 659-663.

Mansfield, L., 2011. Upland Agriculture and the Environment. Badger Press Ltd.

Martin, C., Morgavi, D.P., Doreau, M., 2010. Methane mitigation in ruminants: from microbe to the farm scale. *Animal 4*(3), 351-365.

Mathijs, E., 2015. Exploring future patterns of meat consumption. *Meat Science 109*, 112-116.

McDonald, P., Edwards, R.A., Greenhalgh, J.F.D, Morgan, C.A., 2002. Animal Nutrition, 6th Edition. Longman, London and New York.

McGinty, E., Baldwin, B., Banner, R., 2009. A review of livestock grazing and range management in Utah. State of Utah, Governor's Public Lands Policy Coordination Office. Logan: Utah State University.

Midmore, P., Moore-Colyer, R.J., 2005. Cherished Heartland: Future of the Uplands in Wales. Institute of Welsh Affairs, Cardiff.

Millenium Ecosystem Assessment, 2005. Millenium Ecosystem Assessment. Ecosystems and Human Well Being: Synthesis report. Island Press, Washington, DC, USA.

Milne, J.A., Osoro, K., 1997. The role of livestock in habitat management. In: Laker, J.P., Mile, J.A. (Eds.), *Livestock Systems in European Rural Development,* Proceedings of the 1st Conference of the LSIRD network (Nafplio, Greece). Macaulay Land Use Research Institute, Aberdeen, 75-80.

Mitchell, R.J., Rose, R.J., Palmer, S.C., 2008. Restoration of Calluna vulgaris on grass-dominated moorlands: the importance of disturbance, grazing and seeding. *Biological Conservation 141*(8), 2100-2111.

Moore, K.J., Moser, L.E., Vogel, K.P., Waller, S.S., Johnson, B.E., Pedersen, J.F., 1991. Describing and quantifying growth stages of perennial forage grasses. *Agronomy Journal 83*(6), 1073-1077.

Morgan-Davies, C., Wilson, R., Waterhouse, T., 2017. Impacts of farmers' management styles on income and labour under alternative extensive land use scenarios. *Agricultural Systems 155,* 168-178.

Morgan-Davies, C., Wilson, R., Waterhouse, T., 2015. Use or Delight? History of conflicting hill land uses in Scotland – a review. *Scottish Geographical Journal 131*(2), 98-122.

Morgan-Davies, J., Morgan-Davies, C., Pollock, M.L., Holland, J.P., Waterhouse, A., 2014. Characterisation of extensive beef cattle systems: Disparities between opinions, practice and policy. *Land Use Policy 38*, 707-718.

Morgan-Davies, C., Waterhouse, T., Umstatter, C., 2006. Sustainable hill and upland systems: what do people want the hills to deliver?. *The International Journal of Biodiversity Science and Management 2*(3), 242-244.

Morris, W., Henley, A., Dowell, D., 2017. Farm diversification, entrepreneurship and technology adoption: Analysis of upland farmers in Wales. *Journal of Rural Studies 53*, 132-143.

Mowle, A., Bell, M., 1988. Rural policy factors in land-use change. *Ecological change in the uplands*, 165-182.

Nelson, G.C., Rosegrant, M.W., Palazzo, A., Gray, I., Ingersoll, C., Robertson, R., Tokgoz, S., Zhu, T., Sulser, T.B., Ringler, C., Msangi, S., 2010. Food Security, Farming, and Climate Change to 2050: Scenarios, Results, Policy Options 172. International Food Policy Research Institute.

Nguyen, T.L.T., Hermansen, J.E., Mogensen, L., 2010. Environmental consequences of different beef production systems in the EU. *Journal of Cleaner Production 18*(8), 756-766.

Nolan, T., Connolly, J., 1977. Mixed stocking by sheep and steers – a review. *Herbage Abstracts 47*, 367-374.

Orr, R.J., Rutter, S.M., Penning, P.D., Rook, A.J., 2001. Matching grass supply to grazing patterns for dairy cows. *Grass and Forage Science 56*, 352-361.

Parsons, A.J., Johnson, I.R., 1986. The physiology of grass growth under grazing. In: Grazing. Frame, J. (ed). *Occ. Symp. British Grassland Society 19*, 3-13.

Perry, P.J., 1974. British Farming in the Great Depression 1970 to 1914: an historical geography. David and Charles: Newton Abbot.

Petersen, B., Snapp, S., 2015. What is sustainable intensification? Views from experts. *Land Use Policy 46*, 1-10.

Peyraud, J.L., Mosquera-Losada, R., Delaby, L., 2004. Challenges and tools to develop efficient dairy systems based on grazing: how to meet animal performance and grazing management. *Grassland Science in Europe 9*, 373-384.

Phelan, P., Moloney, A.P., McGeough, E.J., Humphreys, J., Bertilsson, J., O'Riordan, E.G., O'Kiely, P., 2015. Forage legumes for grazing and conserving in ruminant production systems. *Critical Reviews in Plant Sciences 34*(1-3), 281-326.

Pinares-Patiño, C.S., D'hour, P., Jouany, J.P., Martin, C., 2007. Effects of stocking rate on methane and carbon dioxide emissions from grazing cattle. *Agriculture, Ecosystems and Environment 121*(1), 30-46.

Pretty, J., 2008, Agricultural sustainability: concepts, principles and evidence. *Philosophical Transactions of the Royal Society of London Series B 363*(1491), 447–466.

Ratcliffe, D.A., Thompson, D.B.A., 1988. The British uplands their ecological character and international significance in Ecological Change in the Uplands (M.B. Usher and D.B.A. Thompson, eds), 9-36, Blackwell Scientific Publications, Oxford.

Reed, M.S., Arblaster, K., Bullock, C., Burton, R.J.F., Davies, A.L., Holden, J., Hubacek, K., May, R., Mitchley, J., Morris, J., Nainggolan, D., 2009a. Using scenarios to explore UK upland futures. *Futures 41*(9), 619-630.

Reed, M.S., Bonn, A., Slee, W., Beharry-Borg, N., Birch, J., Brown, I., Burt, T.P., Chapman, D., Chapman, P.J., Clay, G.D., Cornell, S.J., 2009b. The future of the uplands. *Land Use Policy 26*, S204-S216.

Resurreccion, A.V.A., 2003. Sensory aspects of consumer choices for meat and meat products. *Meat Science 66*(1), 11-20.

Rojas-Downing, M.M., Nejadhashemi, A.P., Harrigan, T., Woznicki, S.A., 2017. Climate change and livestock: Impacts, adaptation, and mitigation. *Climate Risk Management 16*, 145-163.

Royal Society, 2009, Reaping the Benefits: Science and the Sustainable Intensification of Global Agriculture, Royal Society, London.

Rook, A.J., Dumont, B., Isselstein, J., Osoro, K., WallisDeVries, M.F., Parente, G., Mills, J., 2004. Matching type of livestock to desired biodiversity outcomes in pastures–a review. *Biological Conservation 119*(2), 137-150.

Rutter, S.M., Orr, R.J., Yarrow, N.H., Champion, R.A., 2004a. Dietary preference of dairy cows grazing ryegrass and white clover. *Journal of Dairy Science 87*, 1317–1324.

Rutter, S.M., Orr, R.J., Yarrow, N.H., Champion, R.A., 2004b. Dietary preference of dairy heifers grazing ryegrass and white clover, with and without an anti-bloat treatment. *Applied Animal Behaviour Science 85*, 1–10.

Rutter, S.M., Orr, R.J., Rook, A.J., 2000. Dietary preference for grass and white clover in sheep and cattle: an overview. In: Rook, A.J., Penning, P.D. (Eds.), Grazing Management, the Principles and Practice of Grazing, for Profit and Environmental Gain, Within Temperate Grassland Systems. British Grassland Society, Reading, 73-78.

SAC Consulting, 2014. A Review of the Beef Sector in Wales. [Online]. Available at: https://meatpromotion.wales/images/publications/A_review_of_the_Beef_Sector_in_Wales.pdf. Accessed 16th January 2020.

Scollan, N., Hocquette, J.F., Nuernberg, K., Dannenberger, D., Richardson, I., Moloney, A., 2006. Innovations in beef production systems that enhance the nutritional and health value of beef lipids and their relationship with meat quality. *Meat Science 74*(1), 17-33.

Scottish Natural Heritage, 2002. Natural Heritage Futures: an overview. SNH: Battleby, Perth.

Šimek, M., Brůček, P., Hynšt, J., Uhlířová, E., Petersen, S.O., 2006. Effects of excretal returns and soil compaction on nitrous oxide emissions from a cattle overwintering area. *Agriculture, Ecosystems and Environment 112*(2), 186-191.

Simmons, I.G., 2002. " a grey melancholy hill..." Ten thousand years of environmental history. *Report-Joint Nature Conservation Committee 319*, 26-31.

Smithson, P., Addison, K., Atkinson, K., 2002. Fundamentals of the Physical Environment. Routledge: London.

Sonoda, Y., Oishi, K., Chomei, Y., Hirooka, H., 2018. How do human values influence the beef preferences of consumer segments regarding animal welfare and environmentally friendly production?. *Meat Science 146*, 75-86.

Soteriades, A.D., Foskolos, A., Styles, D., Gibbons, J.M., 2019. Diversification not specialization reduces global and local environmental burdens from livestock production. *Environment International 132*, 104837.

Soteriades, A.D., Gonzalez-Mejia, A.M., Styles, D., Foskolos, A., Moorby, J.M., Gibbons, J.M., 2018. Effects of high-sugar grasses and improved manure management on the environmental footprint of milk production at the farm level. *Journal of Cleaner Production 202*, 1241-1252.

Staerfl, S.M., Amelchanka, S.L., Kälber, T., Soliva, C.R., Kreuzer, M., Zeitz, J.O., 2012. Effect of feeding dried high-sugar ryegrass ('AberMagic') on methane and urinary nitrogen emissions of primiparous cows. *Livestock Science 150*(1), 293-301.

Stapledon, 1937. The Hill Lands of Britain. Faber and Faber: London.

Stejskalová, M., Hejcmanová, P., Pavlů, V., Hejcman, M., 2013. Grazing behavior and performance of beef cattle as a function of sward structure and herbage quality under rotational and continuous stocking on species-rich upland pasture. *Animal Science Journal 84*(8), 622-629.

Swinbank, A., Daugbjerg, C., 2006. The 2003 CAP reform: accommodating WTO pressures. *Comparative European Politics 4*(1), 47-64.

Talbot, M., 2013. Farm tourism in Wales. Products and markets, resources and capabilities. The experience of six farm tourism operators. *European Countryside 5*(4), 275-294.

Thompson, D.B.A, MacDonald, A.J., Marsden, J.H., Galbraith, C.A., 1995. Upland heather moorland in Great Britain: a review of international importance, vegetation change and some objectives for nature conservation. *Biological Conservation 71*, 163–178.

Tolhurst, S., Oates, M., 2001. The Breed Profiles Handbook, English Nature, Peterborough, UK.

Tronstad, R., Teegerstrom, T., 2001. Managing Herd Composition Of Range Cattle: Sale Weight And Seasonal Factors. In: 2001 Annual Meeting, July 8-11, 2001, Logan, Utah (36058). Western Agricultural Economics Association.

UK National Ecosystem Assessment, 2011. The UK National Ecosystem Assessment Technical Report. UNEP-WCMC, Cambridge.

Usher, M.B., Thompson, D.B.A., 1988. Ecological Change in the Uplands. Blackwell Scientific Publications. Oxford, UK.

Vallentine, J.F., 2000. Grazing management. Academic, San Diego, California, USA.

Van Zanten, H.H.E., Mollenhorst, H., Klootwijk, C.W., van Middelaar, C.A., de Boer, I.J.M., 2016. Global food supply: land use efficiency of livestock systems. *International Journal of Life Cycle Assessment 21,* 747-758.

Verbeke, W., Van Wezemael, L., de Barcellos, M.D., Kügler, J.O., Hocquette, J.F., Ueland, Ø., Grunert, K.G., 2010. European beef consumers' interest in a beef eating-quality guarantee: insights from a qualitative study in four EU countries. *Appetite 54*(2), 289-296.

Vulink, J.T., 2001. Hungry herds: Management of temperate lowland wetlands by grazing. Doctoral Thesis, Van Zee tot Land 66, Ministerie van Verkeer en Waterstaat, Lelystad.

Webb, J., Audsley, E., Williams, A., Pearn, K., Chatterton, J., 2014. Can UK livestock production be configured to maintain production while meeting targets to reduce emissions of greenhouse gases and ammonia? *Journal of Cleaner Production 83*, 204-211.

Webster, J., 2011. Management and Welfare of Farm Animals - The UFAW Farm Handbook. 5th edition. John Wiley and Sons.

Welch, D., 1986. Studies in the grazing of heather moorland in north-east Scotland. V. Trends in *Nardus stricta* and other unpalatable graminoids. *Journal of Applied Ecology 23*, 1047-1058.

Welsh Government, 2018. Glastir Advanced 2018. [Online]. Available at: https://gov.wales/sites/default/files/publications/2018-01/glastir-advanced-2018-rules-booklet-2-whole-farm-code-and-management-options.pdf. Accessed 20th April, 2020.

Welsh Government, 2019a. Forecasts of farm incomes in Wales, 2018-19. [Online]. Available at: https://gov.wales/sites/default/files/statistics-and-research/2019-05/farm-incomes-april-2018-to-march-2019-forecasts-938.pdf. Accessed 28th January 2020.

Welsh Government, 2019b. Sustainable Farming and our Land. [Online]. Available at: https://gov.wales/sites/default/files/consultations/2019-07/brexit-consultation-document.pdf. Accessed 28th January 2020.

Welsh Government, 2016. June 2016 Survey of Agriculture and Horticulture: Results for Wales. [Online]. Available at: https://gweddill.gov.wales/docs/statistics/2016/161124-survey-agriculture-horticulture-june-2016-en.pdf. Accessed 19th August 2019.

Whitty, J.L., 2003. For the Department for Environment, Food and Rural Affairs. Hansard, 27 Jan 2003, Column WA137, author's emphasis. [Online]. Available at: https://www.publications.parliament.uk/pa/ld200203/ldhansrd/vo030127/text/30127w03.htm. Accessed 20th August 2019.

Wilkins, R.J., Garwood, E.A., 1986. Effects of treading, poaching and fouling on grassland production and utilization. In *Grazing. Occasional Symposium 19,* 19-31.

Winter, M., Gaskell, P., Short, C., 1998. Upland landscapes in Britain and the 1992 CAP reforms. *Landscape Research 23*(2), 273-288.

Wright, I.A., Jones, J.R., Davies, D.A., Davidson, G.C., Vale, J.E., 2006. The effect of sward surface height on the response to mixed grazing by cattle and sheep. *Animal Science 62*, 271-276.

Yarwood, R., Evans, N., 1999. The changing geography of rare livestock breeds in Britain. *Geography 84*, 80-87.

Zachos, F.E., Habel, J.C. eds., 2011. Biodiversity hotspots: distribution and protection of conservation priority areas. Springer Science and Business Media.

3. The environmental cost of increasing upland pasture productivity

Abstract

The environmental impacts of increasing pasture productivity in the uplands are under-researched. A field trial on a representative upland farm investigated the effect of implementing pasture improvement options on previously unimproved land between 2017 and 2019. The pasture improvement options were cultivation (ploughing and rotovating) and reseeding in combination with lime and fertiliser inputs. Soil pH, grass growth, pasture quality and nitrous oxide (N_2O) emissions from the soil were measured throughout the whole growing season. Carbon dioxide (CO_2) emissions were also measured from cultivation until pasture establishment in 2017. Our findings indicate that, in the year of establishment, reseeding and increased nitrogen (N) fertiliser application led to higher daily grass growth, with the mean over three times higher from the ploughed and reseeded treatment (20.0 kg DM ha^{-1} day^{-1}) as opposed to the control (8.2 kg DM ha^{-1} day^{-1}) during the growing season ($p < 0.05$). Yield effects in subsequent years were dependent upon nutrient input regimes. Despite a significant difference in pasture energy content between the treatments in the year of establishment, there was no effect of reseeding and targeted lime and fertiliser applications on pasture quality in the two subsequent years. Rotovating and ploughing also resulted in increased N_2O emission from the soil, with higher N fertiliser emission factors in 2017 (2.26% ± 0.79 from the rotovated treatment) compared with the IPCC default value of 1.6%. However, low N_2O fluxes in 2018 meant that the emission factors during this growing season were also low (highest value of 0.25% ± 0.04 from the lime and fertiliser input only treatment). Rotovating and ploughing did not significantly impact CO_2 emissions in the establishment period. From a production perspective, the results would favour reseeding upland pastures. However, yield-scaled N_2O emission calculations reported that reseeding was not the most effective method in maximising grass yield while minimising N_2O emissions, and in fact, applying lime and fertiliser to upland permanent pasture resulted in the lowest N_2O emissions per unit of grass produced (0.07 g N_2O kg^{-1} DM of grass). These findings demonstrate the importance of considering both pasture production and N_2O emissions side-by-side when improving upland pasture for maximum nitrogen use efficiency and to reduce negative environmental impacts.

3.1 Introduction

Approximately 42% of the United Kingdom's agricultural land is classified as 'upland' (Reed et al., 2009). Historically, most of this land has been considered as productively poor pasture, being the nation's most extensive semi-natural ecosystem (Ratcliffe, 1977). Prior to the 19th century, upland agriculture was predominantly extensive (Davies, 2010). Constraints on the productivity of upland grassland swards include climatic conditions, acidic soils and nutrient deficits e.g. low N and phosphorus (P) availability restricting plant growth. The introduction of grants to specifically improve upland pasture following the Second World War led to the adoption of practices such as liming to increase soil pH and fertiliser applications (Frame et al., 1985; Newbould, 1974; 1975). These grants led to an increase in pasture productivity over time, particularly in the 1970s and 1980s (Cuttle and James, 1995).

Despite this management change, only a third of grazed uplands in the UK are regarded as improved pasture (Fraser, 2008), while the value for Wales is slightly lower at 26% (Edwards et al., 2007). Although there are no recent data on these land-use proportions, we would not expect much to have changed in recent years. Upland soils remain generally acidic as a result of wetter climates in these areas (Mansfield, 2011). Nonetheless, there is often wide variation in soil pH in the uplands, and plant growth in response to nutrient supply will differ depending on species (Stevens et al., 2016). This can lead to challenges for nutrient planning and application.

Liming soils is an agricultural practice that has been carried out historically in order to improve pasture productivity and quality (Glømme, 1931; Hopkins and Readhimer, 1907). The field operation is implemented to increase soil pH. This can lead to several benefits from a production viewpoint, such as improved soil structure (Paradelo et al., 2015) and an increase in the plant-availability of P and potassium (K) (Karalic et al., 2013), which leads to improved grass sward yield (Mazzetto et al., 2015) and persistence of perennial species (Hayes et al., 2016). Lime is generally applied prior to the use of fertiliser; another form of pasture improvement commonly used in livestock systems to maintain or increase grass yield (AHDB, 2016). Native grass species in upland and rangeland areas have demonstrated a poor response to fertiliser applications on some occasions (Davies and Munro, 1974; Scott et al., 1996). Some suggest that past management practices such as overgrazing has led to some upland habitats already having crossed critical production thresholds, and possibly unable to respond as anticipated to management practices such as nutrient inputs (Davies, 2010). This implies that applying fertiliser in semi-natural ecosystems such as unimproved uplands may be a waste of resources, but there is only a small body of recent literature to support this.

Nitrogen fertiliser is applied to supplement the lack of soil N supply to meet crop demand (AHDB, 2017b). It can often influence pasture quality, by increasing crude protein content (Keady et al., 2000). Results from a long-term experiment on well-fertilised, semi-permanent upland pasture determined that reducing fertiliser inputs over time led to a decline in sown grass species in upland pasture, with these species replaced by less productive weed grasses (Yu et al., 2011). A reduction in grass production potential is a consequence of this. The study by Yu et al. (2011) did not take into account uplands that have received very little, or no inputs in the past. Extrapolating this finding to determine the effect of reduced N application on unimproved upland is therefore challenging.

In general, measures designed to improve pasture productivity may have environmental costs. For instance, of the little available research, liming has shown to contribute to greenhouse gas emissions from agricultural soils due to increased CO_2 emissions as dissolution occurs (IPCC, 2006b; West and McBride, 2005). Furthermore, inefficient application of N fertiliser leads to increased secondary environmental burdens such as nutrient leaching, causing run-off and eutrophication in watercourses (Rütting et al., 2018). It has been widely recognised that N fertiliser inputs are a major driver of N_2O emissions from pasture, whereby surplus N is vulnerable to be released as N_2O gas (Cardenas et al., 2019; Poulton et al., 2018). The default emission factor (EF) value for N_2O emissions was stated as 1% of N applied as fertiliser for direct emissions from managed agricultural soils (IPCC, 2006a). However, recent efforts to revise this default EF in order to take climate and fertiliser type into account has led to an amended value of 1.6% for synthetic N fertiliser applied to managed agricultural soils in wet climates (IPCC, 2019). The UK, along with several other countries have developed country specific N_2O EFs for urea and ammonium nitrate (AN) fertiliser applied to grassland, yet these new country specific N_2O EFs are 'lowland-centric'. Few studies have attempted to calculate the N_2O EF following N fertiliser input to previously unfertilised upland soils, and yet recent research suggests that the low soil pH of such soils may be the reason for low N_2O fluxes following N inputs (Marsden et al., 2019), meaning that it is challenging to test the effectiveness of applying the default value to various sites that differ in environmental factors and land management.

It is often suggested that reseeding is a necessary requirement in order to change pasture composition and maintain, or improve, grassland productivity (Bertora et al., 2007; Rushton et al., 1989). Despite this, past studies have not compared the effectiveness of tillage practices such as ploughing and rotovating on seed establishment. Reseeding is not always a practical operation in the uplands due to the physical features (e.g. steep terrain and remoteness) restricting accessibility (Fraser et al., 2014). In addition to this, the costs of establishment often outweigh the increase in productivity achieved (Mansfield, 2011). Previous research has confirmed that ploughing grassland for conversion to grassland renovation (or another use, e.g. arable land) is responsible for a substantial amount of

N_2O and CO_2 emissions in the Netherlands (Vellinga et al., 2004). However, to our knowledge, direct comparisons of the environmental impact in terms of N_2O emission and carbon (C) loss as CO_2 emission from upland agricultural soils following ploughing and rotovating are very limited.

Whilst not necessarily suitable in every case, incorporation of new grass varieties to permanent pastures is increasingly recommended (AHDB, 2018). However, only a few species are considered suitable for uplands such as red fescue (*Festuca rubra*) and perennial ryegrass (*Lolium perenne*). Seed establishment often fails following reseeding with these species in the uplands due to climatic conditions such as high rainfall, leading to only transient sward improvement (Davies et al., 1984). At the beginning of the 20th century, grass species bred for highly productive and fertile lowland sites were targeted at upland pastures. The need to develop new grass varieties as a means of increasing pasture productivity specifically for marginal uplands was proposed over many decades (Hughes and Munro, 1962). Although the development was an important improvement upon grass varieties bred solely for lowland use, there was little breeding effort during the 20th century. There has been considerable effort by plant breeders in the 21st century to develop grass varieties to increase nitrogen use efficiency (NUE) as well as long-term upland mixtures. Carswell et al. (2019) showed that NUE by high-sugar grasses notably varied across different management sites within the UK. This reiterates the need for upland-specific studies, as limited data exist to determine the effect of physical constraints on plant NUE on marginal land. Evans et al. (2011) found that the inclusion of AberDart, a ryegrass bred for elevated water-soluble carbohydrate concentrations (typically known as "high-sugar grasses") in upland pasture led to increased digestibility of the pasture. This could lead to improved N utilisation in the rumen, and therefore increased efficiency of upland livestock production systems.

Another approach to increase herbage productivity whilst concomitantly reducing atmospheric N is the incorporation of N-fixing legumes in pastures (Peyraud et al., 2009). Forage legumes such as red clover (*Trifolium pratense*) and white clover (*Trifolium repens*) have the ability to reduce N fertiliser requirements and losses while improving grass utilisation (Edwards et al., 2007; Phelan et al., 2015). Such benefits include high N fixation as well as increased legume yield (Carlsson and Huss-Danell, 2003). However, Yu et al. (2011) reported low white clover contents in the absence of N fertiliser in upland pasture, possibly due to competition from other species such as creeping bent grass (*Agrostis stolonifera)*, restricting the growth of clover (Foster and Gross, 1998; Schulte and Neuteboom, 2002). This contradicts the concept of including legumes in pastures to reduce N fertiliser use.

While lime and fertiliser application increase upland pasture production, there is a need to investigate both environmental and economic impacts side-by-side in relation to the uplands. Such analyses are important in light of the considerable attention to the uncertainty revolving around UK uplands in the post-Brexit era, i.e. whether they should be managed for food (livestock) production, or for alternative means (e.g. afforestation). Considering the limited evidence comparing methods of improving the productivity of upland pastures, in this study we conducted a simulated grazing experiment with regular cutting to 1) compare the effectiveness of various field operations on increasing pasture productivity, and 2) estimate the environmental costs of the different field operations over repeated years. The results of this study should help contribute to the discussion on the future of sustainable livestock production in the uplands.

3.2 Materials and methods

3.2.1 Study site - Establishment and set-up

The experiment was conducted at the Henfaes Research Centre, Abergwyngregyn, north Wales, with the study site located at approximately 236 m a.s.l. (53.23°N, 4.01°W) (Figure 3.1). The mean long-term annual rainfall at the site is 1282 mm and time-series data for air and soil temperature are available from the UK Centre for Ecology and Hydrology (2020). Rainfall totals for the growing seasons (May - September) for the experimental years were as follows: 2017 – 654.4 mm, 2018 – 422.9 mm, 2019 – 548.8 mm (Boorman, 2019; personal communication). The soil is classified as an Orthic Podzol, with existing vegetation at the field site classed as the British National Vegetation community (NVC) MG6, *Lolium perenne – Cynosurus cristatus* grassland prior to the experiment. The site was previously grazed by sheep, but stock were excluded for experimental purposes and the study treated as a simulated grazing experiment with regular sward cutting. There have been no interventions (nutrient or lime additions) to the site since the early-1980s.

The site measured approximately 943 m² (23 m × 41 m) and was set out in a randomised complete block design. In the first year of the study (2017), the site consisted of fifteen 9 m × 4 m plots with 1 m buffer between each plot. There were five treatments (Control; Rotovate, spring forage rape followed by autumn grass reseed, lime and fertiliser input; Plough, spring grass reseed, lime and fertiliser input; Rotovate, spring grass reseed, lime and fertiliser input; Lime and fertiliser input only), with three replicates of each (Figures 3.1 and 3.2). Subplots in a split-plot design were set-up in the second year (2018), leading to a total of thirty 4 m × 4 m plots separated by a 1 m buffer for the duration of 2018 and 2019. The treatments are described in Table 3.1.

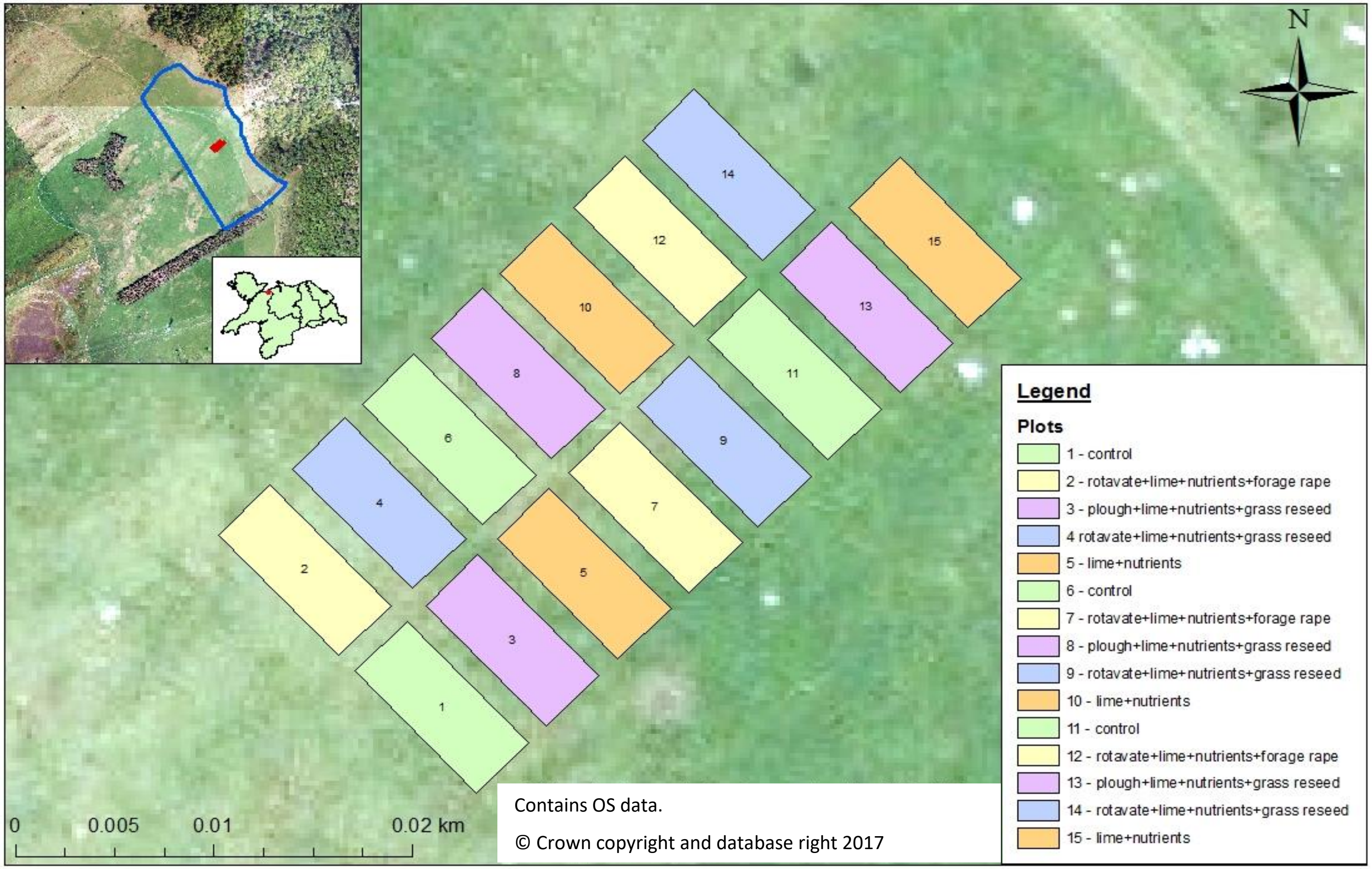

Figure 3.1. Field trial location and set-up. Inset map shows location of the field site at Bangor University's farm, Henfaes Research Centre (53.23°N, 4.01°W).

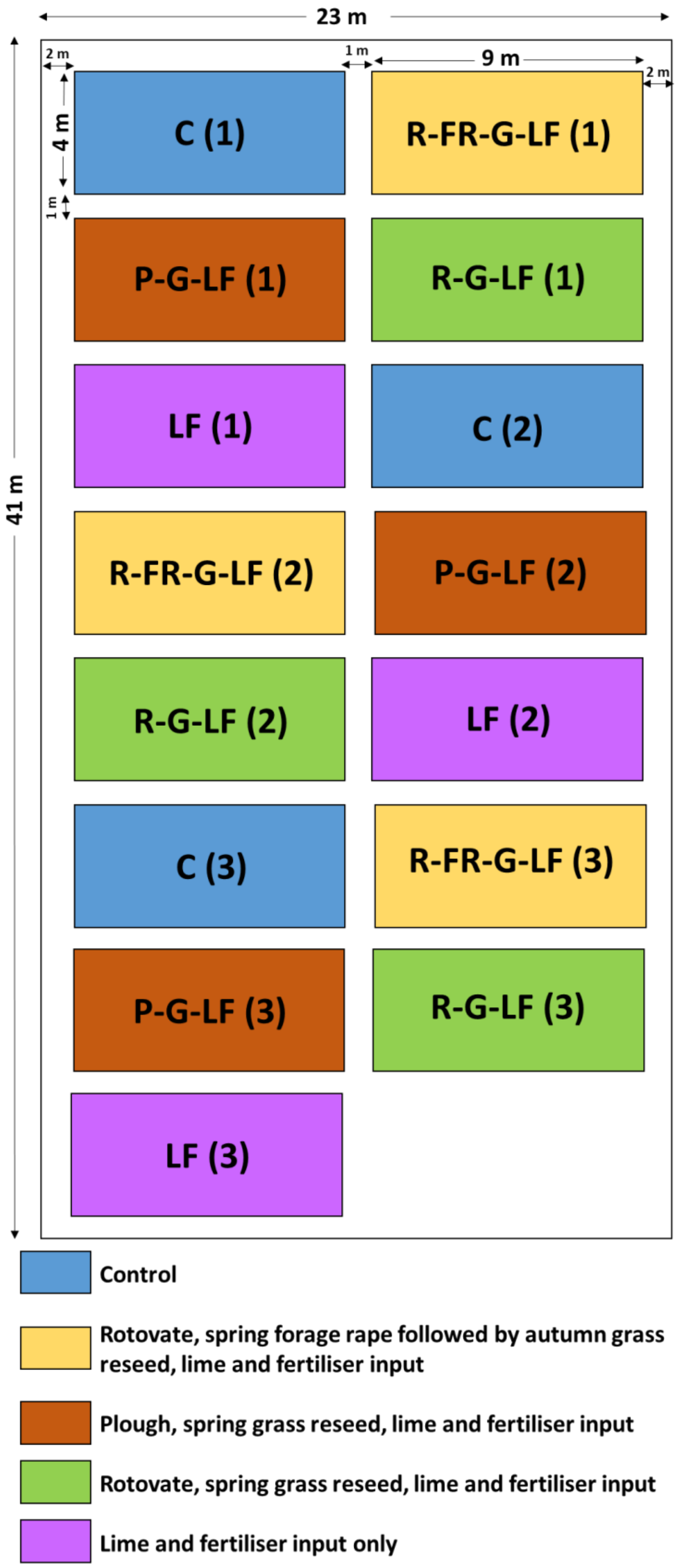

Figure 3.2. Experimental plot layout in 2017. Each plot was divided to subplots in a split-plot design in 2018.

Table 3.1. Nutrient application and management regimes of the different treatments. The range in lime and fertiliser application rates is the replicates' application range for that treatment, based on soil analyses. Fertiliser application rates were determined following soil analyses, in accordance with recommendations in the Nutrient Management Guide (RB209), 8th edition (AHDB, 2017b) for 2017 and split-plots A, and typical application rates in the British Survey of Fertiliser Practice (2017) for split-plots B.

Treatment code	Treatment description	2017					2018				2019			
		Lime application (kg ha^{-1})		Fertiliser application (kg ha^{-1})			Lime application (kg ha^{-1})	Fertiliser application (kg ha^{-1})			Lime application (kg ha^{-1})	Fertiliser application (kg ha^{-1})		
		Spring application	Autumn application	Nitrogen	Phosphorus	Potassium		Nitrogen	Phosphorus	Potassium		Nitrogen	Phosphorus	Potassium
C	Control	-	-	-	-	-	-	-	-	-	-	-	-	-
R FR G	Rotovate, spring forage rape followed by autumn grass reseed, lime and fertiliser input	440 - 995	770 - 940	90	0 – 55	80	375 - 675	A – 130 B - 50	20 - 110	340	495 - 770	A – 130 B - 50	80 - 140	160 - 250
P G	Plough, spring grass reseed, lime and fertiliser input	330 - 1050	330 - 770	60	50 - 80	80	150 - 375	A – 130 B - 50	90 - 120	360-410	440 - 895	A – 130 B - 50	80 - 110	160 - 250
R G	Rotovate, spring grass reseed, lime and fertiliser input	440 - 1050	385 - 770	60	0 - 80	80	75 - 675	A – 130 B - 50	120 - 150	360 - 410	495 - 715	A – 130 B - 50	80 - 140	160 - 250
I	Lime and fertiliser input only	495-885	330-605	60	20	30	0-375	A – 130 B – 50	110 - 140	290 - 340	385 - 715	A – 130 B - 50	95 - 125	170 - 210

3.2.2 Treatment specifications

3.2.2.1 2017

Soil samples were collected from each plot prior to any operations. A minimum of five subsamples were taken from each plot in a 'W' pattern using a soil corer (0 – 10 cm). The subsamples were combined to produce a sample from each plot, which was analysed for pH, P, K, magnesium and sulphur (analysed by NRM Laboratories, Bracknell, Berkshire). Lime and fertiliser applications were determined by the soil analyses in accordance with recommendations noted in the Nutrient Management Guide (RB209), 8th edition (AHDB, 2017b). Fertiliser was applied as Ammonium Nitrate (AN), Triple Super Phosphate (TSP) and Muriate of Potash (MOP) to all treatments at the same time on May 16th, 2017. Granulated Lime, 36% calcium (Calcifert, Runcorn, Cheshire) was applied on May 16th, 2017 for rapid effect. The analyses showed that the plots varied notably in pH (mean pH 5.2 ± 0.12). Therefore, each plot was treated independently with different rates of lime and fertiliser applied, regardless of the treatment implemented. Further soil sampling was conducted on August 30th, 2017, and a second application of lime was applied on October 25th, where needed.

Prior to any field operations, the three treatments that required reseeding were sprayed with Glyphosate (Rosate 36) at a rate of 4 L ha^{-1}. The grass seed mixture sown was 'Lambhill' variety (Limagrain UK Ltd., Rothwell, Lincolnshire), a long-term mixture for marginal land consisting of 66% perennial ryegrass (*Lolium perenne*), 13% timothy (*Phleum pratense*), 7.5% creeping red fescue (*Festuca rubra*), 5.5% white clover (*Trifolium repens*), 5% meadow fescue (*Festuca pratensis*) and 3% alsike clover (*Trifolium hybridum*), sown at a rate of 45 kg ha^{-1}. Where establishment was not achieved due to the dry weather conditions, the plots were resown a month later. For the forage crop treatment, a rape/kale hybrid ('Interval', Limagrain) was sown at 8 kg ha^{-1}. This was sown as a break crop and was harvested on August 23rd, 2017. A subsample of each pasture was extracted for 'wet' Near Infrared analysis of ruminant metabolisable energy (ME) and crude protein content (analysed by Sciantec Analytical Services Ltd., Stockbridge Technology Centre, Cawood). The forage rape treatment plots were reseeded with 'Lambhill' grass mixture on September 26th, 2017.

3.2.2.2 2018 and 2019

As in 2017, soil sampling was conducted prior to lime and fertiliser application. The split-plot design implemented in 2018 was done to investigate the effect of varying fertiliser application rates on the pasture productivity of the five treatments set up in 2017. Nitrogen fertiliser inputs were applied according to the recommended rates in the Nutrient Management Guide (RB209) (AHDB, 2017b) for split-plots A, and typical application rates for upland farms in the British Survey of Fertiliser Practice (2017) for split-plots B. For (A), the N fertiliser inputs were applied as Ammonium Nitrate (AN) in three split doses of 50 kg N ha^{-1}, 40 kg N ha^{-1}, 40 kg N ha^{-1}, and for (B) it was a single application of 50 kg N ha^{-1}.

3.2.3 Measurements

3.2.3.1 Sward quantity and quality

During the growing season, sward height was measured on a weekly basis using a calibrated rising Jenquip EC09 electronic plate meter (Handley Enterprises Ltd) within the plots. Grass samples were obtained for biomass yield by cutting a 1 m × 1 m quadrat from each plot. The timing of collecting grass samples was dependent on grass growth, with samples cut to mimic cattle grazing, thus when the sward was approximately 8–11 cm. This resulted in three separate sampling occasions in 2017, two sampling occasions in 2018 and three sampling occasions in 2019. The samples were then analysed as before, to calculate the crude protein and ME content.

3.2.3.2 Monitoring nitrous oxide and carbon dioxide emissions

A closed static chamber (40 cm × 40 cm × 25 cm) was assigned to each plot, with greenhouse gas samples (20 ml) collected prior to any treatment application (Cardenas et al., 2016). Three time points were taken per chamber (T0 = 0 min, immediately after closing lid of the chamber; T1 = 30 min; T2 = 60 min). The samples were stored in pre-evacuated 20 ml glass vials. The initial gas samples provided a baseline flux before treatments were introduced. Sampling frequency was dependent on field operations timings. The samples were analysed for N_2O emissions using a Varian 450 Gas Chromatograph, with the N_2O and CO_2 fluxes calculated by assuming a linear interpolation from T0 (ambient) to T60 (Smith and Dobbie, 2001). An automatic weather station near the experimental site provided half-hourly temperature measurements, which were used to calculate the gas fluxes (data owned by NERC – Centre for Ecology and Hydrology). Cumulative gas fluxes were then calculated, with the N_2O emissions accumulated over 184 days in 2017, 302 days for the original five treatments in 2018, and 199 days for split-plots (B) in 2018. A seasonal cumulative CO_2 was measured over 100 days post cultivation in 2017 to monitor CO_2 fluxes from cultivation to pasture establishment, with a seasonal cumulative N_2O determined for the same length of time to compare N_2O and CO_2 data. The total cumulative N_2O was used to determine the N_2O EF for each treatment. The seasonal Global Warming Potential (GWP) was determined via the conversion of the seasonal cumulative CO_2 from CO_2-C to CO_2, and seasonal cumulative N_2O from N_2O-N to N_2O. The CO_2 equivalent of the seasonal cumulative N_2O was calculated by multiplying with its GWP (298) and adding the resulting value to the seasonal cumulative CO_2 for each treatment.

3.2.4 Calculations and statistical analyses

Statistical analyses and graphical representations were conducted using packages in the R environment (R Core Team, 2019). Blocked one-way ANOVA was used to determine the difference between treatments and the block effect for grass growth. This was followed by a Tukey post-hoc test. Significance was concluded at the $p < 0.05$ level. The N_2O and CO_2 flux was calculated by inputting data on the vials N_2O/CO_2 concentration, air temperature, atmospheric pressure, chamber height and closure period into a standard spreadsheet, with a mean flux calculated per treatment, following the method of Chadwick et al. (2018), and the equation in Scheer et al. (2014). Cumulative N_2O and CO_2 emissions were calculated by trapezoidal integration between sampling points. Differences in N_2O flux were also compared via a blocked one-way ANOVA. N_2O EFs were calculated by expressing the N_2O emitted from the treatments as a percentage of the N fertiliser applied (equation in Marsden et al., 2016). The baseline control mean was calculated as the mean effect across the three blocks in each blocked one-way ANOVA performed. Control 1 was removed from the N_2O calculations in 2018 due to the anomalous value for this replicate. For the yield-scaled N_2O emissions, the cumulative N_2O emissions per treatment for the 2017 N_2O sampling period of 184 days was used to calculate a daily N_2O emission value for each treatment. The yield-scaled emissions was not calculated for 2018 due to very low N_2O data in 2018, a consequence of the major drought experienced during the sampling period. For the grass yield, total grass production values for the monitored period (224 days for grass measurements) were used to calculate a daily grass production value for each treatment. Daily N_2O emission values were divided by daily grass production values to give the daily yield-scaled emissions for each treatment in order to determine the most efficient treatment in maximising pasture production, at least GHG cost. The daily yield-scaled emissions were expressed as g N_2O kg^{-1} DM of grass produced day^{-1}.

3.3 Results

3.3.1 Grass growth and soil pH

3.3.1.1 2017

Grass growth reached higher values from the reseeded plots than the permanent pasture plots over the 2017 growing season (Figure 3.3), with the mean daily grass growth at 20.0 kg DM ha^{-1} day^{-1} for the ploughed treatments and 18.7 kg DM ha^{-1} day^{-1} for the rotovated treatment. This was particularly evident at the peak of the growing season (July – September). In addition to this, the mean daily grass growth was higher following lime and fertiliser input (12.1 kg DM ha^{-1} day^{-1}) compared to the control

(8.2 kg DM ha^{-1} day^{-1}). Statistical analysis revealed a significant difference in the average seasonal grass growth between treatments (Table 3.2).

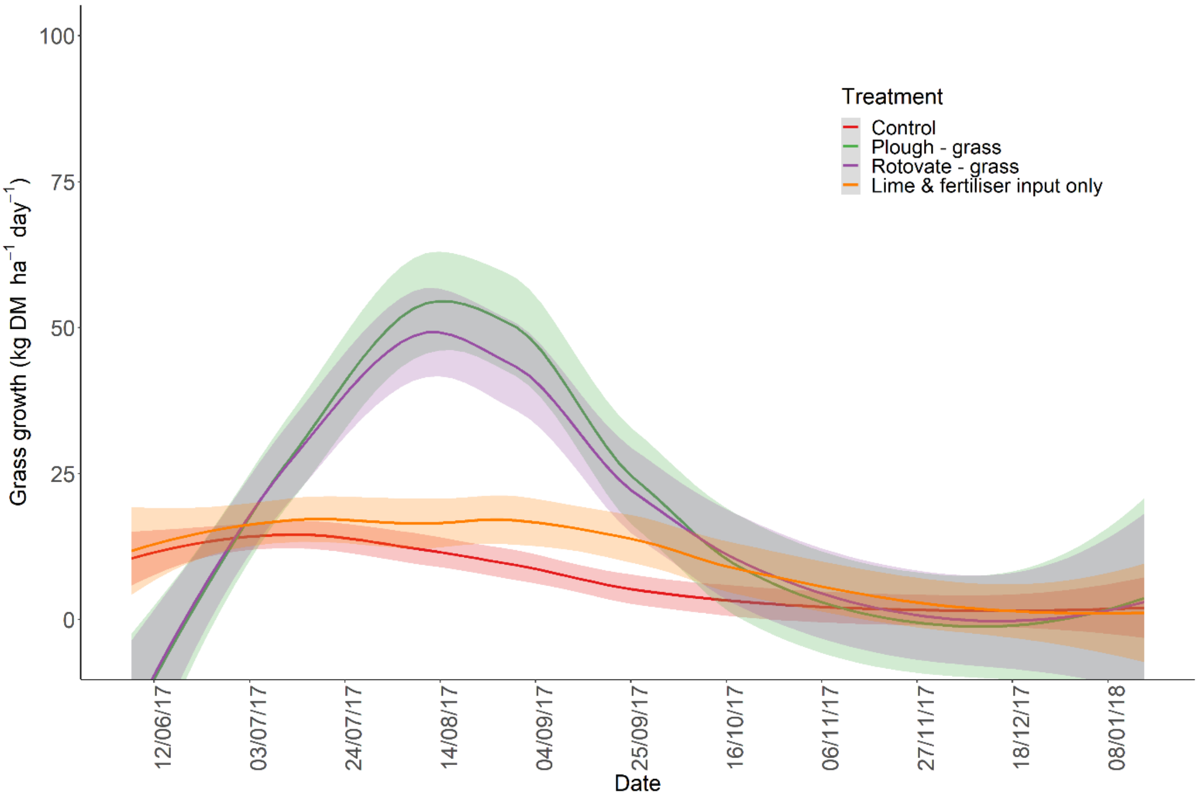

Figure 3.3. Mean daily grass growth of the treatments over 2017 sampling period. Dates are expressed in dd/mm/yy. Coloured lines represent the treatment means (n = 3) and the shaded areas represent the upper and lower bounds of the standard error of the mean (SEM) and appear to exceed the minimum value on the *y*-axis due to the nature of the smoothing curve.

Table 3.2. Daily average grass growth in 2017 (- indicates no results due to autumn grass reseed, so no production measurements for that treatment, n = 3 otherwise, ± = SED). Treatments with different letters were significantly different at $p < 0.05$.

Treatment	Grass growth (kg DM ha^{-1} day^{-1})
Control	$8.22^{a} \pm 2.73$
Forage crop – grass	-
Plough – grass	$20.00^{b} \pm 3.15$
Rotovate – grass	$18.72^{b} \pm 3.15$
Lime and fertiliser input only	$12.09^{ab} \pm 3.15$

3.3.1.2 2018 and 2019 with split-plot nitrogen application

Variation in soil pH following lime application is provided in Appendix 3.3 (Figure A). The mean soil pH was greater at the beginning of 2018 for the treatments that received lime, indicating the positive effect of applying lime. Despite this, the average pH displayed a declining trend in most treatments, apart from the forage crop - grass treatment which remained fairly consistent. The mean soil pH of the plots was 5.9 ± 0.09 on the last sampling date. There was a significant difference in soil pH between the control and improved treatments during this period, once again indicating the effect of lime application in raising soil pH. The blocked one-way ANOVA indicated there was no significant block effect between the three blocks ($p > 0.05$).

Figure 3.4 is a comparison of daily grass growth following split-plot treatment in 2018 and 2019. For 2018, there was no significant difference in grass growth per day between the treatments that received 50 kg N ha^{-1}; likewise when treatments received 130 kg N ha^{-1} (Table 3.3). However, daily grass growth more than doubled, and was significantly higher from the reseeded treatments following a total of 130 kg N ha^{-1} applied when compared with a single dose of 50 kg N ha^{-1} application. There was a significant difference between the mean daily grass growth of the three reseeded (forage crop - grass at 17.7 kg DM ha^{-1} day^{-1}, plough - grass at 13.9 kg DM ha^{-1} day^{-1} and rotovate – grass at 16.3 kg DM ha^{-1} day^{-1}) treatments that received 130 kg N ha^{-1} and the control treatment (3.3 kg DM ha^{-1} day^{-1}). In addition to this, mean grass growth per day was significantly higher from the forage crop – grass 130 kg N ha^{-1} treatment than the forage crop – grass (7.3 kg DM ha^{-1} day^{-1}), plough – grass (4.0 kg DM ha^{-1} day^{-1}), rotovate – grass (6.1 kg DM ha^{-1} day^{-1}) and lime and fertiliser input only (4.7 kg DM ha^{-1} day^{-1}) 50 kg N ha^{-1} treatments ($p < 0.05$). Daily grass growth was also significantly greater in the rotovated treatments that received the highest N application than the plough – grass and lime and fertiliser input that was applied 50 kg N ha^{-1}.

For 2019, daily grass growth was also greater from the treatments than received 130 kg N ha^{-1} compared to 50 kg N ha^{-1}. However, the extent of the difference was not as prominent as in 2018, with no direct significant effect of N application rate. Despite this, mean daily grass growth was significantly greater in both the forage crop – grass treatments (50 kg N ha^{-1} at 17.9 kg DM ha^{-1} day^{-1} and 130 kg N ha^{-1} at 22.6 kg DM ha^{-1} day^{-1}) than the control in 2019 (5.3 kg DM ha^{-1} day^{-1}).

Table 3.3. Daily average grass growth in 2018 and 2019 (n = 3, ± = SEM). Treatments with different letters were significantly different at $p < 0.05$.

Treatment	Grass growth (kg DM ha^{-1} day^{-1})	
	2018	2019
Control	$3.30^{ac} \pm 1.14$	$5.26^{a} \pm 2.59$
Forage crop – grass (A)	$7.26^{adef} \pm 1.46$	$17.86^{b} \pm 3.48$
Forage crop – grass (B)	$17.74^{b} \pm 5.09$	$22.58^{b} \pm 4.64$
Plough – grass (A)	$4.03^{agh} \pm 1.48$	$12.77^{ab} \pm 3.48$
Plough – grass (B)	$13.88^{bdgil} \pm 5.87$	$16.10^{ab} \pm 5.34$
Rotovate – grass (A)	$6.08^{aijk} \pm 1.48$	$14.51^{ab} \pm 3.48$
Rotovate – grass (B)	$16.28^{bej} \pm 5.87$	$16.96^{ab} \pm 5.34$
Lime and fertiliser input only (A)	$4.69^{alm} \pm 1.49$	$8.41^{ab} \pm 3.48$
Lime and fertiliser input only (B)	$8.58^{bcfhkm} \pm 5.87$	$13.26^{ab} \pm 5.60$

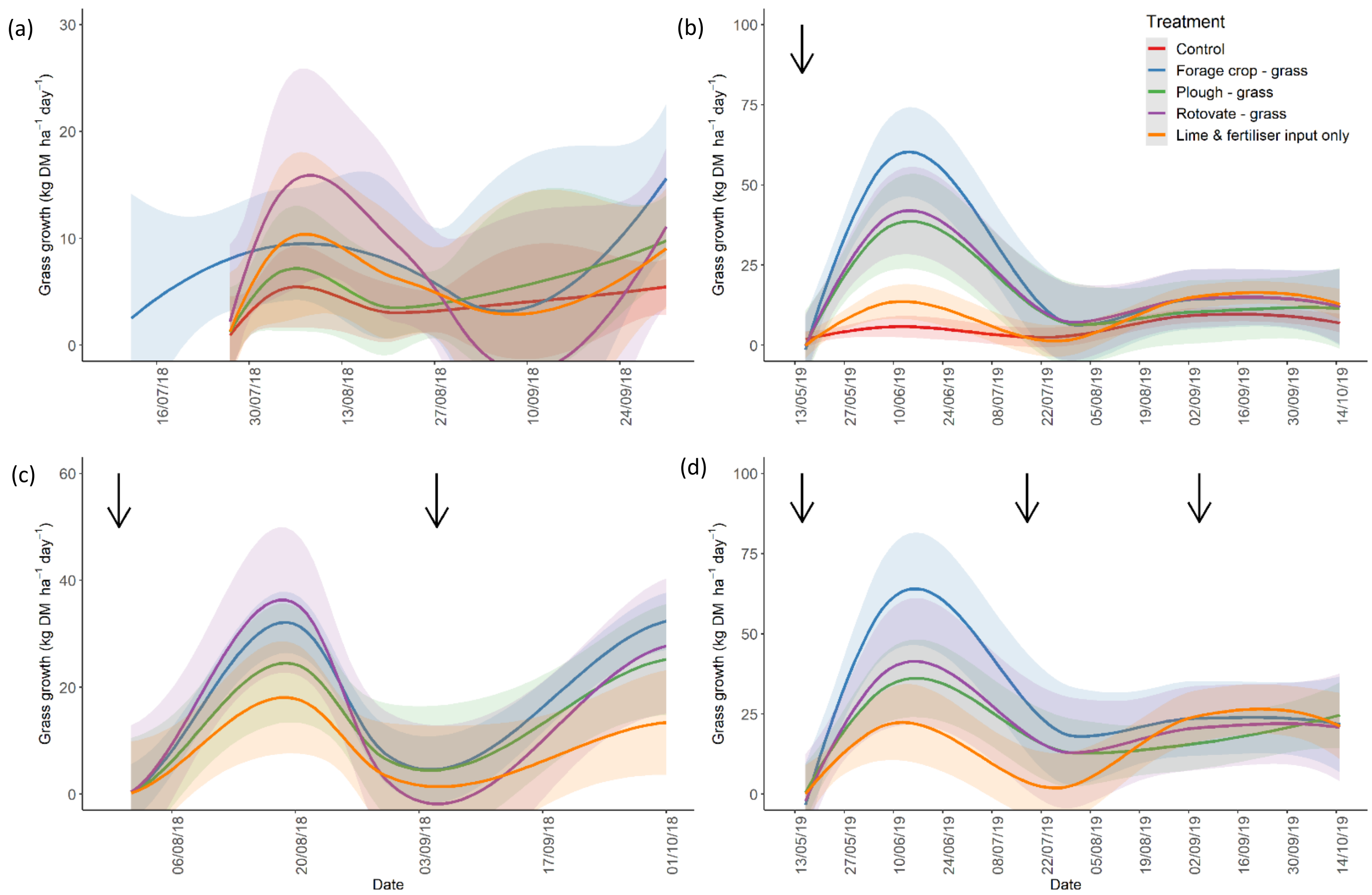

Figure 3.4. Mean daily grass growth of the treatments that received a total of (a) 50 kg N ha^{-1} in 2018, (b) 50 kg N ha^{-1} in 2019, (c) 130 kg N ha^{-1} in 2018 and (d) 130 kg N ha^{-1} in 2019 sampling period. Dates are in dd/mm/yy format. Coloured lines represent the treatment means (n = 3) and the shaded areas represent the upper and lower bounds of the SEM. Arrows denote timing of N fertiliser applications. The first N fertiliser application was made prior to grass growth sampling in 2018. Forage crop – grass treatment curve in (a) commences at an earlier date due to the timing of the sward cuts. Rotovate – grass curve in (a) along with the shaded areas in all plots appear to exceed the minimum value on the *y*-axis due to the nature of the smoothing curve.

3.3.2 Grass quality

Metabolisable energy and crude protein content of the treatments during the establishment year (2017) are displayed in Figure 3.5. There was no significant block effect in grass quality ($p > 0.05$). However, the ME content was significantly higher from the reseed treatments in comparison with the permanent pasture treatments ($p < 0.05$). There was no significant effect of treatments on crude protein content.

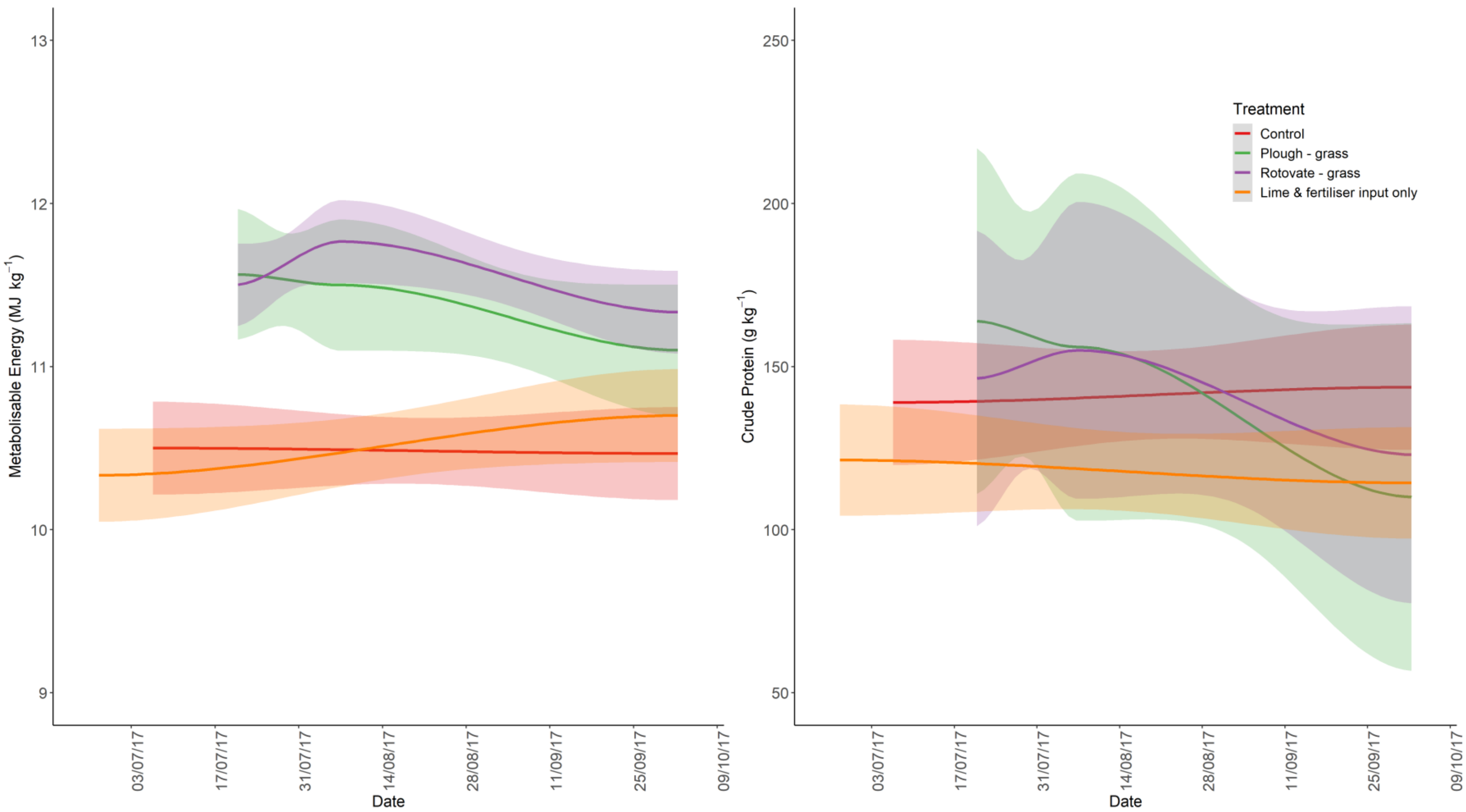

Figure 3.5. Metabolisable energy and crude protein values of the treatments during the 2017 sampling season. Coloured lines represent the treatment means (n = 3) and the shaded areas represent the upper and lower bounds of the SEM. Dates are in dd/mm/yy format. The control and lime and fertiliser input only treatments commence at an earlier date due to the timing of the sward cuts. Shaded areas in the crude protein plot appear to exceed the minimum value on the *y*-axis due to the nature of the smoothing curve.

Both ME and crude protein values were not significantly different between all treatments in both 2018 and 2019 (Table 3.4, Figure 3.6). The ME value, although insignificant, was the highest in the forage rape – autumn grass reseed treatment in both years (11.8 MJ kg^{-1}, equal highest value recorded with the plough – grass treatment in 2018 and 11.9 MJ kg^{-1} in 2019). Of the grass only treatments, the ME value was higher in the reseeded treatments.

The crude protein content was numerically (but not significantly) higher from the treatments that received 130 kg N ha^{-1} in both years, with the greatest value of 228 g kg^{-1} deriving from the lime and fertiliser input only treatment in 2018, and the plough - grass treatment in 2019 at 226 g kg^{-1}. In 2018, the crude protein content of all treatments decreased as the monitored period progressed. In contrast, the crude protein content increased during the 2019 growing season for all treatments.

Table 3.4. Average metabolisable energy and crude protein values in 2018 and 2019 (bulked samples representing each treatment, ± = SEM, treatments not significantly different).

Treatment	Metabolisable Energy (MJ kg^{-1})		Crude Protein (g kg^{-1})	
	2018	2019	2018	2019
Control	10.60 ± 0.24	10.60 ± 0.28	166.50 ± 25.06	171.33 ± 27.72
Forage crop – grass (A)	11.00 ± 0.34	11.00 ± 0.39	147.00 ± 35.44	160.00 ± 39.20
Forage crop – grass (B)	11.60 ± 0.34	11.07 ± 0.39	196.00 ± 35.44	171.67 ± 39.20
Plough – grass (A)	11.05 ± 0.34	10.73 ± 0.39	136.50 ± 35.44	152.67 ± 39.20
Plough – grass (B)	11.40 ± 0.34	10.87 ± 0.39	209.00 ± 35.44	157.67 ± 39.20
Rotovate – grass (A)	11.05 ± 0.34	10.87 ± 0.39	146.00 ± 35.44	137.33 ± 39.20
Rotovate – grass (B)	11.05 ± 0.34	10.67 ± 0.39	167.50 ± 35.44	163.00 ± 39.20
Lime and fertiliser input only (A)	10.55 ± 0.34	10.73 ± 0.39	174.50 ± 35.44	169.67 ± 39.20
Lime and fertiliser input only (B)	10.80 ± 0.34	10.60 ± 0.39	202.50 ± 35.44	196.67 ± 39.20

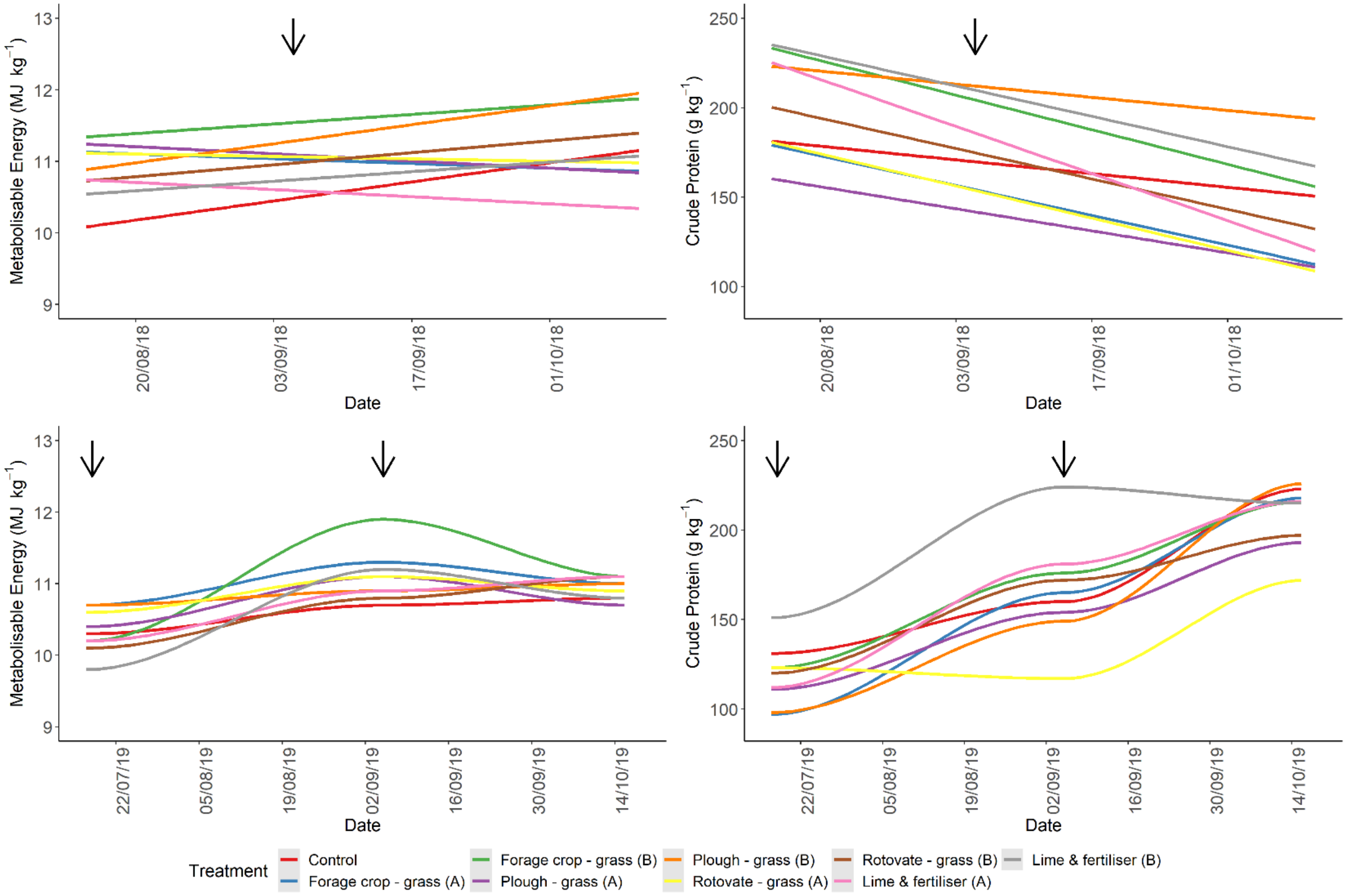

Figure 3.6. Metabolisable energy and crude protein values of the treatments during the 2018 and 2019 sampling season. Coloured lines represent bulked samples of each treatment. (A) and (B) denote N fertiliser application rates (see Table 3.1) and the arrows denote application timings during the sampling seasons. Dates are in dd/mm/yy format. The first two N fertiliser application were made prior to grass quality analysis in 2018. The first N fertiliser application was also made prior to grass quality analysis in 2019.

3.3.3 Nitrous oxide and carbon dioxide emissions

3.3.3.1 Measured fluxes

Mean N_2O fluxes varied between the treatments in 2017 (Figure 3.7). The rotovated - grass treatment had the highest N_2O fluxes between May 12th, 2017 and June 28th, 2017, with a large peak observed for these treatments at this point. Both the control and the treatment that received lime and nutrients input produced relatively small N_2O fluxes throughout the sampling period. The treatments implemented had a significant impact on the N_2O emissions (Table 3.5). However, cultivation and ploughing in particular did not lead to greater C loss via CO_2 fluxes as expected (see Appendix 3.3, Figure B).

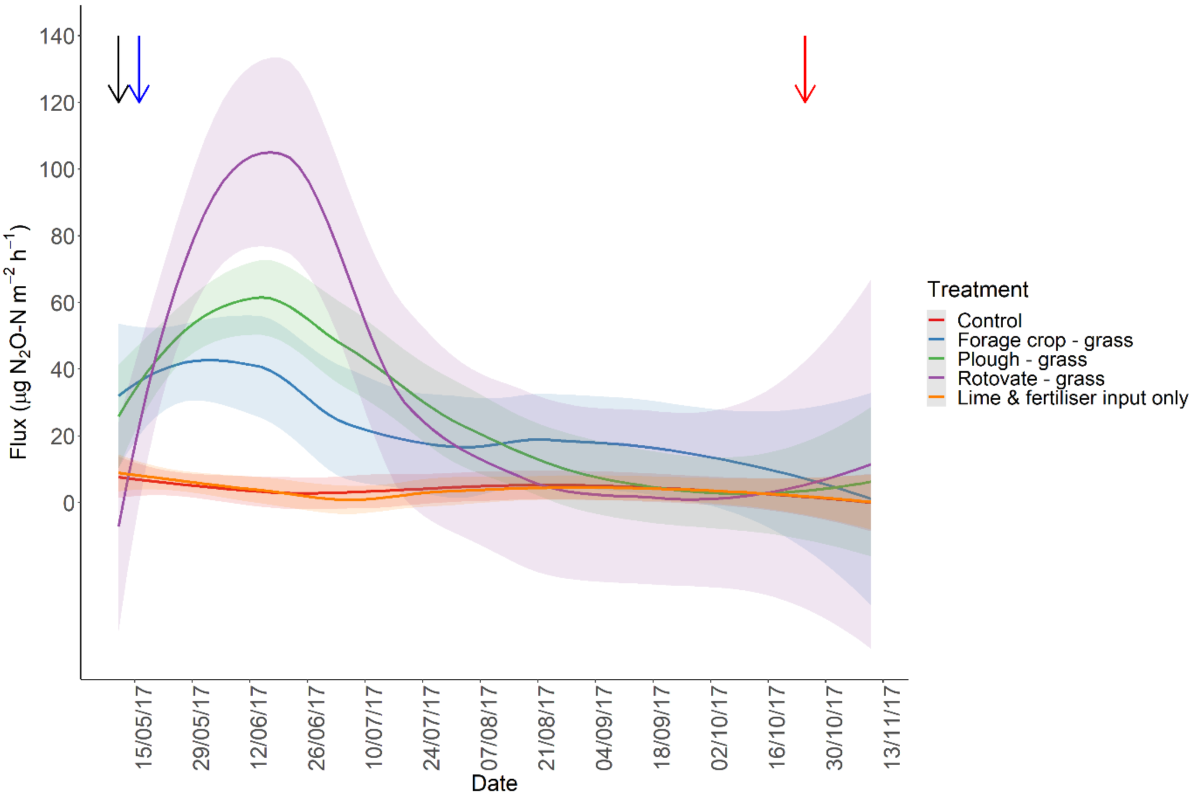

Figure 3.7. Nitrous oxide (µg N_2O-N m^{-2} h^{-1}) emissions from the treatments in 2017. Coloured lines represent the treatment means (n = 3) and the shaded areas represent the upper and lower bounds of the SEM. Dates are in dd/mm/yy format. Amendments were made at the points of the arrows. The black arrow denotes the ploughing and rotovating, the blue arrow denotes the lime and fertiliser application and the red arrow denotes a second lime application in the autumn. Shaded areas appear to exceed the minimum value on the *y*-axis due to the nature of the smoothing curve.

Table 3.5. Average hourly nitrous oxide emissions for the treatments for the 2017 sampling period ($n=3$, treatments with different letters were significantly different at $p < 0.05$, ± = SED).

Treatment	Flux (µg N_2O-N m^{-2} h^{-1})
Control	$4.65^a \pm 5.10$
Forage crop - grass	$27.52^b \pm 6.12$
Plough - grass	$31.70^b \pm 6.02$
Rotovate - grass	$37.35^b \pm 5.99$
Lime and fertiliser input only	$4.24^a \pm 6.14$

N_2O emissions from the treatments in 2018 are displayed in Appendix 3.3 (Figure C). Every treatment produced low N_2O fluxes throughout 2018. There was no significant difference between any of the treatments, indicating that greater and more frequent N fertiliser applications had no effect on N_2O emissions from this soil during the sampling period.

3.3.3.2 Cumulative greenhouse gas emissions and nitrous oxide emission factors

The variation in seasonal cumulative CO_2 and N_2O, total cumulative N_2O, the GWP for the seasonal cumulative CO_2 and N_2O combined, total cumulative N_2O for the sampling period and N_2O EF in relation to treatments over the observation period in 2017 and 2018 is displayed in Table 3.6. There was no significant relationship between the seasonal cumulative CO_2 emissions and ploughing. In 2017, the N_2O EFs from the plough – grass and rotovate – grass treatments were moderately high compared to the present default EF of 1.6% of N applied as fertiliser for direct emissions from agricultural soils (IPCC, 2019). Statistical analysis revealed that treatments had no effect on cumulative N_2O emissions in both 2017 and 2018. Furthermore, there was no significant effect of treatments on N_2O EFs, indicating that ploughing and rotovating early in the summer had no significant effect on the overall EF for the 2017 sampling period. Both the cumulative emissions and EFs from 2018 were low compared to 2017 results and the IPCC default value, with no significant effect of varying rates of N fertiliser applications. In relation to the stage of establishment and grass growth, the cumulative N_2O emissions and EF values were greater in 2017 than 2018; a similar finding to the N_2O fluxes (Figure 3.7).

Table 3.6. Cumulative nitrous oxide emissions and emission factors for the various treatments in 2017 and 2018 (n = 3, ± SEM). (B) treatments refer to the split-plot subplots in 2018 (see Table 3.1). Seasonal cumulative CO_2 and N_2O in 2017, as well as global warming potential are calculated for the first 100 days post cultivation. Total cumulative N_2O in 2017 is based on a sampling period of 184 days post cultivation. Total cumulative N_2O in 2018 is based on a sampling period of 302 days for the original five treatments and 199 days for the split-plot (B) treatments.

Treatment	2017					2018	
	Seasonal cumulative CO_2 (kg CO_2-C ha^{-1})	Seasonal cumulative N_2O (kg N_2O-N ha^{-1})	Seasonal global warming potential (kg CO_2e ha^{-1})	Total cumulative N_2O (kg N_2O-N ha^{-1})	Emission factor (% of N applied)	Total cumulative N_2O (kg N_2O-N ha^{-1})	Emission factor (% of N applied)
Control	706 ± 219	0.08 ± 0.04	2624 ± 821	0.14 ± 0.05	-	0.02 ± 0.05	-
Forage crop – grass	334 ± 37	0.69 ± 0.20	1548 ± 221	0.91 ± 0.26	0.86 ± 0.34	0.02 ± 0.11	-0.03 ± 0.17
Forage crop – grass (B)	-	-	-	-	-	0.12 ± 0.02	0.07 ± 0.03
Plough – grass	872 ± 215	0.91 ± 0.16	3626 ± 733	1.07 ± 0.25	1.55 ± 0.43	-0.01 ± 0.03	-0.08 ± 0.05
Plough – grass (B)	-	-	-	-	-	0.15 ± 0.05	0.09 ± 0.04
Rotovate – grass	821 ± 31	1.33 ± 0.72	3635 ± 300	1.50 ± 0.77	2.26 ± 1.36	0.02 ± 0.02	-0.03 ± 0.04
Rotovate – grass (B)	-	-	-	-	-	0.10 ± 0.06	0.05 ± 0.00
Lime and fertiliser input only	1029 ± 206	0.07 ± 0.03	3803 ± 768	0.13 ± 0.03	-0.01 ± 0.05	0.16 ± 0.03	0.25 ± 0.04
Lime and fertiliser input only (B)	-	-	-	-	-	0.05 ± 0.02	0.01 ± 0.02

3.3.4 Daily yield-scaled nitrous oxide emissions

The N_2O emissions along with pasture production data for 2017 were used to calculate the daily yield-scaled emissions. The 2018 data was omitted due to the abnormal environmental conditions (a major drought) during the sampling period. The data were converted to a daily production figure for both N_2O emissions and pasture production to give a comparable estimate due to the variation in days of measurements. Table 3.7 highlights that the highest daily yield-scaled emissions resulted from the rotovated treatment. The lowest yield-scaled emissions was produced by the lime and fertiliser input only treatment.

Table 3.7. Daily yield-scaled N_2O emissions for the treatments in 2017 (- indicates no results due to autumn grass reseed, so no production measurements).

Treatment	Daily yield-scaled emissions (g N_2O kg^{-1} DM of grass produced day^{-1})
Control	0.11
Forage crop - grass	-
Plough - grass	0.33
Rotovate - grass	0.48
Lime and fertiliser input only	0.07

3.4 Discussion

3.4.1 Comparative grass production and quality

Daily grass growth was significantly higher following reseeding when compared to the permanent pasture treatments in 2017. During this sampling year, daily grass growth had in fact more than doubled from the reseeded treatments, as opposed to the permanent pasture treatments at the peak of the growing season. It is well-known that reseeding is an effective method of increasing pasture response to fertiliser, as noted by AHDB (2017a). Carswell et al. (2019) reported that reseeding in lowland sites, with a new grass variety led to increased pasture productivity from the new sward as opposed to the sward it replaced. However, by contrast, they reported no significant yield improvement from the reseeded swards relative to permanent pasture, with reduced NUE during reseed years for both livestock- and grass-production systems. Comparisons between experiments of this sort should be done with caution due to differing climate, soil and environmental factors. For example, Carswell et al. (2019) conducted their experiment across a series of lowland farmlets, predominantly on stony clay soils; which were remarkably different conditions to our study site. This study was conducted within one upland site, which is reflective of other upland regions due to the typical upland characteristics.

The grass growth curves of the reseeded treatments in 2018 and 2019 (Figure 3.4) are comparable with the trends of a multi-site industry-led project, “Forage for Knowledge” (AHDB, 2020). Daily grass growth at the peak of the growing season was higher from the Forage for Knowledge project than our study. However, this project collated data from various organic and conventional dairy farms across Great Britain, and therefore was not reflective of the reduced yield that one would expect to see on upland pastures relative to more productive dairy farms.

The addition of forage rape as a break crop in 2017 prior to the grass reseed in the autumn increased grass growth in 2018 and 2019. This was particularly evident in 2019, whereby daily grass growth was in fact highest from this treatment. The role of break crops such as forage rape has shown to be fundamental in terms of disease resistance and controlling pests that may be carried over from one sward to the next when reseeding on-farm (AHDB, 2017a). However, relating disease prevention by a forage crop along with the effect on pasture production when reseeding upland pastures requires more investigation.

We hypothesised that pasture improvements in the form of reseeding and lime and fertiliser applications would lead to increased grass production due to the additional nutrients supplied. The findings presented here support this hypothesis, with the lowest daily grass growth from 2017 - 2019 deriving from the control treatment (Figures 3.3 and 3.4). Figure 3.4 implies that applying N in accordance with Defra’s RB209 guidelines led to significantly higher daily grass production as opposed to the typical application rates as reported in the British Survey of Fertiliser Practice ($p < 0.05$). It is therefore likely that soil conditions were favourable and plant uptake was very responsive to multiple N fertiliser applications. Although daily grass growth was also higher from the treatments that received N fertiliser according to RB209 in 2019, the variation in daily grass growth between the treatments were far less in 2019 in comparison to 2018. To our knowledge, this is the first study to compare differences in grass growth between RB209 and practised (British Survey of Fertiliser Practice) N application rates. RB209 is a nutrient management guide based on research emulated as practical guidance, while the British Survey of Fertiliser Practice is a review of what is practised on-farm, based on the June Agricultural Survey (Defra, 2019). Although we found that greater application of N based on the RB209 recommendations led to increased grass growth, we recognise that there would be an additional economic cost to this approach relative to typical application rates. Further work is necessary to determine the economic cost-benefits of variable N application rates on a range of pasture productivity gradients.

In our study, the pasture ME content was significantly higher from the reseeded treatment in comparison with the permanent pasture treatments during the establishment year. However, this difference was not evident between the treatments in the two subsequent study years. The crude protein content was highest in the ploughed treatment that received 130 kg N ha^{-1}, at 226 g kg^{-1} for one of the replicates at its peak in 2019 (Figure 3.6). The ME content was highest in the forage crop – grass treatment in the same year, peaking at 11.9 MJ kg^{-1} for the treatment that received 130 kg N ha^{-1} fertiliser. This value corresponds well with the data from the Forage to Knowledge project, whereby the average ME recorded was slightly higher at 13 MJ kg^{-1} (AHDB, 2020). Once again, the agricultural system implemented, in terms of grass species present in the sward along with pasture management would have affected the grass quality results as well as the grass growth. The results of this study are in contrast to Yu et al. (2011), who recorded changes in sward composition and a reduction in crude protein content in particular along with an increase in litter and mosses percentages following a reduction in nutrient inputs to pasture over time.

3.4.2 Comparative nitrous oxide and carbon dioxide emissions

There was great variation in N_2O emissions between the treatments during the 2017 sampling period (Figure 3.7). The results showed that reseeding led to higher N_2O emissions. This is also reflective in other studies, e.g. Vellinga et al. (2004) found increased tillage led to higher N_2O and CO_2 emissions in a studied year in the Netherlands. Despite this, their study also highlights the importance of renovating pastures in the longer term to prevent deterioration in pasture quality and hence, grass yield, that has to be substituted with the purchase of other feeds to fulfil the nutritional requirements of livestock. Here, we have an apparent trade-off between managing the pasture to maintain grass growth and quality and controlling the implications on N_2O emissions produced. Reseeding with legumes has been identified as an effective grassland management option in order to improve soil fertility through N-fixation, reducing the reliance on chemical-N fertiliser inputs (Galbally et al., 2010).

As previously mentioned, some studies have demonstrated greater N_2O emissions produced following ploughing. However, our study recorded the highest N_2O emissions following rotovation; a cultivation practice that is not well studied. Rotovating is a process that disturbs and breaks up the topsoil layer, creating bare ground prior to sowing. In doing so, this agronomic practice may have led to increased mineralisation of the N pool in the soil, like other cultivation processes that lead to soil disturbance (Mutegi et al., 2010). Mutegi et al. (2010) also reported higher N_2O emissions following direct drilling and harrowing in comparison with conventional tillage in a winter barley field experiment. In contrast, Burford et al. (1981) reported that cultivation practices which destroy the topsoil layer, ploughing in particular, does not always produce the highest N_2O fluxes, with greater soil-derived N_2O emissions from direct-drilled soils. Nevertheless, their study was conducted on clay soils. Valuable estimations of N_2O emissions from soils can vary substantially from others due to differences in soil type and cultivation dates. Ball et al. (2007) reported early spring as the most favourable period for ploughing organic pastures due to the moderately low temperature limiting N mineralisation in the soil, yet adequate N available and offtake for crop growth with increases in temperature later on in the spring.

Contradictory to the findings of others (Ogle et al., 2004; Reinsch et al., 2018; Smith et al., 2007; Vellinga et al., 2004; Willems et al., 2011), cultivation for land-use conversion, ploughing and rotovating for reseeding pasture in our experiment, did not lead to considerable CO_2 emissions released from the soil. Several previous studies have regularly monitored CO_2 fluxes within the first 24 hours post-cultivation (Álvaro-Fuentes et al., 2007, Ellert and Janzen, 1999). Measurements for our experiment occurred an hour after ploughing and rotovating, and daily from then onwards. It is therefore possible that the peak fluxes could have occurred in between measurements within the first 24 hours. Willems et al. (2011) monitored the effect of ploughing on CO_2 emissions and recorded a single brief CO_2 peak at 6.91 g CO_2 m^{-2} h^{-1} immediately after ploughing, followed by a rapid decline in CO_2 fluxes produced. This value is significantly higher than the greatest peak recorded in our experiment. However, the daily rainfall was far greater for their sampling season than that of ours in 2017, which was reflected in their results in that CO_2 emissions positively correlated with volumetric water content. In contrast, Yamulki and Jarvis (2002) observed greater CO_2 emissions from undisturbed pasture than ploughed land. Similar to our experiment, this finding was observed following summer ploughing of long-term grassland sward.

An apparent decline in soil pH from 2018 to 2019 suggests that the lime application on June 5th, 2018 was ineffective. This was unexpected, as granulated lime is highly reactive and therefore, we hypothesised an increase in soil pH over the application years. However, the sampling period experienced abnormal weather conditions due to lack of rain in 2018. Fystro and Bakken (2005) observed an instant increase in soil pH in the top 25 mm of the soil following lime application. However, beyond this soil depth, changes in soil pH was dependent on site precipitation. It is therefore possible that poor soil moisture conditions due to low rainfall during the 2018 sampling period may have restricted the soil's ability to react to the lime applied in our study, leading to N surplus (Gibbons et al., 2014). Several studies have identified that changes in environmental conditions, such as rainfall, soil and air temperature can lead to strong temporal variability in N_2O emissions (Cantarel et al., 2012; Flechard et al., 2007; Harty et al., 2016). This could explain the lack of N_2O emission peaks from the treatments during the summer of 2018, with three split doses of N fertiliser applied to some of the treatments. Splitting N applications over the growing season would likely reduce the risk of N_2O emissions as opposed to one large N application. However, cumulative emissions of N_2O from the soil would be higher due to a greater total N application rate. It is advised in RB209 to reduce N application to swards in the event of a drought due to restricted grass growth and hence, poor N uptake by the plant (Defra, 2019). In the event of this occurring, N loss captured as N_2O emissions deriving from the soil would be minimal. Wang and Cai (2008) found that increased levels of soil moisture (100% water-holding capacity) led to greater N_2O emissions produced from arable soils that received N fertiliser. We saw similar trends in our study in that the N_2O fluxes produced were very low in 2018, when rainfall and soil moisture conditions were also low. This would also have had an impact on the EFs, with high EFs for the reseeded treatments in 2017 in comparison with the IPCC default for fertiliser applied to managed agricultural soils. However, the EF for the lime and fertiliser input only treatment was significantly lower than the default value (IPCC, 2019). In contrast, the 2018 EFs for all treatments were significantly lower than the IPCC default value of 1.6% and the uncertainty range, possibly due to the dry climate (Bell et al., 2015).

The yield-scaled emissions were calculated in order to directly compare N_2O emissions and grass production from each treatment in 2017, expressed as g N_2O emitted kg^{-1} DM of grass produced. To our knowledge, there are no existing data on this in relation to upland swards. The treatment that resulted in the highest yield-scaled emissions was the rotovated treatment. The lowest yield-scaled emission was from the lime and fertiliser only treatment. These results suggest that ploughing and rotovating were the most dominant source of N_2O in 2017, while applying lime and fertiliser alone to permanent pasture was the most efficient option in terms of maximising grass production for the least N_2O emissions. However, further consideration should be given to wider potential GHGs associated with the lime and fertiliser treatment, such as the GHG emissions produced from the production of lime and fertiliser (FAO, 2017), resulting in higher overall GHG production from these treatments. Further work is necessary to explore other potential sources of N loss from upland pastures and specifically from cattle systems in these areas e.g. N_2O and ammonia (NH_3) from urine patches, to advise evidence-based policy decision making for the future of upland land use.

3.5 Conclusions

Land operations in the form of ploughing and rotovating followed by reseeding led to increased grass production from upland pasture. Neither ploughing or rotovating resulted in greater C loss from the soil via CO_2 emissions. However, N_2O emissions from the soil also increased following reseeding, with N_2O EFs greater than the IPCC default value. However, N_2O EFs from the improved pasture at the upland site was much lower than the IPCC default value in the 2018 growing season, which was impacted by a drought. There was a significant difference in pasture ME content following seed establishment in 2017. However, there was no significant interaction between reseeding, lime and fertiliser application and pasture quality in 2018 and 2019. Yield-scaled emissions indicated that lime and fertiliser applications alone to upland permanent pasture was the most effective treatment in increasing pasture production for the least N_2O loss from the soil. These results support the efficient use of nutrient applications where required in order to increase pasture productivity in upland regions. Extrapolating the experiment to several other upland farms would be advantageous to fully capture the variability within grazing systems in terms of pasture growth rate and N_2O emissions produced. The results could be used to demonstrate the various methods that can be implemented to increase upland pasture productivity for upland livestock farmers, and their environmental effect in terms of N_2O production. In summary, our findings demonstrate the importance of considering both pasture production and N_2O emissions side-by-side when improving upland pasture for maximum NUE and to reduce negative environmental impacts.

3.6 References

AHDB, 2020. Forage for Knowledge – Grass growth. [Online]. Available at: https://projectblue.blob.core.windows.net/media/Default/Dairy/FFK/FFK_April%202.pdf. Accessed 5th April, 2020.

AHDB, 2018. Improving pasture for Better Returns. [Online]. Available at: http://beefandlamb.ahdb.org.uk/wp-content/uploads/2018/04/pastureBR_WEB_2018_04_25.pdf. Accessed 5th April, 2020.

AHDB, 2017a. Grassland Reseeding Guide. ed. AHDB, Warwickshire, UK. [Online]. Available at: http://beefandlamb.ahdb.org.uk/wp-content/uploads/2018/06/Grassland-reseeding-guide.pdf. Accessed 15th February, 2020.

AHDB, 2017b. Nutrient Management Guide (RB209). Section 1: Principles of nutrient management and fertiliser use. [Online]. Available at:

https://ahdb.org.uk/documents/RB209/RB209_Section1_WEB_2017-12-01.pdf. Accessed 15th February, 2020.

AHDB, 2016. How to use fertiliser strategically. [Online]. Available at: http://beefandlamb.ahdb.org.uk/how-to-use-fertiliser-strategically/. Accessed 28th May, 2020.

Álvaro-Fuentes, J., Cantero-Martínez, C., López, M.V., Arrúe, J.L., 2007. Soil carbon dioxide fluxes following tillage in semiarid Mediterranean agroecosystems. *Soil and Tillage Research 96*(1-2), 331-341.

Ball, B.C., Watson, C.A., Crichton, I., 2007. Nitrous oxide emissions, cereal growth, N recovery and soil nitrogen status after ploughing organically managed grass/clover swards. *Soil Use and Management 23*(2), 145-155.

Bell, M.J., Rees, R.M., Cloy, J.M., Topp, C.F.E., Bagnall, A., Chadwick, D.R., 2015. Nitrous oxide emissions from cattle excreta applied to a Scottish grassland: Effects of soil and climatic conditions and a nitrification inhibitor. *Science of the Total Environment 508*, 343-353.

Bertora, C., Van Vliet, P.C., Hummelink, E.W. and Van Groenigen, J.W., 2007. Do earthworms increase N_2O emissions in ploughed grassland?. *Soil Biology and Biochemistry 39*(2), 632-640.

Boorman, D., 2019. COSMOS-UK Henfaes data. [Email]. Sent 8th October, 2019. Accessed 12th October, 2019.

Burford, J.R., Dowdell, R.J., Crees, R., 1981. Emission of nitrous oxide to the atmosphere from direct-drilled and ploughed clay soils. *Journal of the Science of Food and Agriculture 32*(3), 219-223.

Cantarel, A.A., Bloor, J.M., Pommier, T., Guillaumaud, N., Moirot, C., Soussana, J.F., Poly, F., 2012. Four years of experimental climate change modifies the microbial drivers of N_2O fluxes in an upland grassland ecosystem. *Global Change Biology 18*(8), 2520-2531.

Cardenas, L.M., Bhogal, A., Chadwick, D.R., McGeough, K., Misselbrook, T., Rees, R.M., Thorman, R.E., Watson, C.J., Williams, J.R., Smith, K.A., Calvet, S., 2019. Nitrogen use efficiency and nitrous oxide emissions from five UK fertilised grasslands. *Science of the Total Environment 661*, 696-710.

Cardenas, L.M., Misselbrook, T.M., Hodgson, C., Donovan, N., Gilhespy, S., Smith, K.A., Dhanoa, M.S., Chadwick, D., 2016. Effect of the application of cattle urine with or without the nitrification inhibitor DCD, and dung on greenhouse gas emissions from a UK grassland soil. *Agriculture, Ecosystems and Environment 235,* 229-241.

Carlsson, G., Huss-Danell, 2003. Nitrogen fixation in perennial forage legumes in the field. *Plant and Soil 253*(2), 353-372.

Carswell, A.M., Gongadze, K., Misselbrook, T.H., Wu, L., 2019. Impact of transition from permanent pasture to new swards on the nitrogen use efficiency, nitrogen and carbon budgets of beef and sheep production. *Agriculture, Ecosystems and Environment 283,* 106572.

Chadwick, D.R., Cardenas, L.M., Dhanoa, M.S., Donovan, N., Misselbrook, T., Williams, J.R., Thorman, R.E., McGeough, K.L., Watson, C.J., Bell, M., Anthony, S.G., 2018. The contribution of cattle urine and dung to nitrous oxide emissions: Quantification of country specific emission factors and implications for national inventories. *Science of the Total Environment 635*, 607-617.

Clough, T.J., Kelliher, F.M., Sherlock, R.R., Ford, C.D., 2004. Lime and soil moisture effects on nitrous oxide emissions from a urine patch. *Soil Science Society of America Journal 68*(5), 1600-1609.

Cuttle, S.P., James, A.R., 1995. Leaching of lime and fertilisers from a reseeded upland pasture on a stagnogley soil in mid-Wales. *Agricultural Water Management 28*(2), 95–112.

Davies, A., 2010. Review of the historical environmental changes in the UK uplands relevant to management and policy. University of Stirling.

Davies, D.A., Munro, J.M.M., Morgan, T.E.H., 1984. Potential pasture production in the uplands of Wales. 6. The relative performance of sown species. *Grass and Forage Science 39*, 229–238.

Davies, D.A., Munro, J.M.M., 1974. Potential pasture production in the uplands of Wales: 4. Nitrogen response from sown and natural pastures. *Grass and Forage Science 29*(2), 149-158.

Defra, 2019. The British Survey of Fertiliser Practice – Fertiliser use on farm crops for crop year 2018. [Online]. Available at:

https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/872453/fertiliseruse-report2018-13mar20.pdf. Accessed 14th April, 2020.

Edwards, G.R., Parsons, A.J., Rasmussen, S., 2007. High sugar grasses for dairy systems. In: Meeting the challenges for pasture-based dairying. In: *Proceedings of the 3rd Dairy Science Symposium,* 307-334. Eds D.F. Chapman, D.A. Clark, K.L. Macmillan, D.P. Nation. Melbourne, Australia: National Dairy Alliance.

Ellert, B.H., Janzen, H.H., 1999. Short-term influence of tillage on CO_2 fluxes from a semi-arid soil on the Canadian Prairies. *Soil Tillage Research 50*(1), 21-32.

Evans, J.G., Fraser, M.D., Owen, I., Davies, D.A., 2011. An evaluation of two perennial ryegrass cultivars (AberDart and Fennema) for sheep production in the uplands. *The Journal of Agricultural Science 149*(2), 235-248.

FAO, 2017. Global database of GHG emissions related to feed crops. [Online]. Available at: http://www.fao.org/3/a-i8276e.pdf. Accessed 28th May, 2020.

Flechard, C.R., Ambus, P., Skiba, U., Rees, R.M., Hensen, A., Van Amstel, A., Van Den Pol-Van Dasselaar, A., Soussana, J.F., Jones, M., Clifton-Brown, J., Raschi, A., 2007. Effects of climate and management intensity on nitrous oxide emissions in grassland systems across Europe. *Agriculture, Ecosystems and Environment 121*(1-2), 135-152.

Foster, B.L., Gross, K.L., 1998. Species richness in a successional grassland: effects of nitrogen enrichment and plant litter. *Ecology 79,* 2593-2602.

Fraser, M.D., Moorby, J.M., Vale, J.E., Evans, D.M., 2014. Mixed grazing systems benefit both upland biodiversity and livestock production. *PloS one*, *9*(2), p.e89054.

Fraser, M.D., 2008. Managing change in the uplands. [Online]. Available at: https://www.aber.ac.uk/en/media/departmental/ibers/pdf/ibersinnovationsen08/chap_2.pdf. Accessed 5th April, 2020.

Frame, J., Newbould, P., Munro, J.M.M., 1985. Herbage production from the hills and uplands hill and upland livestock production. Occasional Publication of the British Society of Animal Production, 10, 9-37. Edinburgh: BSAP.

Fystro, G., Bakken, A.K., 2005. Soil reaction, yields and herbage element content as affected by lime applied on established leys in a multi-site field trial. *The Journal of Agricultural Science 143*(5), 407-420.

Galbally, I.E., Meyer, M.C., Wang, Y.P., Smith, C.J., Weeks, I.A., 2010. Nitrous oxide emissions from a legume pasture and the influences of liming and urine addition. *Agriculture, Ecosystems and Environment 136*(3-4), 262-272.

Gibbons, J.M., Williamson, J.C., Williams, A.P., Withers, P.J., Hockley, N., Harris, I.M., Hughes, J.W., Taylor, R.L., Jones, D.L., Healey, J.R., 2014. Sustainable nutrient management at field, farm and regional level: Soil testing, nutrient budgets and the trade-off between lime application and greenhouse gas emissions. *Agriculture, Ecosystems and Environment 188*, 48-56.

Glømme, H., 1931. Kalkning. Et forsømt kulturmiddel i det norske jordbruk (in Norwegian). *Scientific Reports from the Agricultural College of Norway 11*, 101–112.

Harty, M.A., Forrestal, P.J., Watson, C.J., McGeough, K.L., Carolan, R., Elliot, C., Krol, D., Laughlin, R.J., Richards, K.G., Lanigan, G.J., 2016. Reducing nitrous oxide emissions by changing N fertiliser use from calcium ammonium nitrate (CAN) to urea based formulations. *Science of the Total Environment 563*, 576-586.

Hayes, R.C., Li, G.D., Conyers, M.K., Virgona, J.M., Dear, B.S., 2016. Lime increases productivity and the capacity of lucerne (Medicago sativa L.) and phalaris (Phalaris aquatica L.) to utilise stored soil water on an acidic soil in south-eastern Australia. *Plant and Soil 400*(1-2), 29-43.

Hopkins, C.G., Readhimer, J.E., 1907. Soil improvement for the worn hill lands of Illinois. *Illinois Agricultural Experiment Station Bulletins 115*, 431–443.

Hughes, R., Munro, J.M.M., 1962. The improvement of herbage varieties for hill reseeding. *Report of the Welsh Plant Breeding Station for 1961*, 80-84.

IPCC, 2019. 2019 Refinement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories. [Online]. Available at:

https://www.ipcc.ch/report/2019-refinement-to-the-2006-ipcc-guidelines-for-national-greenhouse-gas-inventories/. Accessed 5th February, 2020.

IPCC, 2006a. Chapter 11: N2O Emissions from Managed Soil and CO2 Emissions from Lime and Urea Application. In: De Klein, C., Novoa, R.S.A., Ogle, S., Smith, K.A., Rochette, P., Wirth, T.C., McConkey, B.G., Mosier, A., Rypdal, K. (Eds.), Intergovernmental Panel on Climate Change, 2006. Guidelines for National Greenhouse Gas Inventories Agriculture, Forestry, and Other Land Use Intergovernmental Panel on Climate Change. [Online]. Available at: http://www.ipcc-nggip.iges.or.jp/public/2006gl/pdf/4_Volume4/V4_11_Ch11_N2OandCO2.pdf. Accessed 8th January, 2020.

IPCC, 2006b. In: Eggleston, H.S., Buendia, L., Miwa, K., Ngara, T., Tanabe, K. (Eds.), 2006. IPCC Guidelines for National Greenhouse Gas Inventories, Prepared by the National Greenhouse Gas Inventories Programme. IGES, Japan.

Karalic, K., Loncaric, Z., Popovic, B., Ivezic, V., 2013. Phosphorus and potassium availability change by liming of acid soils. In: *Fourth International Scientific Symposium" Agrosym 2013", Jahorina, Bosnia and Herzegovina, 3-6 October, 2013. Book of Proceedings*, 119-124. Faculty of Agriculture, University of East Sarajevo.

Keady, T.W.J., Mayne, C.S., Fitzpatrick, D.A., 2000. Prediction of silage feeding value from the analysis of the herbage at ensiling and effects of nitrogen fertilizer, date of harvest and additive treatment on grass silage composition. *The Journal of Agricultural Science 134,* 353-368.

Mansfield, L., 2011. Upland Agriculture and the Environment. Badger Press Ltd.

Marsden, K.A., Holmberg, J.A., Jones, D.L., Charteris, A.F., Cárdenas, L.M., Chadwick, D.R., 2019. Nitrification represents the bottle-neck of sheep urine patch N_2O emissions from extensively grazed organic soils. *Science of the Total Environment 695*, 133786.

Marsden, K.A., Jones, D.L., Chadwick, D.R., 2016. The urine patch diffusional area: An important N_2O source? *Soil Biology and Biochemistry 92,* 161-170.

Mazzetto, A.M., Feigl, B.J., Schils, R.L., Cerri, C.E.P., Cerri, C.C., 2015. Improved pasture and herd management to reduce greenhouse gas emissions from a Brazilian beef production system. *Livestock Science 175*, 101-112.

Mutegi, J.K., Munkholm, L.J., Petersen, B.M., Hansen, E.M., Petersen, S.O., 2010. Nitrous oxide emissions and controls as influenced by tillage and crop residue management strategy. *Soil Biology and Biochemistry 42*(10), 1701-1711.

Newbould, P., 1975. Improvement of hill pastures for agriculture – Review 2. *Journal of the British Grassland Society 30,* 41-44.

Newbould, P., 1974. Improvement of hill pastures for agriculture – Review 1. *Journal of the British Grassland Society 29,* 241-247.

Ogle, S.M., Conant, R.T., Paustian, K., 2004. Deriving grassland management factors for a carbon accounting method developed by the intergovernmental panel on climate change. *Environmental Management 33*, 474–484.

Paradelo, R., Virto, I., Chenu, C., 2015. Net effect of liming on soil organic carbon stocks: a review. *Agriculture, Ecosystems and Environment 202*, 98-107.

Peyraud, J.L., Le Gall, A., Luscher, A., 2009. Potential food production from forage legume-based-systems in Europe: An overview. *Irish Journal of Agricultural and Food Research 48,* 115-135.

Phelan, P., Moloney, A.P., McGeough, E.J., Humphreys, J., Bertilsson, J., O'Riordan, E.G., O'Kiely, P., 2015. Forage legumes for grazing and conserving in ruminant production systems. *Critical Reviews in Plant Sciences 34*(1-3), 281-326.

Poulton, P., Johnston, J., Macdonald, A., White, R., Powlson, D., 2018. Major limitations to achieving "4 per 1000" increases in soil organic carbon stock in temperate regions: Evidence from long-term experiments at Rothamsted Research, United Kingdom. *Global Change Biology 24*(6), 2563-2584.

R Core Team, 2019. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. [Online]. Available at: https://www.R-project.org. Accessed April 26th, 2020.

Ratcliffe, D. A., 1977. A Nature Conservation Review. Cambridge University Press, Cambridge.

Reed, M.S., Arblaster, K., Bullock, C., Burton, R.J.F., Davies, A.L., Holden, J., Hubacek, K., May, R., Mitchley, J., Morris, J., Nainggolan, D., 2009. Using scenarios to explore UK upland futures. *Futures 41*(9), 619-630.

Reinsch, T., Loges, R., Kluß, C., Taube, F., 2018. Effect of grassland ploughing and reseeding on CO_2 emissions and soil carbon stocks. *Agriculture, Ecosystems and Environment 265*, 374-383.

Rushton, S.P., Luff, M.L., Eyre, M.D., 1989. Effects of pasture improvement and management on the ground beetle and spider communities of upland grasslands. *Journal of Applied Ecology 26*, 489-503.

Rütting, T., Aronsson, H., Delin, S., 2018. Efficient use of nitrogen in agriculture. *Nutrient Cycling in Agroecosystems 110*, 1-5.

Scheer, C., Rowlings, D.W., Firrel, M., Deuter, P., Morris, S., Grace, P.R., 2014. Impact of nitrification inhibitor (DMPP) on soil nitrous oxide emissions from an intensive broccoli production system in sub-tropical Australia. *Soil Biology and Biochemistry 77*, 243-251.

Schulte, R.P.O., Neuteboom, J.H., 2002. Advanced analysis of dry-weight-rank data to discriminate direct and indirect interactions between white clover and grasses in a multi-species pasture under a range of management strategies. *Grass and Forage Science 57,* 113-123.

Scott, D., Keoghan, J.M., Allan, B.E., 1996. Native and low-input grasses – a New Zealand high country perspective. *New Zealand Journal of Agricultural Research 39*(4), 499-512.

Smith, P., Martino, D., Cai, Z., Gwary, D., Janzen, H., Kumar, P., McCarl, B., Ogle, S., O'Mara, F., Rice, C., Scholes, B., Sirotenko, O., 2007. Agriculture. Climate Change 2007: Mitigation. Cambridge University Press, Cambridge.

Smith, K.A., Dobbie, K.E., 2001. The impact of sampling frequency and sampling times on chamber-based measurements of N_2O emissions from fertilized soils. *Global Change Biology 7*(8), 933-945.

Stevens, C.J., Payne, R.J., Kimberley, A., Smart, S.M., 2016. How will the semi-natural vegetation of the UK have changed by 2030 given likely changes in nitrogen deposition?. *Environmental Pollution 208*, 879-889.

UK Centre for Ecology and Hydrology, 2020. Site: Henfaes Farm. [Online]. Available at: https://cosmos.ceh.ac.uk/sites/HENFS. Accessed 16th March, 2020.

Vellinga, T. V., Van den Pol-van Dasselaar, A., Kuikman, P. J. 2004. The impact of grassland ploughing on CO_2 and N_2O emissions in the Netherlands. *Nutrient Cycling in Agroecosystems 70*(1), 33-45.

Wang, L., Cai, Z., 2008. Nitrous oxide production at different soil moisture contents in an arable soil in China. *Soil Science and Plant Nutrition 54*, 786-793.

West, T.O., McBride, A.C., 2005. The contribution of agricultural lime to carbon dioxide emissions in the United States: dissolution, transport, and net emissions. *Agriculture, Ecosystems and Environment 108*(2), 145-154.

Willems, A.B., Augustenborg, C.A., Hepp, S., Lanigan, G., Hochstrasser, T., Kammann, C., Müller, C., 2011. Carbon dioxide emissions from spring ploughing of grassland in Ireland. *Agriculture, Ecosystems and Environment 144*(1), 347-351.

Yamulki, S., Jarvis, S., 2002. Short-term effects of tillage and compaction on nitrous oxide, nitric oxide, nitrogen dioxide, methane and carbon dioxide fluxes from grassland. *Biology and Fertility of Soils 36*(3), 224-231.

Yu, Y.W., Fraser, M.D., Evans, J.G., 2011. Long-term effect on sward composition and animal performance of reducing fertilizer inputs to upland permanent pasture. *Grass and Forage Science 66*(1), 138-151.

4. Variability in cattle performance and nitrous oxide emissions from fertiliser use and excretal deposition from improved and unimproved upland pasture

Abstract

Many UK upland regions have the potential to increase pasture production and utilisation, and this is increasingly important for these systems to remain economically viable. However, there is a lack of studies on how measures sought to improve pasture productivity (e.g. addressing soil nutrient deficiencies through fertiliser application) alter the degree of environmental impacts from livestock systems. Such impacts include the emissions of powerful greenhouse gases such as nitrous oxide (N_2O); which is associated with fertiliser use and cattle excretion. This study assessed cattle liveweight change from improved and unimproved upland pasture at a typical upland site, along with the environmental trade-offs in the form of N_2O emissions produced from both treatments via fertiliser application, urine, and dung deposition. On-farm data was collected and a Life Cycle Assessment (LCA) tool was used to predict the environmental footprint of both systems per kg of liveweight (LW) of cattle produced. There was no statistically significant difference in cattle daily liveweight gain (DLWG) between the treatments. We found that N_2O emissions were significantly greater following urine deposition in comparison with the control, fertiliser, and dung deposition, particularly from the unimproved grazing treatment ($p < 0.001$). The combined excretal N_2O emission factor values were 0.27% and 0.3% for the improved and unimproved grazing treatments, respectively. The annual environmental footprint was greater from the improved system (11.13 kg CO_2-eq kg^{-1} LWG) than the unimproved system (8.23 kg CO_2-eq kg^{-1} LWG). An improved understanding of these systems is valuable to help identify opportunities to increase production efficiencies on-farm as well as reduce greenhouse gas emissions from upland cattle systems.

4.1 Introduction

Given the projected increase in global population over the coming decades, the demand for livestock products is anticipated to remain constant, and/or increase (Thornton, 2010), which may be necessitate to increase production in some areas. As previously mentioned, ruminant livestock produce two of the main GHG emissions: nitrous oxide (N_2O) and methane (CH_4). Livestock-derived CH_4 is mainly produced via enteric fermentation along with manure management, whilst N_2O is generated primarily from fertiliser applied to crops fed to animals, and direct deposition of urine and dung on pasture (IPCC, 2019). Nitrous oxide is a powerful GHG, with a global warming potential (GWP) 298 times that of carbon dioxide (CO_2) (IPCC, 2007). It is also accountable for the natural destruction of stratospheric ozone due to nitrogen oxides reacting with ozone (Bhatia et al., 2010; Wuebbles, 2009). In the UK, agriculture is responsible for 75% of anthropogenic N_2O emissions produced (Skiba et al., 2012). Despite a decrease in UK N_2O emissions from agriculture in recent decades (Salisbury et al., 2014), there is increased pressure on the agricultural industry globally to reduce these further. This is due to an increase in global N_2O emissions from agriculture (Tubiello et al., 2013), as well as a significant reduction in N_2O emissions produced from other sectors (Skiba et al., 2012).

A considerable proportion of UK land is classified as Less Favoured Areas (LFAs), and these are largely upland regions. Livestock production is a significant land use in these areas, with ruminant livestock utilising poor-quality forage and converting it into valuable protein in the form of meat (DeRamus et al., 2003). Sheep production systems generally predominate the LFAs due to limited grazing and past upland agricultural policy (Mansfield, 2011). However, data from Defra (2002) show the importance of these areas for beef cattle herds in recent decades, with a population of 1.2 million in the LFAs of the UK, accounting for approximately 60% of the UK's total (more recent information does not provide the same breakdown of cattle distribution). Hence, a significant percentage of the beef produced therefore derives from breeding herds that graze in the uplands. Despite this, the beef breeding herd numbers have decreased by 7.9% between 2008 and 2018, with a reduction of 3.1% in total numbers of cattle and calves within the same time period (Defra, 2019; 2013). Based on this decline, it can be estimated that the beef cow herd population will have also reduced in the LFAs over the last decade.

Although a large proportion of land area is designated to grazing systems in the uplands, pasture productivity in these areas is often low. This is due to several reasons, primarily environmental and land constraints (Battaglini et al., 2014; Kowalczyk et al., 2014), such as slope and exposure, climatic conditions, and soil physical properties. In the period 1950-1980s, measures to improve agricultural productivity in the uplands were incentivised, including ploughing and draining the land, as well as establishing new grass swards (Newbould, 1974). In more recent times, land improvements in the form of lime and fertiliser usage are relatively low in these areas and therefore, these areas are often proposed for afforestation. Historically, the returns received by upland farmers for their outputs were low, which demotivated maintaining soil nutrient profiles when grant aid was removed (Gibbons et al., 2014), meaning that a lack of targeted nutrient management in the past may have led to their current (often unproductive) state.

There is a growing appreciation of the role of pasture-based livestock systems in the uplands to contribute towards food production, as such areas cannot be utilised for arable crop production (Eisler et al., 2014; de Vries et al., 2015). Despite this, to our knowledge, there has been little attention paid to the effect of pasture type on livestock performance in these regions. A notable exception is Fraser et al. (2009), whereby the performance of native and continental breeds of cattle was assessed when grazing grass/clover swards and semi-natural rough pasture. Their findings indicated that pasture type, specifically poorer nutritional environments in the uplands, as well as the shorter growing season limiting productivity, can significantly affect cattle performance in these areas. Studies of this sort, solely based on liveweight gain (LWG) and output per unit area, are few. The majority of other experimental studies are based on mixed grazing systems, cattle and sheep grazing sequentially (e.g. Critchley et al., 2008; Fraser et al., 2014; Fraser et al., 2007), or are a comparison of various grazing systems implemented (Stejskalová et al., 2013).

A significant proportion of N ingested by grazing cattle is not utilised by means of milk and meat protein conversion, and is consequently excreted onto pasture via urine and dung (Saggar et al., 2013). This percentage value is dependent on the grazing system implemented as well as environmental conditions, and may be as high as 75-95% in temperate regions (Eckard et al., 2010). Previous research has shown that N deposition can be as high as an equivalent of 600-1000 kg N ha^{-1} in one urination patch (Welten et al., 2013; Zaman and Blennerhassett, 2010). This equates to 2-3 times more N than the pasture can use (Zaman et al., 2009). Urine deposition on pasture leads to N cycling in the soil, resulting in the release of N_2O emissions to the atmosphere. Some say that urine is the largest driver of N_2O emissions in agriculture (Wakelin et al., 2013; Welten et al., 2013). Dung, however, is rich in C and N, and pats excreted on pasture can inhibit aeration and in doing so, promote soil anaerobicity (van Groenigen et al., 2005), leading to N_2O emissions produced via denitrification (Cardenas et al., 2016). Increased future demand for meat could therefore lead to increased environmental burden from grazed pasture in the future, at a time when reducing such impacts is a high priority.

The production of N_2O emissions is influenced by several factors, such as the livestock production system implemented, vegetation type, as well as environmental elements e.g. temperature, rainfall and soil type (Allen et al., 1996; Cardenas et al., 2016). The default emission factor (EF) for N_2O

emissions from cattle excreta during grazing was set at 2% in the IPCC guidelines, estimating that 2% of the N deposited is emitted to the atmosphere (IPCC, 2006). This value may mask the marked variation in EFs calculated from differing studies, at a time when there is a growing literature surrounding EFs from livestock production systems. Consequently, an update to the IPCC inventory report has led to EF values for both cattle and sheep in wet and dry climates individually, with the default EF for cattle urine and dung in wet climates calculated as 0.77% and 0.13%, respectively (IPCC, 2019). However, N_2O emissions and EFs are known to be largely affected by other environmental factors; highlighting the requirement for further research on N_2O emissions from extensive grazing systems. Van Groenigen et al. (2005) found that EFs for urine on grazed pasture ranged from 0.07% up to 15.5% from five UK studies. In addition to this, Bell et al. (2015) reported a significantly higher EF value following urine application in the summer (1.07%) as opposed to EFs following spring (0.2%) and autumn (0.31%) urine application. Chadwick et al. (2018) quantified country-specific EFs for the UK, and measured N_2O EF from cattle urine after grazing as 0.69% and cattle dung after grazing as 0.19%, resulting in a combined excretal N_2O EF (EF_3) of 0.49%. However, their study focused on treating lowland mineral soils, where urine and dung were collected from cattle fed lowland diets. Therefore, it is uncertain whether these new N_2O EF data from Chadwick et al. (2018) applies to cattle grazing the uplands. Whilst some experimental studies have investigated the differences in N_2O EFs between lowland and upland grazed lamb (Marsden et al., 2018), there is a lack of studies looking at N_2O EFs from cattle excretion to upland pastures.

The overall aim of our study was to explore the relationship between cattle DLWG, environmental outcomes and grazed pasture management in the uplands. For environmental outcomes, we focused on N_2O emissions from the grazing systems and the emission intensity of each system. A life cycle assessment model was used to quantify the environmental effects of the treatments implemented, by determining the environmental footprint (quantified in terms of GWP, kg CO_2 equivalents) of the LW of cattle produced on improved and unimproved upland pasture. Furthermore, the environmental efficiency of the grazing treatments was assessed, accounting for the stocking densities and fertiliser application effects.

4.2 Materials and methods

4.2.1 Study site

The study took place at Bangor University's farm, Henfaes Research Centre, Abergwyngregyn, north Wales. The experimental site measured 9.58 ha in total (7.09 ha excluding bracken areas that couldn't be grazed) and was located within an enclosed, permanent pasture parcel of land, behind a coniferous shelter belt at approximately 294 m a.s.l. Rainfall totals during the experimental period (May – September inclusive), provided by the UK Centre of Ecology and Hydrology was 422.9 mm in 2018 and 548.8 mm in 2019 (Boorman, 2019; personal communication). The soil type at the site is classified as Orthic Podzol. This land has been managed as grazing land supporting a low stocking density under Welsh Government agri-environment schemes for several years and is currently within the farm's Glastir scheme. The field was used as a site for Defra's Sustainable Intensification Research Platform project. As part of this study, two divisions of the field received optimised lime, phosphorous (P) and potassium (K) fertiliser input from 2015 to 2017. Prior to this, the site had no nutrient or lime inputs since 1982 (Williams et al., 2018). In addition to this, one treatment received dilute glyphosate spray in 2015 prior to slot seeding with high-sugar grass (HSG) varieties in 2015 and 2016 (Williams et al., 2018). The high-sugar grass varieties sown were Germinal HSG3 + timothy mix (Germinal 2017), compromising of perennial ryegrass (*Lolium perenne*), timothy (*Phleum pratense*) and white clover (*Trifolium repens*). Prior to this project, a National Vegetation Classification survey reported the vegetation as 'National Vegetation Classification MG6' grassland, dominated by *Lolium perenne-Cynosurus cristatus* grassland, *Lolio- Cynosuretum cristati* (Williams et al., 2018).

4.2.2 Experimental design

For the purpose of our experiment, the field was split to four plots of equal sizes, consisting of two grazing treatments; improved and unimproved pasture (n = 2 of each treatment) (Figure 4.1). Soil samples were collected from each plot prior to any interventions following a grid soil sampling procedure. A minimum of twenty samples were taken from each plot in a 'W' pattern using a soil corer (0 – 10 cm) and were combined for analysis of the following: pH, P, K, magnesium (Mg) and sulphur (S) (analysed by NRM Laboratories, Bracknell, Berkshire). Lime was applied in 2018 to the improved plots as Granulated Lime, 36% calcium (Calcifert, Runcorn, Cheshire) on June 14th for rapid effect, with a target soil pH of 6. Further soil sampling at the start of the 2019 experimental period indicated no need for further lime application. Fertiliser was applied to the improved plots, with P and K application based on recommendations from the Nutrient Management Guide (RB209), 8th edition (AHDB, 2017), and N application as typical for upland pastures, according to the British Survey of Fertiliser Practice (2017) (Defra, 2018). The N application rate was determined via this method rather than RB209 as the results from another field experiment indicated that pasture growth was sufficient with a single dose of N fertiliser for grazing livestock (see Figure 3.4 in Chapter 3). Therefore, a total of 50 kg ha^{-1} of N fertiliser was applied in 2019, along with 80 kg ha^{-1} of phosphorus (P_2O_5) fertiliser and 30 kg ha^{-1} of potassium (K_2O) fertiliser.

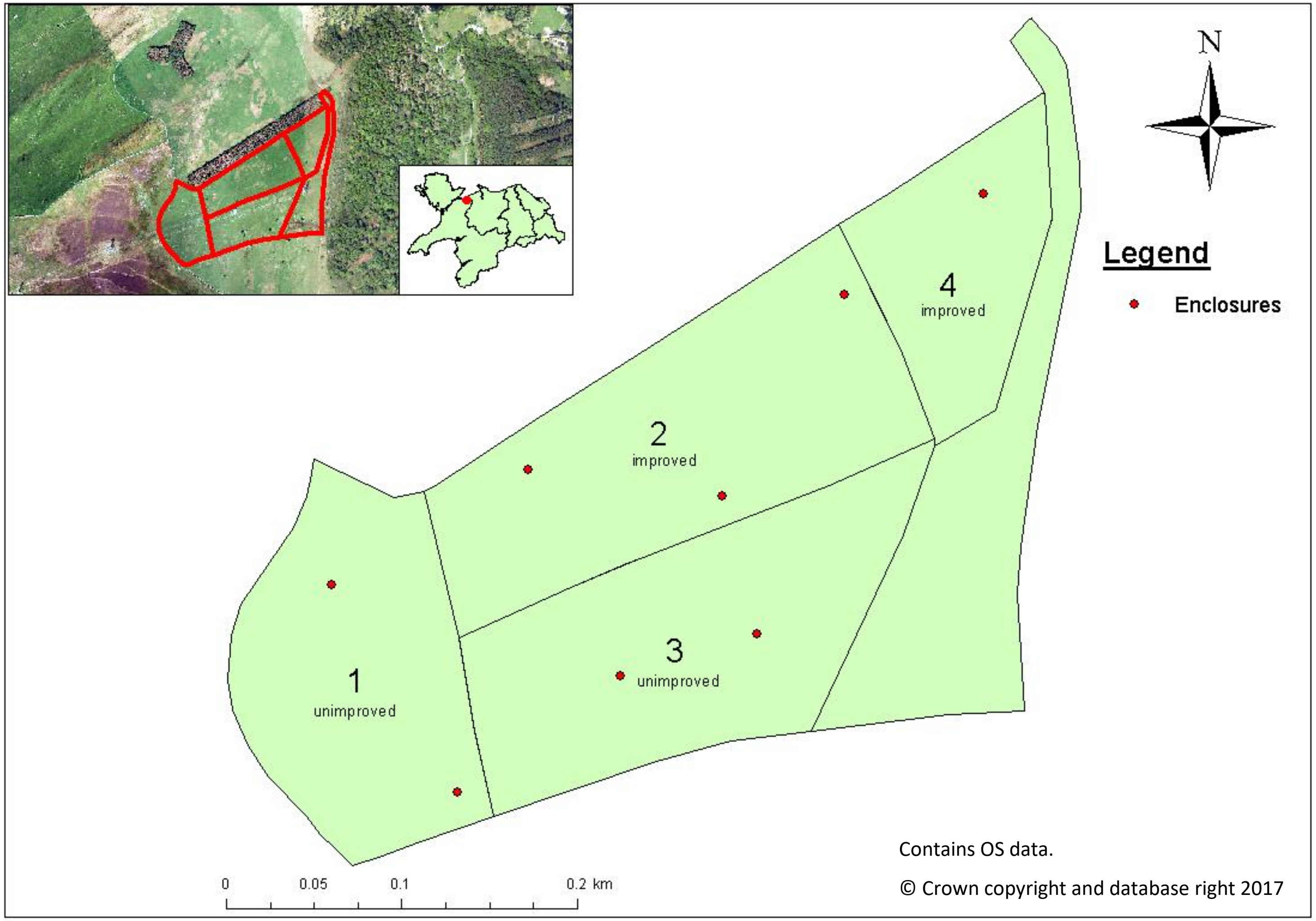

Figure 4.1. Field trial set-up, red points denote the enclosures used for data collection. Inset map shows the location of the field site at Bangor University's farm, Henfaes Research Centre (53.28°N, 4.01°W).

Four enclosures were set up within each treatment (see Figure 4.1), with gas chambers located in each enclosure in order to monitor GHG emissions. The locations of the enclosures were randomly selected using a sub metre GPS unit in 2018, with alternative locations for the enclosures in 2019 via the same selection method. GHG flux measurements were monitored from the following treatments - 1) unimproved pasture (control), 2) improved pasture (applied lime and fertiliser), 3) urine (collected from cattle grazing the improved and unimproved treatments), and 4) dung (collected from cattle grazing the improved and unimproved treatments).

The experiment was treated as a pilot study in 2018 due to the lack of previous studies on the land's ability to sustain cattle throughout the growing season (no cattle had grazed the land in the last ca. 35 years). A group of 12-23 month old heifers of three breed types (British Blue, Hereford and Belted Welsh Black) was split based on breed and LW to graze both treatments, with eight heifers grazing the improved plot and four heifers grazing the unimproved plot from June 18th, 2018 – September 26th, 2018. The mean LW of the heifers at the start of the trial was 338 kg (± 7.11; SEM). Grazing was split between the replicated treatment plots to enable a rest period for grass regrowth between grazing. All animals were treated with a Moxidectin-based wormer (Cydectin 0.5% Pour-On) prior to grazing.

The results from the pilot study in 2018 were used to determine the stocking rates for 2019. A herd of 18-23 month old heifers, with an average LW of 400 kg (± 12.48; SEM) was selected to graze the plots. The herd consisted of four different breed types in 2019: Aberdeen Angus, Aberdeen Angus X, Welsh Black and Hereford X, and plot allocation was again based on breed and LW. The heifers were again treated with the Moxidectin-based wormer prior to turnout. Grazing commenced on June 11th, 2019 until September 25th, 2019. Six heifers grazed each of the improved replicates and five heifers grazed both the unimproved replicates. Given that grass availability was sufficient in 2018, in what was an abnormally dry year that impacted on grass production, the stocking density was increased in 2019 and the replicated treatment plots were grazed simultaneously.

4.2.3 Data collection

4.2.3.1 Cattle liveweight gain

The LW of each individual heifer was recorded prior to, and at the end of the grazing period to calculate the DLWG on both a per head and per hectare basis.

4.2.3.2 Sward height and quality

Sward height was recorded on a monthly basis during the growing season using a calibrated plate meter within the plots. This data was collected in order to monitor pasture utilisation and to ensure sufficient forage for the cattle. Grass samples were collected from random locations across the plots on four occasions during the grazing period in 2019: July 30th, August 23rd, September 2nd and

September 30th to assess pasture quality during the grazing period. Samples were collected by cutting the sward with shears to stimulate cattle grazing, therefore to approximately 8-11 cm. These samples were bulked on a per treatment basis and analysed in the lab for dry matter content, 'wet' Near Infrared analysis of crude protein, sugar, metabolisable energy and digestibility.

4.2.3.3 Urine and dung collection and analysis

Urine and dung samples was collected from the heifers approximately 2-4 weeks after grazing had commenced. The heifers were walked individually through a race and into a mobile crush in order to collect a urine sample from each animal. Dung samples were collected in sealed containers by constant monitoring of the herd over a day to collect a fresh sample from each heifer. Urine and dung samples were immediately placed in a cool box, then frozen (-20 °C) after returning to the lab until required, to minimise losses of N.

Prior to field application, the urine and dung samples were defrosted for at least 12 hours to allow the samples to reach ambient temperature prior to application to the soil (Chadwick et al., 2018). Both urine and dung samples collected from the individual heifers were bulked per treatment, mixed thoroughly, and bottled, in order to provide a single urine sample of a homogenous composition and a single homogenous dung sample for each grazing treatment. Water containing Brilliant Blue dye was used to determine the urine patch size for application to the soil. This area was traced with plastic sheeting and marker pen for measuring, to ensure an accurate patch size for the experimental vegetation type. The bulked urine sample was applied directly to the static gas chamber of that treatment (n = 4). Each chamber received 1.4 L of urine, adjusted from the 1.8 L recorded by Misselbrook et al. (2016) as the cattle used in this experiment were smaller. The sample was applied manually to the allocated chambers positioned within the enclosures using a measuring container from a heifer's height to mimic an urination event.

Dung was applied at a rate of 2.41 kg per chamber and was spread to an even thickness across the chamber area. While a dung patch does not typically have defined edges, this approach of application, as used in Chadwick et al. (2018) was believed to be the most appropriate for reproducing the dung pat as there is no evidence to suggest that this method of application would affect the N_2O emissions produced (Marsden et al., 2016). Urine and dung samples, as well as fertiliser inputs were applied to the chambers on September 10th, 2018 and July 2nd, 2019. Application was delayed in 2018 due to the unusually dry weather conditions during the summer months.

Three replicates of the applied urine and dung samples were retained for lab analysis. These samples were analysed for dry matter (DM) (oven-dried at 105 °C for at least 12 hours), pH (dung in water) and EC using standard electrodes. The total N and total organic carbon content for urine was analysed by a TOC analyser, Multi N/C 2100S analyser (AnalytikJena, Jena, Germany), with the samples diluted by ×100 and ×1000. Total N and total organic carbon content for the dung samples were determined using the same method, except following dilution by ×50 and ×100. The readily available N content, that is ammonium N (NH_4^+) and nitrate N (NO_3^-), was also determined for both urine and dung samples.

4.2.3.4 Greenhouse gas emissions

Gas samples (20 ml) were collected via the static chamber technique to measure N_2O fluxes (Cardenas et al., 2016). As previously stated, a series of chambers (40 cm × 40 cm × 25 cm inserted to a soil depth of 5 cm) were set out within enclosure areas to monitor GHG emissions from a) unimproved pasture (control), b) improved pasture (applied lime and fertiliser), c) urine (collected from cattle grazing the improved and unimproved pasture), and d) dung (collected from cattle grazing the improved and unimproved pasture) for both grazing treatments (n = 4). As the unimproved grazing treatment did not receive any lime or fertiliser input, chamber b) was excluded from this grazing treatment. The order of the chambers was determined randomly. Chamber headspace sampling was conducted by measuring the accumulated N_2O within the chamber headspace during the period of time that the chamber was closed (Chadwick et al., 2014). Gas samples were collected prior to any field operations, and sampling frequency varied depending on treatment application. Samples were collected daily for the first week following treatment application, three times per week for the second week, twice in the third week, reducing to weekly sampling by the fourth week. Gas samples were taken at 0 and 60 minutes following chamber lid closure and were stored in pre-evacuated 20 ml glass vials. The samples were then analysed for N_2O using a Varian 450 Gas Chromatograph in the laboratory, with the N_2O flux calculated by assuming a linear interpolation from T0 to T60 (Smith and Dobbie, 2001). An automated weather station provided by the UK Centre of Ecology and Hydrology was situated in an adjacent field to the experimental site, which provided temperature measurements every 30 minutes. This data was used to calculate the N_2O fluxes (Boorman, 2019; personal communication). Cumulative N_2O fluxes were calculated for the sampling period of 112 days, and this value used to determine the treatments' N_2O EFs.

4.2.4 Data processing and statistical analysis

Statistical analyses and graphical representations were carried out in R (R Core Team, 2019). The N_2O flux for each chamber was determined via data entry into a standard spreadsheet. The inputted data included N_2O concentrations from the vials and chamber closure period (Chadwick et al., 2018). The fluxes were corrected for the following parameters – volume, height and base of the chamber (chamber volume to surface area ratio), air temperature and atmospheric pressure (Scheer et al., 2014). Cumulative N_2O fluxes were calculated through calculating the area under the curves via trapezoidal integration (Cardenas et al., 2010). The N_2O EFs, which was the % of applied urine-N released as N_2O-N, were calculated by subtracting the mean control N_2O-N from the mean N_2O-N for each treatment, dividing by the total N applied for the treatment, and multiplying the resulting value by 100%. Differences between the cumulative N_2O emissions were compared via one-way ANOVA.

Life Cycle Assessment (LCA) was used to quantify the environmental impacts of these grazing systems, in particular the global warming potential (GWP). The functional unit for the assessment was taken to be a kg of cattle LWG on the implemented grazing treatments, therefore the results expressed as net annual environmental burden (Styles et al., 2015). Feed intake was calculated based on weight gain, using data collected on average cattle weight and LWG, number of heifers, and pasture digestibility. Pasture digestibility was expressed as a mean value of four sampling occasions during the grazing period. Lime input was calculated based on a five-year spreading interval (Chambers and Garwood, 1998). Diesel consumption was calculated based on fuel costs per tractor hour from Nix (2019), fuel prices per litre (AHDB, 2020) and tractor hours from data collection. Parent data as well as the phase prior to turnout was excluded from the LCA. The baseline reference values used as a default EF from direct N_2O emissions in managed soils for N additions from synthetic fertiliser calcium ammonium nitrate (CAN) were derived from IPCC default values in wet climates (IPCC, 2019). The default EF of urine and dung N deposited on pasture, range and paddock by cattle was determined as the combined excretal N_2O EF (EF_3) value of 0.49% calculated for the UK in Chadwick et al. (2018). A sensitivity analysis was conducted to test the potential effect of applying the country specific grazing excretal N_2O EF calculated in Chadwick et al. (2018) as opposed to the IPCC (2019) default value for cattle in wet climates as an alternative.

The production-scaled emissions for each grazing treatment were determined by calculating the N_2O emissions for the control, fertiliser, urine and dung individually on a per hectare basis. The effect of urine and dung treatments were calculated by determining the number of urine patches and dung pats within a hectare, based on the stocking density of each grazing treatment. For the urine, the annual patch number from Dennis et al. (2011) was used to calculate the number of urine patches per hectare for the experimental period of 112 days. Similarly, existing data on the number of dung pats excreted per cow per day were used to determine dung pats per hectare (MacDiarmid and Watkin, 1972; MacDiarmid, 1972; Marsh and Campling, 1970). Production-scaled emissions for the improved treatment were calculated as the sum of the fertiliser, urine and dung N_2O emissions per hectare divided by the kg LWG produced per hectare for that grazing treatment. For the unimproved grazing treatment, the production-scaled emissions were calculated as the sum of the urine and dung N_2O emissions per hectare, along with the control emissions accounting for the remainder of the hectare. This value was then divided by the kg LWG produced per hectare from the unimproved treatment.

4.3 Results

4.3.1 Pasture quality

The metabolisable energy (ME) and crude protein content for grass from both grazing treatments are in Figure 4.2 below. The crude protein content was higher from the improved than the unimproved treatment throughout the experimental period, as was largely the case for ME. However, this was not statistically different for crude protein nor ME ($p > 0.05$). Similarly, pasture digestibility was comparable from both treatments.

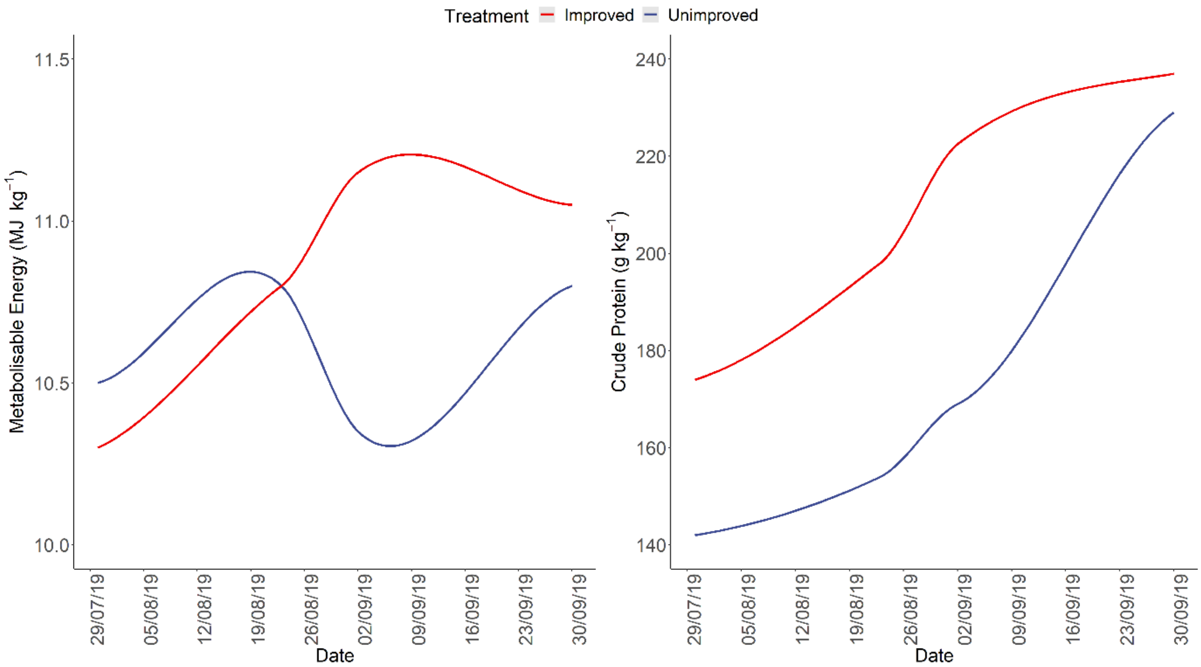

Figure 4.2. Metabolisable energy and crude protein content of the grazing treatments during the 2019 experimental period. Coloured lines represent bulked samples of each treatment.

4.3.2 Cattle liveweight gain

The DLWG (kg ha^{-1}) was higher from the unimproved treatment than the improved, recorded at 2.5 kg ha^{-1} and 2.2 kg ha^{-1}, respectively, though this difference was not significant ($p > 0.05$). The land area covered by bracken was removed for these calculations in order to calculate DLWG per area of grazeable pasture. Calculating DLWG including the bracken area reversed the trend but the difference remained not significant (data included in Appendix 4.3 - Table A). There was also no significant difference in the DLWG in the pilot year (2018), with a DLWG value of 1.7 kg ha^{-1} for the improved treatment and 1.6 kg ha^{-1} for the unimproved treatment (once again, excluding bracken areas).

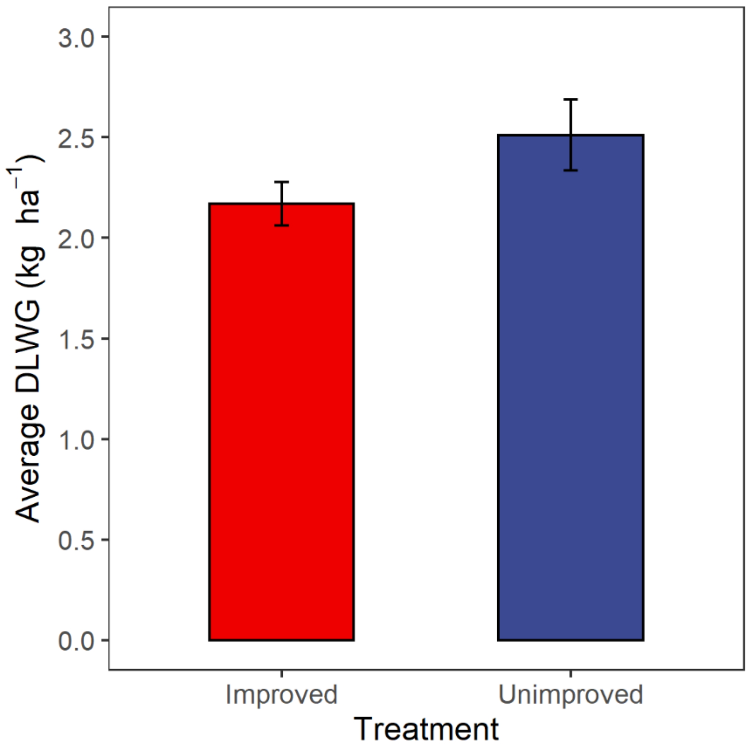

Figure 4.3. Cattle daily liveweight gain on a per hectare basis from both treatments during the 2019 grazing period (excluding bracken area). Error bars represent upper and lower standard error of the mean (SEM).

4.3.3 Cattle urine and dung characteristics

The N content of the urine was comparable for both grazing treatments, ranging from 5.75 to 7.41 g N L^{-1} for the cattle grazing the improved pasture, and from 5.20 to 7.80 g N L^{-1} for the cattle grazing the unimproved pasture in 2019 (Table 4.1). There was a significant difference in nitrate-N content between the grazing treatments, with values ranging between 0.44 and 0.60 mg L^{-1} for the improved treatment, and between 1.05 and 1.30 mg L^{-1} for the unimproved treatment. These values are within the range recorded by Chadwick et al. (2018). The ammonium-N content, however, is notably lower than the values generated by Chadwick et al. (2018). There were significantly different NH_4^+-N values for both grazing treatments, ranging between 33.06 - 38.06 mg L^{-1} for the improved urine and 41.79 - 46.70 mg L^{-1} for the unimproved urine samples.

For the dung samples, the total N content ranged from 0.30 to 0.38 g kg^{-1} in wet weight from the improved pasture, and from 0.34 to 0.36 g kg^{-1} in wet weight from the unimproved pasture. There was no significant difference in DM content of the dung samples between grazing treatments, with the improved dung DM content varying from 9.5 – 10.4% and the unimproved dung DM content varying from 9.6 – 9.9%.

Table 4.1. Average urine and dung characteristics (n = 3). Different letters indicate a significant different for the given characteristic between the treatments at $p < 0.05$.

Treatment	Urine			Dung	
	Total N (g N L^{-1})	Nitrate-N (mg L^{-1})	Ammonium-N (mg L^{-1})	Total N (g kg^{-1})	DM (%)
Improved	6.36^a	0.54^a	35.51^a	0.33^a	9.96^a
Unimproved	6.56^a	1.15^b	44.68^b	0.35^a	9.71^a

4.3.4 Nitrous oxide emissions

Following applications of fertiliser, urine, and dung, mean N_2O fluxes varied between the treatments during the monitoring period. Figure 4.4 below shows the N_2O fluxes throughout the monitored period, from July 1st to October 21st. There was a highly significant difference in N_2O emissions between the urine treatment and the other chamber treatments ($p < 0.001$). This was particularly evident for the unimproved grazing treatment. Following fertiliser, urine and dung applications to the chambers (timing of application annotated with an arrow in Figure 4.4), there was a clear peak in N_2O fluxes from the improved and unimproved grazing treatments at approximately 5 - 7 weeks post application. The urine application produced the highest N_2O fluxes from all of the inputs; with a peak reaching 195.4 µg N_2O-N m^{-2} h^{-1} and 652.9 µg N_2O-N m^{-2} h^{-1} from the improved and unimproved grazing treatments, respectively. The fluxes from the control (as well as the fertiliser and dung) were mostly zero. The N_2O flux data collected during the pilot year showed relatively low emissions from any of the inputs applied to the chambers, compared with the 2019 results (data not shown).

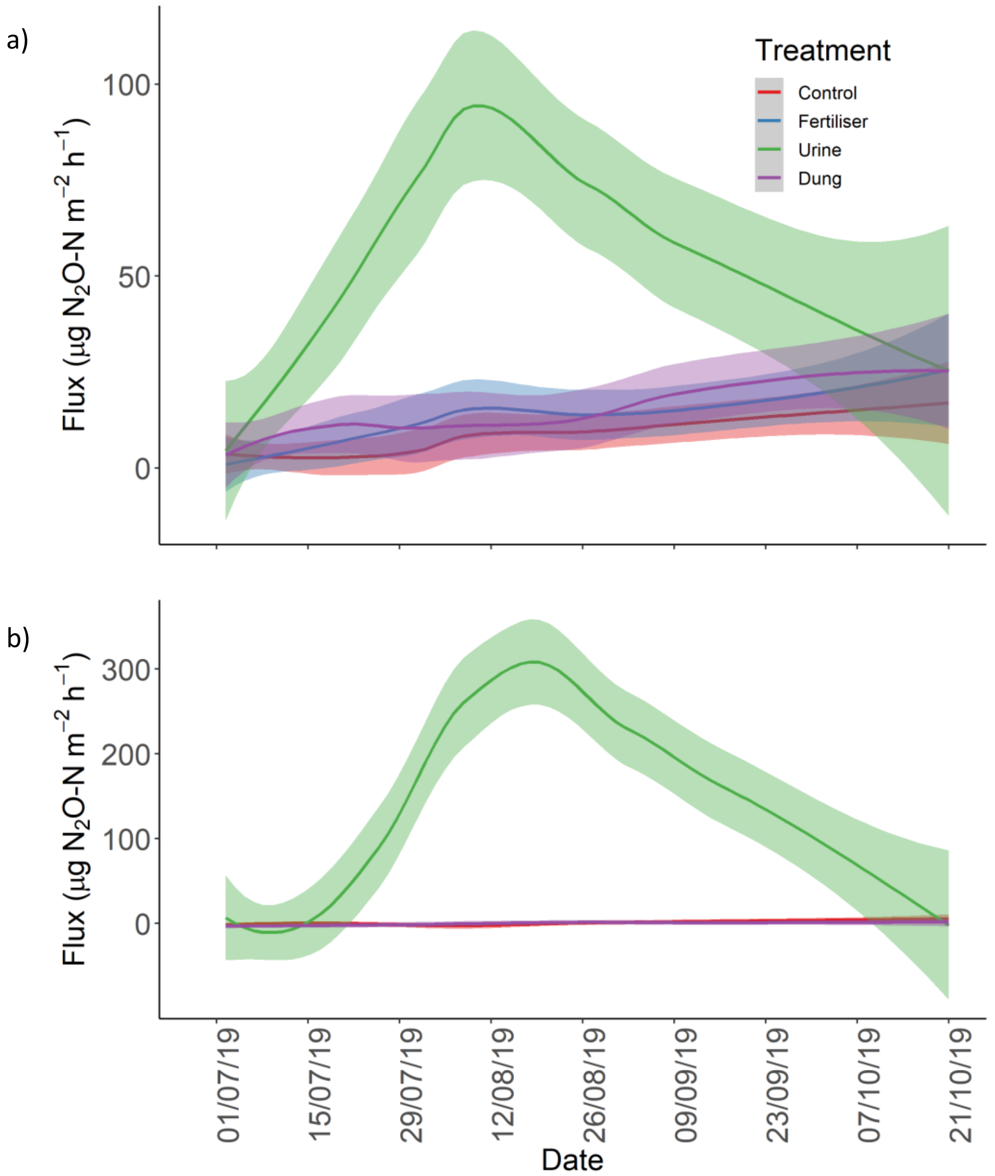

Figure 4.4. Nitrous oxide emissions (μg N_2O-N m^{-2} h^{-1}) from a) improved pasture and b) unimproved pasture for the monitored period in 2019. Coloured lines represent the treatment means (n = 4) and the shaded areas represent the upper and lower bounds of the SEM.

Cumulative annual N_2O emissions and EFs varied markedly between treatments (Table 4.2). The cumulative N_2O emissions of the control from the improved field treatment ranged from 0.13 – 0.47 kg N_2O-N ha^{-1} from the different replicate locations. There was a somewhat negative cumulative and EF value for the unimproved dung treatment. This corresponds to some of the cumulative N_2O emissions generated by Misselbrook et al. (2014). We calculated a combined excretal N_2O EF of 0.27% for the improved treatment, and 0.3% for the unimproved treatment.

Table 4.2. Cumulative nitrous oxide emissions and emission factors for the various treatments in 2019 (± SEM).

Treatment	Cumulative N_2O (kg N_2O-N ha^{-1})	EF (% of N applied)
Control	0.24 ± 0.08	-
Fertiliser	0.35 ± 0.15	0.22 ± 0.18
Urine - improved	1.34 ± 0.30	0.19 ± 0.04
Dung - improved	0.44 ± 0.16	0.39 ± 0.10
Control - unimproved	0.04 ± 0.03	-
Urine - unimproved	3.41 ± 0.50	0.56 ± 0.09
Dung - unimproved	-0.01 ± 0.04	-0.09 ± 0.02

4.3.5 Production-scaled emissions

Data collected on N_2O emissions produced per chamber following treatment application, along with LWG of the grazing regimes was used to generate production-scaled emissions for each pasture treatment. It was assumed that the cattle were grazing the pasture for 112 days for this calculation, which was the equivalent of the sampling period for calculating cumulative N_2O fluxes. The highest production-scaled emissions, expressed as g N_2O kg^{-1} LWG derived from the improved pasture at 1.75 g N_2O kg^{-1} LWG. In comparison, the production-scaled emissions was less than half from the unimproved pasture at 0.63 g N_2O kg^{-1} LWG.

4.3.6 LCA results and sensitivity analysis

The LCA results showed that the annual environmental footprint, quantified in terms of GWP, was 11.1 kg CO_2-eq kg^{-1} LWG for the improved grazing treatment and 8.2 kg CO_2-eq kg^{-1} LWG for the unimproved grazing treatment. For the experimental period, the GWP was 3.4 kg CO_2-eq kg^{-1} LWG for the improved treatment and 2.5 kg CO_2-eq kg^{-1} LWG for the unimproved treatment. Therefore, unimproved land resulted in lower emissions intensity than the improved treatment. The sensitivity analysis carried out tested the influence of utilising alternative EF default values for cattle excretion. The results of the sensitivity analysis showed that when using the IPCC default EF value (2019) for cattle excretion, compared to the UK country specific EF for cattle excretion value of 0.49% (Chadwick et al., 2018) adopted in our study, the total emissions per kg LWG would be only 1.89% higher from the improved grazing treatment and 2.07% from the unimproved grazing treatment than estimated.

4.4 Discussion

Cattle DLWG was comparable for both grazing treatments in 2018 and 2019. By contrast, Fraser et al. (2009) reported a significant difference in cattle performance when grazing different pasture types. However, caution should be taken when comparing their study to our experiment as the data collected by Fraser et al. (2009) was based on steer performance, whereas cattle DLWG was monitored from a herd of heifers in our study. Schwartzkopf-Genswein et al. (2002) showed that factors such as dry matter intake and average daily gain can differ significantly between steers and heifers, and despite their study being a penned experiment, it is an indication of potential variation in feeding behaviour between different stock types and sex.

In our study, the DLWG was recorded as 2.5 kg ha^{-1}, equivalent to 0.8 kg hd^{-1} from the unimproved grazing treatment. Fraser et al. (2011) assessed the LW change of Welsh Black heifers on semi-natural rough grazing land, and reported DLWG values between 0.2 - 0.5 kg hd^{-1} for 2003 - 2005, but a weight loss was recorded in 2007 and 2008. However, their experiment was conducted on land that was *Molinia*-dominated; an invasive grass species that is often viewed as forage of low nutritional value (Moorby et al., 2015). We recognise our limitations in that environmental conditions such as the exceptionally dry weather conditions in 2018 meant that the land's carrying capacity may have differed to a typical grazing season. A long-term experiment found no significant difference in lamb growth rates with less fertiliser application on upland permanent pasture (Yu et al., 2011). However, in their study, reduced fertiliser inputs led to a decline in stock carrying capacity and overall productivity. Our study would have benefitted from assessing the effect of lime and fertiliser input over a longer time period. Fertiliser input on previously unimproved land is not always successful, whilst it is well-known that fertiliser input can notably increase pasture productivity when applied to the appropriate sward (see Chapter 3).

While the attempt to partially reseed the improved treatment with a HSG variety in 2015 and 2016 appeared to be unsuccessful at early establishment, the later increase in pasture production (Williams et al., 2018), along with grass growth results following reseeding in Chapter 3 suggest that establishment did occur. Targeted breeding of grass varieties and forage crops in order to improve nutrient use efficiency is continually evolving (Marshall et al., 2012). These grass varieties can improve ruminant nutrition by enhancing the conversion of forage N to microbial N in livestock by increasing the readily available energy within the grasses (Ellis et al., 2012). However, our results imply that the cattle did not utilise the additional grass produced in the improved treatment of our study, as the grazing treatments did not significantly differ in cattle DLWG.

Gas fluxes were not monitored for a full year, which is the suggested period to provide IPCC compliant N_2O-N EFs. However, it is believed that we captured the main N_2O emission window by monitoring for 112 days as the values for all treatments were similar to the control values by the end of the sampling period. Several studies have shown that N_2O emission period is less than four months (de Klein et al., 2011; Marsden et al., 2018; Van der Weerden et al., 2011), with Kelliher et al. (2014) suggesting that in fact, most of the total N_2O emissions typically occur within the first month post N application. The N_2O fluxes from the pilot year were relatively low from the urine treatments in both improved and unimproved plots. The monitoring period during the pilot year in 2018 saw very dry weather as opposed to the latter year, and N_2O emissions differed between the two years, with greater fluxes produced in 2019. Production of N_2O fluxes in soils is largely influenced by environmental factors (Ammann et al., 2020), with this affecting gas transportation in soil and therefore, can lead to modulating the microbial activity. Cardenas et al. (2016) suggested that larger N_2O fluxes were associated with drier weather, with increased rainfall leading to N losses by leaching. Our result is to the contrary of this, with minimal N_2O fluxes produced during 2018, when rainfall was very low. We believe that the atypical dry season experienced in 2018 may have limited contact between the inputs (fertiliser, urine and dung) and the soil, therefore restricting, in particular, urine infiltration. High available N content deriving from urine excretion along with water deposited and accumulated within the soil profile following each urination event may lead to high N_2O emissions in 2019 (Cardenas et al., 2016).

Lime and fertiliser applications have shown to produce greater N_2O emissions (see Chapter 3; also Cardenas et al., 2010). Therefore, a higher N_2O EF would have been expected from the urine deriving from improved pasture than the urine deriving from unimproved pasture. Our urine N_2O EFs were as follows for the grazing treatments in 2019 - improved pasture calculated as 0.19% and unimproved pasture calculated as 0.56%. An important finding of our study was a significant difference in urine N content between the grazing regimes implemented, with a significantly greater nitrate and ammonium N content in the unimproved urine, leading to higher N_2O emissions and consequently, EFs from the urine deriving from unimproved pasture. Robust evidence suggests that changes in diet composition has an effect on N excretion (Cheng et al., 2017). Furthermore, high-sugar grasses have been shown to reduce N excretion in urine and dung, a finding reported by McAuliffe et al. (2020) in a similar study to ours conducted on lowland pasture. This may result in less N leaching and N_2O emissions produced (Foskolos and Moorby, 2017; Parsons et al., 2004). Lower N content and N_2O EFs from urine derived out of the improved pasture imply that the establishment of the HSG in the improved treatment was ultimately successful, and contributed to reduced emissions from urinary N excretion in the improved grazing treatment. Nevertheless, the total N content of both improved and unimproved urine samples were typical of cattle urine (Dijkstra et al., 2013; Gardiner et al., 2016; Misselbrook et al., 2016; Selbie et al., 2015), despite being slightly lower, on average, than the N content of cattle urine in Chadwick et al. (2018). The unimproved pasture urine N_2O EF from our study, however, is similar to that of the cattle derived EF for the UK generated by Chadwick et al. (2018).

A 60:40 split was assumed between the total N excreted in urine and dung in order to calculate a provisional excretal N_2O EF, as described in Webb and Misselbrook (2004). We estimate a combined excretal N_2O EF of 0.27% for the improved pasture and 0.3% for the unimproved pasture in our study, resulting in an overall mean combined excretal N_2O EF of 0.285% for the experiment. Chadwick et al. (2018) determined a country-specific combined excretal N_2O EF value for the UK of 0.49%. However, this greater N_2O EF was generated from a study implemented on lowland mineral soils, using urine and dung excreted by cattle grazing lowland pastures. The variability in N_2O EF across different studies has emphasised the importance of a country-specific combined excretal value such as the figure calculated in their study. However, the environmental conditions of their study contrast to those of our experiment. It could be argued that other factors, such as such soil type and diet are more significant in affecting GHG emissions, with previous research reporting a reduction in emission intensity from dairy cattle following the implementation of a low forage (high concentrate) diet (Ross et al., 2014). Kelliher et al. (2014) reported significantly lower mean EFs for cattle urine and dung from soils on sloped terrain than from lowland soils. This strengthens the requirement for specific urine and dung EFs that accounts for terrain.

The production-scaled emissions were higher from the improved than the unimproved grazing treatment. This was due to fertiliser input as well as a lack of corresponding improvement in LWG. Furthermore, there was no significant difference in pasture digestibility between the improved and unimproved pasture. We found that the improved grazing system produced a higher emission intensity (11.13 kg CO_2-eq kg^{-1} LWG) than the unimproved grazing system (8.23 kg CO_2-eq kg^{-1} LWG), based on the experimental monitoring period. This was primarily due to the inorganic fertiliser input along with the associated machinery fuel use. The annual emission intensity and therefore the environmental footprint of both systems highlighted in our study was relatively low in comparison with other reported values for cattle grazing systems (e.g. McAuliffe et al., 2018). However, it is difficult to compare the findings from our study with other studies as the results presented here have only been drawn in the context of year round grazing. Many studies have considered other factors that may contribute to the emission intensity of cattle on an annual basis, for example manure and feed from the housed phase (McAuliffe et al., 2018). Previous studies have highlighted that modifying the LCA approach using different conversion factors can significantly alter the results of a study (McAuliffe et al., 2018). In this experiment, however, we tested two conversion factors for cattle excretion, that is the a) country-specific combined excretal N_2O EF determined by Chadwick et al. (2018) and b) IPCC EF for cattle grazing N excretion in wet climate (IPCC, 2019). There was no significant difference between the GWP findings when substituting the conversion factors, thus not affecting the results.

4.5 Conclusions

Other studies have shown that pasture improvement options such as reseeding with a new grass variety along with lime and fertiliser inputs can increase grass growth. Relationships between lime and fertiliser applications, urine composition and N_2O emissions implies that improving upland pasture does not always stimulate greater N_2O production from the urine patch, with higher N_2O emissions produced from the urine deriving from the unimproved pasture diet. In fact, successful establishment of a HSG variety in the improved pasture prior to this experiment may have led to reduced N_2O emissions from this grazing treatment due to less urinary N excretion. A lack of significant difference in cattle DLWG following grazing improved and unimproved uplands suggests that the additional pasture produced in the improved grazing treatment was not fully utilised by the cattle. The development of a country specific grazing excretal N_2O EF has proved to be key in identifying N_2O sources along with variability between sites that may affect a calculated EF (Chadwick et al., 2018). An update to the 2006 IPCC inventory report has amended the default values for wet and dry climates. However, the combined excretal N_2O EF for our experiment (0.285%) was less than half that of the IPCC default (0.6%) and was lower than the country specific grazing excretal N_2O EF (0.49%). This evidence highlights the potential impacts of environmental factors such as slope, soil characteristics and climatic conditions on N_2O emissions from the soil, and consequently grazing excreta N_2O EF calculations. This may be of significance to upland cattle production systems when working to reduce their environmental impacts and specifically, production of greenhouse gas emissions, which will help meet the agricultural sector's emission reduction targets in the future. Further questions arise in terms of extrapolating the data from our study to other regions that may vary in terms of the environmental factors mentioned above.

4.6 References

AHDB, 2020. Fuel prices. [Online]. Available at: https://ahdb.org.uk/fuel-prices. Accessed February 20th, 2020.

AHDB, 2017. Nutrient Management Guide (RB209). Section 1: Principles of nutrient management and fertiliser use. [Online]. Available at:

https://ahdb.org.uk/documents/RB209/RB209_Section1_WEB_2017-12-01.pdf. Accessed 15th February, 2020.

Allen, A.G., Jarvis, S.C., Headon, D.M., 1996. Nitrous oxide emissions from soils due to inputs of nitrogen from excreta return by livestock on grazed grassland in the UK. *Soil Biology and Biochemistry 28,* 597-607.

Ammann, C., Neftel, A., Jocher, M., Fuhrer, J., Leifeld, J., 2020. Effect of management and weather variations on the greenhouse gas budget of two grasslands during a 10-year experiment. *Agriculture, Ecosystems and Environment 292*, 06814.

Battaglini, L., Bovolenta, S., Gusmeroli, F., Salvador, S., Sturaro, E., 2014. Environmental sustainability of Alpine livestock farms. *Italian Journal of Animal Science 13*(2), 3155.

Bell, M.J., Rees, R.M., Cloy, J.M., Topp, C.F.E., Bagnall, A., Chadwick, D.R., 2015. Nitrous oxide emissions from cattle excreta applied to a Scottish grassland: Effects of soil and climatic conditions and a nitrification inhibitor. *Science of the Total Environment 508*, 343-353.

Bhatia, A. Sasmal, S., Jain, N., Pathak, H., Kumar, R., Singh, A., 2010. Mitigating nitrous oxide emissions from soil under conventional and no-tillage in wheat using nitrification inhibitors. *Agriculture, Ecosystems and Environment 136,* 247-253.

Boorman, D., 2019. COSMOS-UK Henfaes data. [Email]. Sent 8th October, 2019. Accessed 12th October, 2019.

Cardenas, L.M., Misselbrook, T.M., Hodgson, C., Donovan, N., Gilhespy, S., Smith, K.A., Dhanoa, M.S., Chadwick, D., 2016. Effect of the application of cattle urine with or without the nitrification inhibitor DCD, and dung on greenhouse gas emissions from a UK grassland soil. *Agriculture, Ecosystems and Environment 235*, 229-241.

Cardenas, L.M., Thorman, R., Ashlee, N., Butler, M., Chadwick, D.R., Chambers, B., Cuttle, S., Donovan, N., Kingston, H., Lane, S., Scholefield, D., 2010. Quantifying annual N_2O emission fluxes from grazed grassland under a range of inorganic fertiliser nitrogen inputs. *Agriculture, Ecosystems and Environment 136*, 218-226.

Chadwick, D.R., Cardenas, L.M., Dhanoa, M.S. Donovan, N., Misselbrook, T., Williams, J.R., Thorman, R.E., McGeough, K.L., Watson, C.J., Bell, M., Anthony, S.G., Rees, R.M., 2018. The contribution of cattle urine and dung to nitrous oxide emissions: Quantification of country specific emission factors and implications for national inventories. *Science of the Total Environment 635*, 607-617.

Chadwick, D.R., Cardenas, L.M., Misselbrook, T.H., Smith, K.A., Rees, R.M., Watson, C.J., McGeough, K.L., Williams, J.R., Cloy, J.M., Thorman, R.E., Dhanoa, M.S., 2014. Optimizing chamber methods for measuring nitrous oxide emissions from plot-based agricultural experiments. *European Journal of Soil Science 65,* 295-307.

Chambers, B.J., Garwood, T.W.D., 1998. Lime loss rates from arable and grassland soils. *The Journal of Agricultural Science 131*, 455-464.

Cheng, L., McCormick, J., Hussein, A.N., Logan, C., Pacheco, D., Hodge, M.C., Edwards, G.R., 2017. Live weight gain, urinary nitrogen excretion and urination behaviour of dairy heifers grazing pasture, chicory and plantain. *The Journal of Agricultural Science 155*(4), 669.

Critchley, C.N.R., Adamson, H.F., McLean, B.M.L., Davies, O.D., 2008. Vegetation dynamics and livestock performance in system-scale studies of sheep and cattle grazing on degraded upland wet heath. *Agriculture, Ecosystems and Environment 128*(1-2), 59-67.

de Klein, C.A.M., Cameron, K.C., Di, H.J., Rys, G., Monaghan, R.M., Sherlock, R.R., 2011. Repeated annual use of the nitrification inhibitor dicyandiamide (DCD) does not alter its effectiveness in reducing N_2O emissions from cow urine. *Animal Feed Science and Technology 166-67*, 480-491.

de Vries, M., van Middelaar, C.E., de Boer, I.J.M., 2015. Comparing environmental impacts of beef production systems: a review of life cycle assessments. *Livestock Science 178*, 279-288.

Defra, 2019. Farming Statistics – Livestock Populations at 1 December 2018, UK. [Online]. Available at: https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/869061/structure-dec18-ukseries-28feb20.pdf. Accessed March 11th, 2020.

Defra, 2018. The British Survey of Fertiliser Practice – Fertiliser Use on Farm Crops for Crop Year 2018. [Online]. Available at:

https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/854404/fertiliseruse-report2018-20dec19.pdf. Accessed April 19th, 2020.

Defra, 2013. Farming Statistics – Livestock Populations at 1 December 2012, UK and England. [Online]. Available at:

https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/183227/Defra-stats-foodfarm-landuselivestock-farmstats-dec2012-130314.pdf. Accessed April 27th, 2020.

Defra, 2002. Economic conditions on cattle and sheep farms in the hills and uplands of the United Kingdom: statistical tables. London, UK: Economics (Farm Business) Division, Department for Environment, Food and Rural Affairs.

Dennis, S.J., Moir, J.L., Cameron, K.C., Di, H.J., Hennessy, D., Richards, K.G., 2011. Urine patch distribution under dairy grazing at three stocking rates in Ireland. *Irish Journal of Agricultural and Food Research 50*, 149-160.

DeRamus, H.A., Clement, T.C., Giampola, D.D., Dickison, P.C., 2003. Methane emissions of beef cattle on forages: Efficiency of grazing management systems. *Journal of Environmental Quality 32*, 269-277.

Dijkstra, J., Oenema, O., Van Groenigen, J.W., Spek, J.W., Van Vuuren, A.M., Bannink, A., 2013. Diet effects on urine composition of cattle and N_2O emissions. *Animal 7*, 292–302.

Eckard, R.J., Grainger, C., de Klein, C.A.M., 2010. Options for the abatement of methane and nitrous oxide from ruminant production: A review. *Livestock Science 130,* 47-56.

Eisler, M.C., Lee, M.R.F., Tarlton, J.F., Martin, G.B., Beddington, J., Dungait, J.A.J., Greathead, H., Lui, J., Mathew, S., Miller, H., Misselbrook, T., Murray, P., Vinod, V.K., Saun, R.V., Winter, M., 2014. Agriculture: step to sustainable livestock. *Nature 507,* 32-34.

Ellis, J.L., Dijkstra, J., France, J., Parsons, A.J., Edwards, G.R., Rasmussen, S., Kebreab, E., Bannink, A., 2012. Effect of high-sugar grasses on methane emissions simulated using a dynamic model. *Journal of Dairy Science 95*(1), 272-285.

European Environment Agency, 2015. Agriculture and climate change. [Online]. Available at: https://www.eea.europa.eu/signals/signals-2015/articles/agriculture-and-climate-change. Accessed February 21st, 2020.

Foskolos, A., Moorby, J., 2017. The use of high sugar grasses as a strategy to improve nitrogen utilization efficiency: a meta-analysis. *British Society of Animal Science Annual Conference: The Future of Animal Science*, 72.

Fraser, M.D., Moorby, J.M., Vale, J.E., Evans, D.M., 2014. Mixed grazing systems benefit both upland biodiversity and livestock production. *PloS One*, *9*(2).

Fraser, M.D., Theobald, V.J., Dhanoa, M.S., Davies, O.D., 2011. Impact on sward composition and stock performance of grazing Molinia-dominant grassland. *Agriculture, Ecosystems and Environment 144*(1), 102-106.

Fraser, M.D., Davies, D.A., Vale, J.E., Nute, G.R., Hallett, K.G., Richardson, R.I., Wright, I.A., 2009. Performance and meat quality of native and continental cross steers grazing improved upland pasture or semi-natural rough grazing. *Livestock Science 123(1)*, 70-82.

Fraser, M.D., Davies, D.A., Vale, J.E., Hirst, W.M., Wright, I.A., 2007. Effects on animal performance and sward composition of mixed and sequential grazing of permanent pasture by cattle and sheep. *Livestock Science 110*(3), 251-266.

Gardiner, C.A., Clough, T.J., Cameron, K.C., Di, H.J., Edwards, G.R., de Klein, C.A.M., 2016. Potential for forage diet manipulation in New Zealand pasture ecosystems to mitigate ruminant urine derived N_2O emissions: a review. *New Zealand Journal of Agricultural Research 59*(3), 301–317.

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

Gibbons, J.M., Williamson, J.C., Williams, A.P., Withers, P.J., Hockley, N., Harris, I.M., Hughes, J.W., Taylor, R.L., Jones, D.L., Healey, J.R., 2014. Sustainable nutrient management at field, farm and regional level: Soil testing, nutrient budgets and the trade-off between lime application and greenhouse gas emissions. *Agriculture, Ecosystems and Environment 188*, 48-56.

Goulding, K.W.T., 2016. Soil acidification and the importance of liming agricultural soils with particular reference to the United Kingdom. *Soil Use and Management 32*(3), 390-399.

Holland, J.E., Bennett, A.E., Newton, A.C., White, P.J., McKenzie, B.M., George, T.S., Pakeman, R.J., Bailey, J.S., Fornara, D.A., Hayes, R.C., 2018. Liming impacts on soils, crops and biodiversity in the UK: a review. *Science of the Total Environment 610*, 316-332.

IPCC, 2019. 2019 Refinement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories. [Online]. Available at:

https://www.ipcc-nggip.iges.or.jp/public/2019rf/pdf/4_Volume4/19R_V4_Ch11_Soils_N2O_CO2.pdf. Accessed February 20th, 2020.

IPCC, 2007. Summary for Policymakers. Climate Change 2007: Synthesis Report of Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, United Kingdom.

IPCC, 2006. 2006 IPCC guidelines for National Greenhouse Gas Inventories, prepared by the National Greenhouse Gas Inventories Programme. In: Eggleston, H.S., Buendia, L., Miwa, K., Ngara, T., Tanabe, K. (Eds.), 2006 IPCC Guidelines for National Greenhouse Gas Inventories. Published: IGES, Japan.

Kelliher, F.M., Cox, N., van der Weerden, T.J., de Klein, C.A.M.., Luo, J., Cameron, K.C., Di, H.J., Giltrap, D., Rys, G., 2014. Statistical analysis of nitrous oxide emission factors from pastoral agriculture field trials conducted in New Zealand. *Environmental Pollution 186,* 63-66.

Kowalczyk, A., Kuźniar, A. and Kostuch, M., 2014. Analysis of criteria for delimiting less favoured mountain areas (LFA). *Journal of Water and Land Development 22*(1), 17-24.

Lambert, M.G., Mackay, A.D., Devantier, B.P., McDougall, D.B., Barker, D.J., Park-Ng, Z.A., 2003. Redefining the production potential of hill pastures using fertiliser nitrogen. In: *Proceedings of the New Zealand Grassland Association 65*, 35-40.

MacDiarmid, B.N., Watkin, B.R., 1972. The cattle dung patch: 2. Effect of a dung patch on the chemical status of the soil, and ammonia nitrogen losses from the patch. *Grass and Forage Science* 27(1), 43-47.

MacDiarmid, B.N., 1972. The cattle dung patch: 3. Distribution and rate of decay of dung patches and their influence on grazing behaviour. *Journal of the British Grassland Society 27*, 43-54.

Mansfield, L., 2011. Upland Agriculture and the Environment. Badger Press Ltd.

Marsden, K.A., Holmberg, J.A., Jones, D.L., Chadwick, D.R., 2018. Sheep urine patch N_2O emissions are lower from extensively-managed than intensively-managed grasslands. *Agriculture, Ecosystems and Environment 265*, 264-274.

Marsden, K.A., Jones, D.L., Chadwick, D.R., 2016. The urine patch diffusional area: An important N2O source?. *Soil Biology and Biochemistry 92*, 161-170.

Marsh, R., Campling, R.C., 1970. Fouling of pastures by dung. *Herbage Abstracts 40*, 123-130.

Marshall, A.H., Abberton, M.T., Matthew, R., MacDuff, J.H., 2012. Genetic approaches to increase phosphorus uptake and utilisation of white clover (*Trifolium repens* L.). *Grassland Science in Europe 17,* 622-624.

McAuliffe, G.A., López-Aizpún, M., Blackwell, M.S.A., Castellano-Hinojosa, A., Darch, T., Evans, J., Horrocks, C., Le Cocq, K., Takahashi, T., Harris, P., Lee, M.R.F., 2020. Elucidating three-way interactions between soil, pasture and animals that regulate nitrous oxide emissions from temperate grazing systems. *Agriculture, Ecosystems and Environment 300*, 106978.

McAuliffe, G.A., Takahashi, T., Orr, R.J., Harris, P., Lee, M.R.F., 2018. Distributions of emissions intensity for individual beef cattle reared on pasture-based production systems. *Journal of Cleaner Production 171*, 1672-1680.

McLean, E.O., 1983. Soil pH and lime requirement. *Methods of soil analysis: Part 2 Chemical and Microbiological Properties 9*, 199-224.

Misselbrook, T., Fleming, H., Camp, V., Umstatter, C., Duthie, C.-A., Nicoll, L., Waterhouse, T., 2016. Automated monitoring of urination events from grazing cattle. *Agriculture, Ecosystems and Environment 230*, 191-198.

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

Moorby, J.M., Fleming, H.R., Theobald, V.J., Fraser, M.D., 2015. Can live weight be used a proxy for enteric methane emissions from pasture-fed sheep? *Scientific reports 5*(1), 1-9.

Newbould, P., 1974. The improvement of hill pastures for agriculture: A review part 1. *Grass and Forage Science 29*(4), 241-247.

Nix, J., 2019. John Nix Pocketbook for Farm Management. 49th Edition.

Professional Agriculture Analysis Group, 2015. Collation of data from routine soil analysis in the UK 2014/15. [Online]. Available at:

http://www.nutrientmanagement.org/nfu-online/triedtested/external-info/paag/paag-report-2014-15/. Accessed March 11th, 2020.

Parsons, A.J., Rasmussen, S., Xue, H., Newman, J.A., Anderson, C.B., Cosgrove, G.P., 2004. Some 'high sugar grasses' don't like it hot. *Proceedings of the New Zealand Grassland Association 66,* 265-271.

R Core Team, 2019. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. [Online]. Available at: https://www.R-project.org. Accessed April 26th, 2020.

Reed, M.S., Arblaser, K., Bullock, C., Burton, R.J.F., Davies, A.L., Holden, J., Hubacek, K., May, R., Mitchley, J., Morris, J., Nainggolan, D., Potter, C., Quinn, C.H., Swales, V., Thorp, S., 2009. Using scenarios to explore UK upland futures. *Futures 41,* 619-630.

Ross, S.A., Chagunda, M.G.G., Topp, C.F.E., 2014. Effect of cattle genotype and feeding regime on greenhouse gas emissions intensity in high producing dairy cows. *Livestock Science 170,* 158-171.

Saggar, S., Jha, N., Deslippe, J., Bolan, N.S., Lo, J., Giltrap, D.L., Kim, D.G., Zaman, M., Tillman, R.W., 2013. Denitrification and N2O:N2 production in temperate grasslands: processes, measurements, modelling and mitigating negative impacts. *Science of the Total Environment 465,* 173-195.

Salisbury, E., Thistlethwaite, G., Young, K., Claxton, R., Cardenas, L., 2014. Greenhouse Gas Inventories for England, Scotland, Wales and Northern Ireland: 1990-2012. Department of Energy and Climate Change, London, UK, 1-111.

Schwartzkopf-Genswein, K.S., Atwood, S., McAllister, T.A., 2002. Relationships between bunk attendance, intake and performance of steers and heifers on varying feeding regimes. *Applied Animal Behaviour Science 76*(3), 179-188.

Selbie, D.R., Buckthought, L.E., Shepherd, M.A., 2015. Chapter four-the challenge of the urine patch for managing nitrogen in grazed pasture systems. *Advances in Agronomy 129*, 229–292.

Scheer, C., Rowlings, D.W., Firrel, M., Deuter, P., Morris, S., Grace, P.R., 2014. Impact of nitrification inhibitor (DMPP) on soil nitrous oxide emissions from an intensive broccoli production system in sub-tropical Australia. *Soil Biology and Biochemistry 77*, 243-251.

Skiba, U., Jones, S.K., Dragosits, U., Drewer, J., Fowler, D., Rees, R.M., Pappa, V.A., Cardenas, L., Chadwick, D., Yamulki, S., Manning, A.J., 2012. UK emissions of the greenhouse gas nitrous oxide. *Philosophical Transactions of the Royal Society B 367,* 1175-1185.

Smith, K.A., Dobbie, K.E., 2001. The impact of sampling frequency and sampling times on chamber-based measurements of N_2O emissions from fertilized soils. *Global Change Biology 7*(8), 933-945.

Stejskalová, M., Hejcmanová, P., Pavlů, V., Hejcman, M., 2013. Grazing behavior and performance of beef cattle as a function of sward structure and herbage quality under rotational and continuous stocking on species-rich upland pasture. *Animal Science Journal 84*(8), 622-629.

Styles, D., Gibbons, J., Williams, A.P., Stichnothe, H., Chadwick, D.R., Healey J.R., 2015. Cattle feed or bioenergy? Consequential life cycle assessment of biogas feedstock options on dairy farms. *Global Change Biology Bioenergy 7*(5), 1034-1049.

Thornton, P.K., 2010. Livestock production: recent trends, future prospects. *Philosophical Transactions of the Royal Society B: Biological Sciences 365*(1554), 2853-2867.

Tubiello, F.N., Salvatore, M., Rossi, S., Ferrara, A., Fitton, N., Smith, P., 2013. The FAOSTAT database of greenhouse gas emissions from agriculture. *Environment Research Letters 8*(1), 015009.

United Nations, 2019. Growing at a slower pace, world population is expected to reach 9.7 billion in 2050 and could peak at nearly 11 billion around 2100. Department of Economic and Social Affairs, United Nations. [Online]. Available at: www.un.org/development/desa/en/news/population/world-population-prospects-2019.html. Accessed January 20th, 2020.

Van der Weerden, T.J., Luo, J., de Klein, C.A.M., Hoogendoorn, C.J., Littlejohn, R.P., Rys, G.J., 2011. Disaggregating nitrous oxide emission factors for ruminant urine and dung deposited onto pasture soils. *Agriculture, Ecosystems and Environment 141*, 426-436.

Van Groenigen, J.W., Kuikman, P.J., de Groot, W.J.M., Velthof, G.L., 2005. Nitrous oxide emission from urine-treated soil as influenced by urine composition and soil physical conditions. *Soil Biology Biochemistry 37*(3), 463-473.

Wakelin, S.A., Clough, T.J., Gerard, E.M., O'Callaghan, M., 2013. Impact of short-interval, repeat application of dicyandiamide on soil N transformation in urine patches. *Agriculture, Ecosystems and Environment 167*, 60-70.

Webb, J., Misselbrook, T.H., 2004. A mass-flow model of ammonia emissions from UK livestock production. *Atmospheric Environment 38,* 2163-2176.

Welten, B.G., Ledgard, S.F., Schipper, L.A., Waller, J.E., Kear, M.J., Dexter, M.M., 2013. Effects of prolonged oral administration of dicyandiamide to dairy heifers on excretion in urine and efficacy in soil. *Agriculture, Ecosystems and Environment 173,* 28-36.

Williams, A.P., Gibbons, J., Styles, D., Chadwick, D.R., 2017. Sustainable intensification of an upland sheep system: evidence from Henfaes Research Station, Bangor University. Defra Publications, London, UK.

Wuebbles, D.J., 2009. Nitrous oxide: no laughing matter. *Science 326*(5949), 56-57.

Yu, Y.W., Fraser, M.D., Evans, J.G., 2011. Long-term effects on sward composition and animal performance of reducing fertilizer inputs to upland permanent pasture. *Grass and Forage Science 66*(1), 138-151.

Zaman, M., Blennerhassett, J.D., 2010. Effects of different rates of urease and nitrification inhibitors on gaseous emissions of ammonia and nitrous oxide, nitrate leaching and pasture production from urine patches in an intensive grazed pasture system. *Agriculture, Ecosystems and Environment 136*(3-4), 236-246.

Zaman, M., Blennerhassett, J.D., Singh, J., 2009. Effect of urease and nitrification inhibitors on N transformation, gaseous emissions of ammonia and nitrous oxide and N uptake in grazed pasture systems. *Soil Biology and Biochemistry 41*, 1270-80.

5. Can increasing upland pasture productivity allow Wales to maintain upland beef production and meet its afforestation targets?

Abstract

Agriculture and forestry are currently the two dominant land uses in the Welsh uplands. Livestock production in particular significantly contributes to climate change by producing methane and nitrous oxide emissions while woodland is considered an effective method of climate change mitigation. There is increasing interest in the extent as to what the uplands will be managed for in the future, with fluctuation in agricultural activity in these areas. This will be largely driven by changes in policy and the availability of subsidies and incentives. In this study, we identified areas for future expansion of beef production in the uplands, based on data from Chapters 3 and 4. We also mapped current forestry land, along with areas of Wales that could be considered for afforestation in the future. Our results highlight that in 2018, potential competition for land between beef production and forestry was low. However, future projections estimated that 54.9% of the land identified for potential beef production would be in competition with new woodland creation. With considerable pressure to reduce greenhouse gas (GHG) emissions from agriculture, sustainably intensifying beef production in some areas in order to spare other land for afforestation, and offsetting beef GHGs in doing so could be a potential solution to this. Furthermore, improving individual cattle performance, for instance, via increasing rearing percentage, offspring growth rates, and carcass weights, could lead to sufficient and/or increased beef production from less stock. This would also contribute to the challenge of mitigating climate change in the future.

5.1 Introduction

From the mid-20th century, the United Kingdom's agricultural policy generally promoted land improvement as a means of increasing agricultural production (Mansfield, 2011). As described in Chapter 2, following the Second World War, there was a large emphasis on increasing food production in upland agriculture to ensure food was affordable and to feed a growing population. Upland agricultural policy has evolved since this period, with a higher profile for environmental management of the land since the 1990s. However, with increased population growth, policies that compromise food security may yet be seen negatively (Hallett et al., 2017). Establishment of the Forestry Commission by the Forestry Act in 1919 promoted the planting of woodlands to increase timber supply and the growth of the forest industries (Oosthoek, 2000). This led to a large emphasis on the expansion of plantations, in particular in the uplands (Pringle, 1994), as land of better physical conditions was in use for agriculture (Oosthoek, 2000). Tree planting in UK uplands was particularly high during the 1920s and 1930s (Mansfield, 2011). However, forestry schemes specifically targeting uplands were limited towards the end of the 20th century (Illbery and Kidd, 1992). In more recent times, UK new planting rates have decreased. Provisional figures by the Forestry Commission (2020) have demonstrated that only 13,460 ha of new woodland was created in the UK in 2019-20, less than half the annual target of 30,000 ha. Of this new woodland, as little as 80 ha was planted in Wales, which equates to 4% of the annual target set by Welsh Government (2018c).

As previously mentioned, livestock production is responsible for ca. 14.5% of anthropogenic greenhouse gas (GHG) emissions produced globally (FAO, 2019). Greenhouse gas emissions from livestock equated to 63% of UK total agricultural emissions in 2018 (Committee on Climate Change, 2018). Enteric fermentation along with livestock waste via excretion and manure management was primarily responsible for the production of methane (CH_4) and nitrous oxide (N_2O) emissions. The Welsh Government is committed to mitigating GHG production, with a target of 95% reduction in GHGs produced relative to those produced in 1990 in Wales (Committee on Climate Change, 2019). However, Wales has set out in its Economic Action Plan the aim to further reduce GHG production in order to reach net zero by 2050 (Welsh Government, 2019c).

Broader policies, not specifically targeted at reducing GHG emissions, may still influence the emissions from agriculture. For instance, proposals are in place to extend land areas classified as Nitrogen Vulnerable Zones (NVZ), which may result in an all-Wales NVZ (Welsh Government, 2020b). If implemented, this could reduce fertiliser use, and could also lead to large reductions in associated GHG emissions. However, this is unlikely to have a significant effect on upland, extensive agricultural systems due to relatively low fertiliser use in these areas (Goulding et al., 2000). As stated earlier, as well as actions to mitigate GHG production, off-setting emissions through sequestration is acknowledged to play an important role in achieving the net zero target.

Alternative uses for agricultural land, afforestation in particular, is deemed necessary to achieve the net zero target by 2050, with a proposal to use approximately 22% of UK agricultural land for this purpose (Committee on Climate Change, 2020a). Soil carbon sequestration has been considered the most effective method in mitigating agricultural GHG production (Sousanna et al., 2010). However, there has been some debate regarding the capacity of carbon sequestration in grasslands due to high levels of variation dependent on management, e.g. targeted fertiliser application along with low grazing intensities have shown to increase soil organic carbon stocks in the past (Conant et al., 2001). Furthermore, grasslands are unable to continue to increase their carbon store once their capacity has been maximised (Committee on Climate Change, 2020a). Therefore, converting existing grasslands to forest has been identified as an additional way of increasing the amount of carbon sequestered and consequently reducing net GHG emissions (Committee on Climate Change, 2020a). A possible scenario to achieve this aim is the proposal to increase UK tree planting to at least 30,000 ha per year by 2050. From a Welsh perspective, this equates to planting 152,000 ha of new woodland in total by 2050 (Committee on Climate Change, 2020b). It is inevitable that afforestation to this extent will require agricultural land in many regions across Wales. Anecdotally, it appears that land graded as 3b in the Agricultural Land Classification system, classified as moderate quality agricultural land, is often proposed as potential land for future afforestation in Wales. Furthermore, it is likely that land that is production-limited from an agricultural perspective due to bio-physical constraints, e.g. uplands, could once again be considered particularly suitable for tree planting (Committee on Climate Change, 2020b). There is much debate as to for what purposes such land should be managed for in the future e.g. for biodiversity, ecosystem services, afforestation, agricultural intensification or conversion to bioenergy crops (Hardaker et al., 2020; Reed et al., 2009). Costs may be higher in upland regions due to establishment and management costs, e.g. due to access difficulties. This should be factored into account when assessing land suitability for afforestation in the future. However, in the scenario proposed by the Committee on Climate Change (2020b), it is certain that some of the uplands currently used as grazing land for livestock production will be transitioned for afforestation.

To facilitate uptake by landowners, economic incentives for woodland creation and restoration currently exist in Wales e.g. the Glastir Woodland Creation scheme, part of Wales' agri-environment scheme (Welsh Government 2020c; 2018b). The private costs and revenue of woodland creation proposed by the Committee on Climate Change (2020a) fluctuate depending on other factors. However, consideration should also be given to the potential negative unintended consequences from such policy changes, such as the displacement of the impacts of food production elsewhere.

A number of location-specific characteristics influences the potential of viable beef production within the uplands of Wales. These include variation in slope, aspect, elevation, precipitation and temperature, which affect growing and grazing seasons, and consequently agricultural production from these areas (Mansfield, 2011). However, whilst such factors cannot be altered, the management of such systems can notably change the grass production potential of upland areas, and therefore the viability of beef production. Furthermore, altering the management of cattle to improve the economic viability of beef production will impact on both total carbon (GHG) emissions from such systems and the carbon intensity of beef produced; which should also be considered in the discussion about reconfiguration of the uplands. Exploring how management changes can increase the beef production efficiency and/or capacity of such land will help inform the discussion as to whether such land should be used for cattle systems or for meeting Wales' afforestation targets.

As explained above, tree planting in recent years has been inadequate in relation to governmental targets. However, there is a clear policy drive for afforestation rates to rapidly increase. It is therefore anticipated that this will drive land use change in Wales' uplands. The aim of this assessment was to examine whether the agriculture and forestry sectors will be in competition for land in the future due to calls for new woodland creation in the uplands. One objective was to identify the extent of similar locations to the field sites in Chapters 3 and 4 across Wales, where increased agricultural productivity could be delivered through altered management regimes, as well as to quantify the potential agricultural productivity in doing so. The second objective was to determine land areas currently managed as woodland along with areas identified as potential forestry land in the future. Lastly, to assess the overlap of land identified for the first two objectives and examine whether such land would be (more) suitable for livestock production or afforestation, and the implications for both sectors.

5.2 Materials and methods

5.2.1 Upscaling field trial data

The results were upscaled from two field trials that were set up at Bangor University's research station, Henfaes Research Centre, Abergwyngregyn, north Wales (approximately 250 m a.s.l) in order to explore upland pasture productivity (Chapter 3) and to determine the effect of land improvement in the form of targeted lime and fertiliser applications on pasture productivity and cattle performance (Chapter 4). The experimental results that were included for the upscaling from each chapter are summarised in Figure 5.1. This produced values for potential grass production following different pasture improvement options and the associated N_2O emissions from the soil, along with cattle liveweight gain. For the potential grass production, the baseline value of 4.5 t DM ha^{-1} dry matter yield from the Nutrient Management Guide (RB209), 8th edition (AHDB, 2017) was applied to the unimproved (existing long-term permanent pasture referred to as the control in Chapters 3 and 4), and used to calculate the pasture production percentage increase for the other treatments based on mean daily grass production. The total potential cattle liveweight gain was determined based on the potential pasture production percentage increase of each treatment.

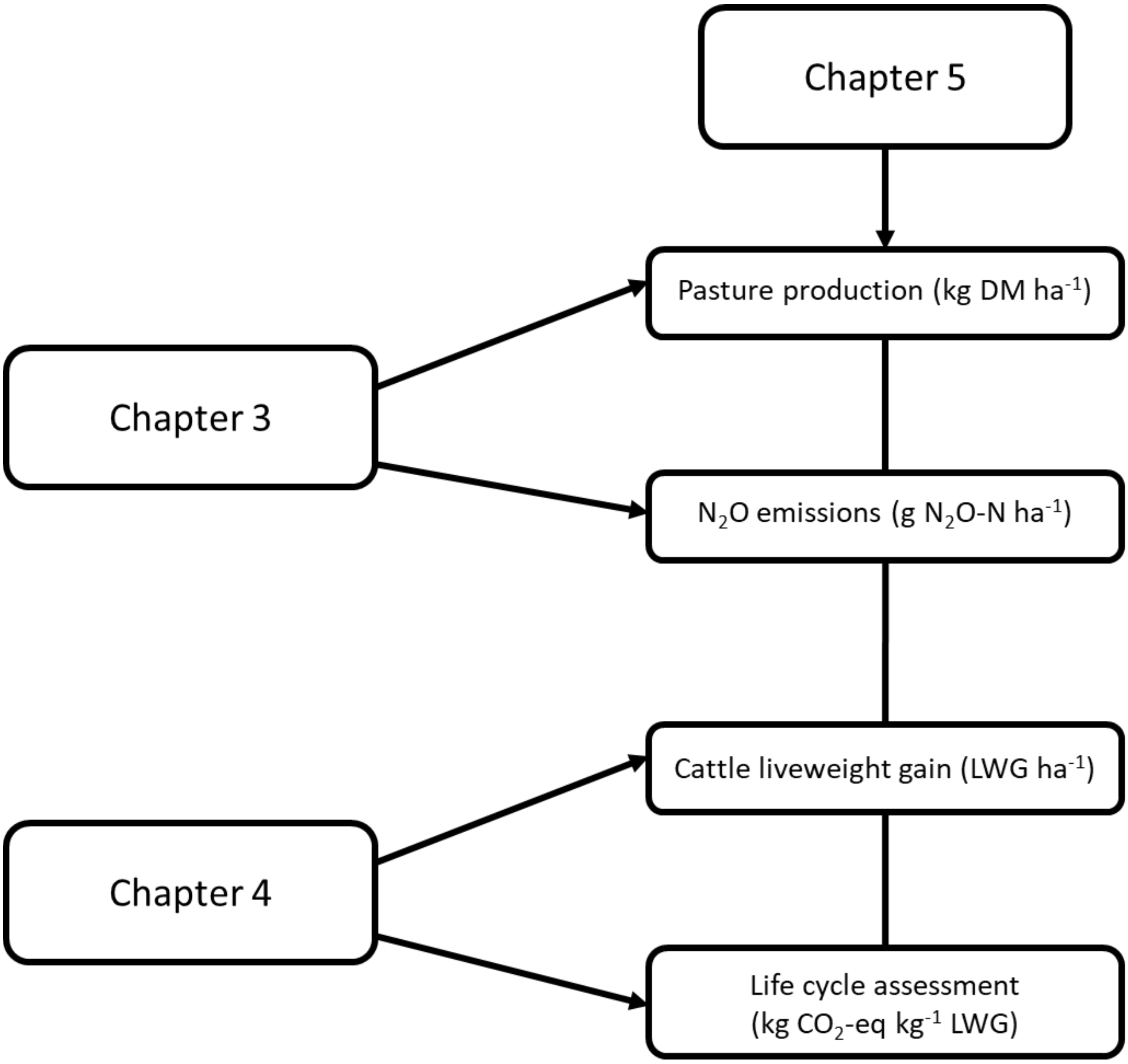

Figure 5.1. Experimental results from Chapters 3 and 4 used for upscaling that will be used to generate Chapter 5 results.

5.2.2 Spatial selection for site matching

A range of categorical and continuous data sources were used as parameters to match similar sites to that of the field trials. The core data used, as well as the parameters for selection, are summarised in Table 5.1. Data sources were selected based on their representation in relation to Wales. For the agricultural land classification categorical variable, the location of the field site was assessed as Agricultural Land Classification (ALC) Grade 4. For this reason, areas classified as 3b and 4 land (3b being moderate quality agricultural land, and 4 being poor quality agricultural land) (Welsh Government, 2020a) were considered, to ensure that the selection was not too restrictive. This variable was combined with soil type and land cover variables to match to the field trial site. For the continuous data for selection, all variables were normalised to a mean of zero and standard deviation of one in order to make them comparable. The Euclidean distance from the field site was estimated for each variable. This common distance measure often used for calculating distance of quantitative data in four-dimensional space was deemed appropriate for identifying how similar or divergent each continuous variable was to the field site. The distance was calculated between the sites (equation = $\mathrm{sqrt}((\chi 1 - y1)^2)$, where $\chi 1$ was the constant variable at the field site and $y1$ was the variable at other locations). Each Euclidean distance layer was then combined to produce a single distance layer for the continuous variables. Lastly, the distance layer was combined with the categorical selection to identify how much/the degree of land that matched within 0.25, 0.5 and 1.0 standard deviation (SD). The SDs are calculated as distance units, where 0.25 distance units = 0.25 SD, 0.5 distance units = 0.5 SD and 1.0 distance units = 1.0 SD. Figure 5.2 illustrates the development of the spatial data layers carried out using ArcGIS software (ESRI, 2020) leading to the final selection of locations across Wales.

Table 5.1. Data sources for the chapter.

Variable		Parameters	Data source	Resolution/Scale
Categorical	Continuous			
Agricultural Land Classification (ALC)	-	3b and 4 land	Predictive Agricultural Land Classification for Wales (Welsh Government, 2019b)	1:50,000
Soil type	-	Freely draining acid loamy soils over rock	National Soil Map (NATMAP) Soilscapes (National Soils Resources Institute - Cranfield University, 2006)	1:250,000
Land cover	-	Acid grassland	Land cover map for Wales (CEH, 2017)	0.5 ha
-	Elevation	-	OS 10 m Land-Form Profile (Ordnance Survey, 2009)	1:10,000 (10 m), 5 km × 5 km
	Slope	-	OS 10 m Land-Form Profile (Ordnance Survey, 2009)	1:10,000 (10 m), 5 km × 5 km
-	Aspect	-	OS 10 m Land-Form Profile (Ordnance Survey, 2009)	1:10,000 (10 m), 5 km × 5 km
-	Average annual rainfall	-	NATMAP SEISMIC Agroclimatic data (National Soils Resources Institute - Cranfield University, 2002)	5 km × 5 km
	Total cattle numbers	Cattle Tracing System values for beef cows	Agricultural Small Area Statistics (Welsh Government, 2018a)	1:2,500
-	Current forestry land use distribution	-	National Inventory of Woodland and Trees (Natural Resources Wales, 2018)	1:25,000
-	Potential extent of future afforestation	Woodland creation scores from 0-37*	Glastir Woodland Creation Opportunities map (Welsh Government, 2020c)	20 m × 20 m

*The land is scored from 0-37, depending on the suitability for new woodland creation (0 being least suitable, 37 being most suitable).

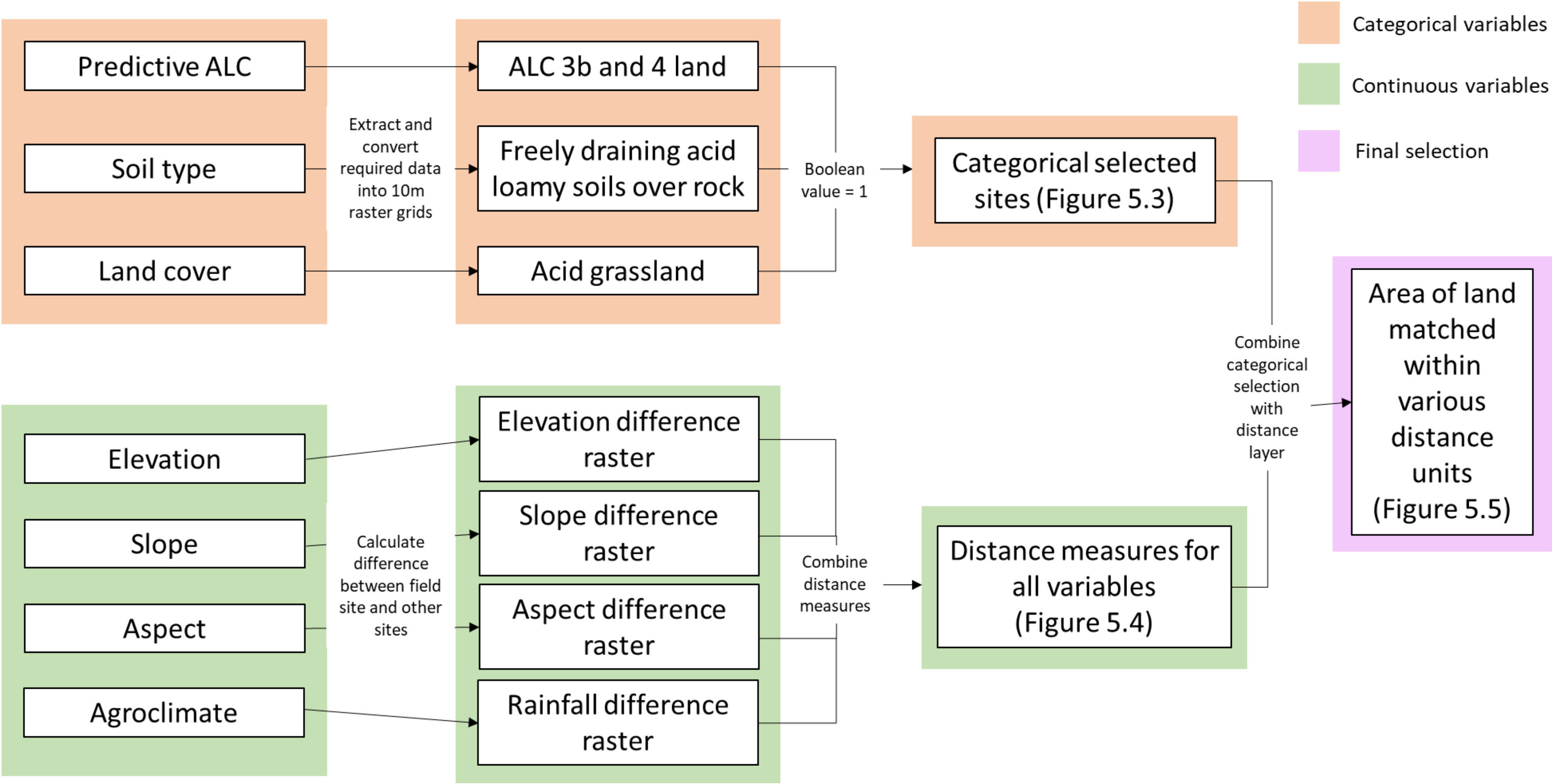

Figure 5.2. Flow diagram of spatial data layer development for location selection.

The results of the field trials summarised in Figure 5.1 and expressed on a per hectare basis was applied for the categorical, continuous and final selection land areas by multiplying the number of hectares suitable by the experimental results. The resulting N_2O values were then converted from N_2O-N to N_2O, expressed as CO_2 equivalent, and used to calculate the proportion of Wales' agricultural GHG production.

Data on beef cow numbers across Wales, obtained from the Cattle Tracing System for the Agricultural Small Area Statistics for Wales (Welsh Government, 2018a), were used to determine spatial patterns in current cattle production, which could then be used alongside the locations selected to estimate the carrying capacity for cattle production from the uplands under different management regimes. Data on the current distribution of forestry land and potential extent of afforestation in the future were used along with the categorical selection to produce a binary classification in order to quantify the overlapping land areas.

5.3 Results

5.3.1 Categorical selection

The categorical selection displayed in Figure 5.3 below is the upper bound of areas that could be improved or considered for afforestation. A total of 62,676 ha of land, which is 7.3% of the land classified as Less Favoured Areas (LFA) and Severely Disadvantaged Areas (SDA) permanent grass in Wales (Welsh Government, 2019a), were identified, based on the three categorical parameters highlighted in Table 5.1 and Figure 5.2.

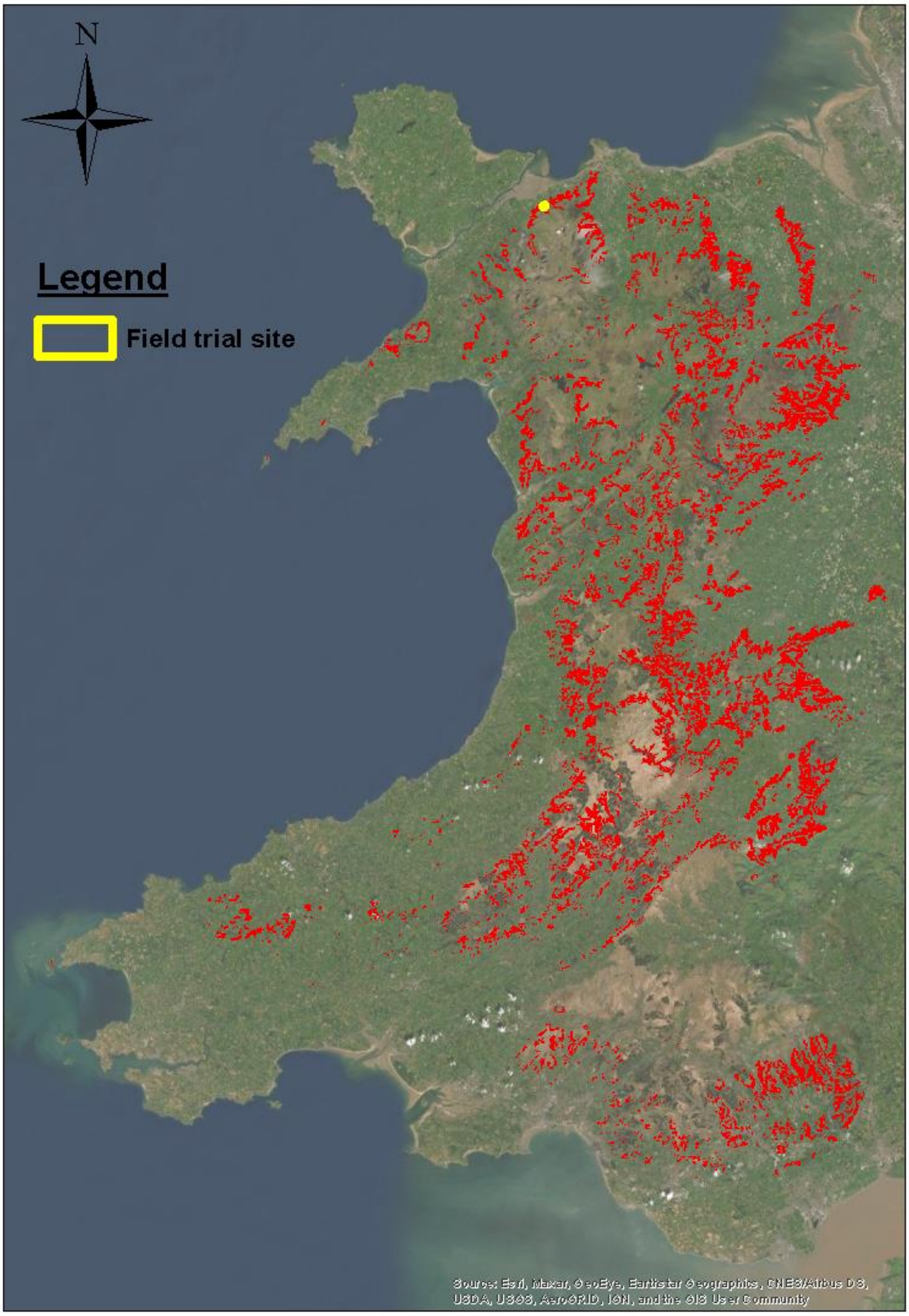

Figure 5.3. Locations in Wales identified via categorical selection as matching the parameters of the field trial site (ALC 3b and 4 land, freely draining acid loamy soils over rock and acid grassland).

Upscaling the N_2O results associated with varying management options from Chapter 3 based on the categorical selection area (Figure 5.3) quantified potential N_2O emissions in proportion to national GHG production figures. The N_2O emissions equate to 0.141% from the unimproved pasture and 0.135% from pasture receiving lime and fertiliser input of Wales' agricultural sector GHG emissions in 2016 (Welsh Government, 2019c), based on the categorical selection land area. If all the area selected in Figure 5.3 was unimproved, but used for cattle production, we estimate that the annual environmental footprint calculated via an LCA method would be 471.27 Gg CO_2-eq. In contrast, if the categorical selection land area was managed as an improved grazing system (lime and fertiliser input), the annual environmental footprint calculated via an LCA method would equate to 551.12 Gg CO_2-eq, a 16.94% increase from the unimproved footprint. Note that these values will include emissions not produced from the agricultural sector.

Table 5.2. Potential grass production and nitrous oxide emissions, along with cattle liveweight gain based on the categorical selection experimental results. Values are expressed as an annual total. Total N_2O emissions (N_2O-N converted to N_2O) is expressed as CO_2 equivalent, and used to calculate the proportion of Wales' agricultural GHG production.

Management	Total pasture production (Gg DM)	Total N_2O emissions (Mg N_2O-N)	Proportion of Wales' agricultural GHG production (%)	Total actual cattle liveweight gain based on experimental results (Gg)	Total potential cattle liveweight gain based on potential pasture production (Gg)
Unimproved	282.04	17.17	0.141	57.47	
Lime and fertiliser input only	286.71	16.45	0.135	49.65	58.42
Rotovate, spring forage rape followed by autumn grass reseed, lime and fertiliser input	-	113.56	0.930	-	-
Plough, spring grass reseed, lime and fertiliser input	609.97	132.80	1.088	-	124.30
Rotovate, spring grass reseed, lime and fertiliser input	571.01	185.86	1.522	-	116.36

5.3.2 Continuous selection

The Euclidean distance from the field site was calculated for each continuous variable (see Appendix 5.3, Figures A-D), with the variables combined to produce the final continuous Euclidean distance layer (Figure 5.4). Without including the categorical selection, a total of 68,908 ha matched with the field trial site within 0.25 SD (equating to 8% total LFA and SDA areas in Wales), a total of 637,612 ha within 0.5 SD (74% of LFA and SDA areas in Wales) and 1,987,233 ha within 1.0 SD (230.6% of LFA and SDA areas in Wales). Potential pasture production, N_2O emissions and cattle liveweight gain is expressed for each SD measure in Tables A and B in Appendix 5.3. The N_2O emissions estimated for the selection within 0.25 SD total to 0.155% for the unimproved and 0.148% for the improved treatment of Wales' agricultural sector emissions in 2016 (Welsh Government, 2019c). For the selection within 0.5 SD, this equates to 1.430% and 1.370% of Wales' agricultural GHG emissions in 2016 from the unimproved and improved treatments respectively. The area matched within 1.0 SD equates to 4.457% from the unimproved and 4.271% from the improved pasture of Wales' agricultural emissions in 2016.

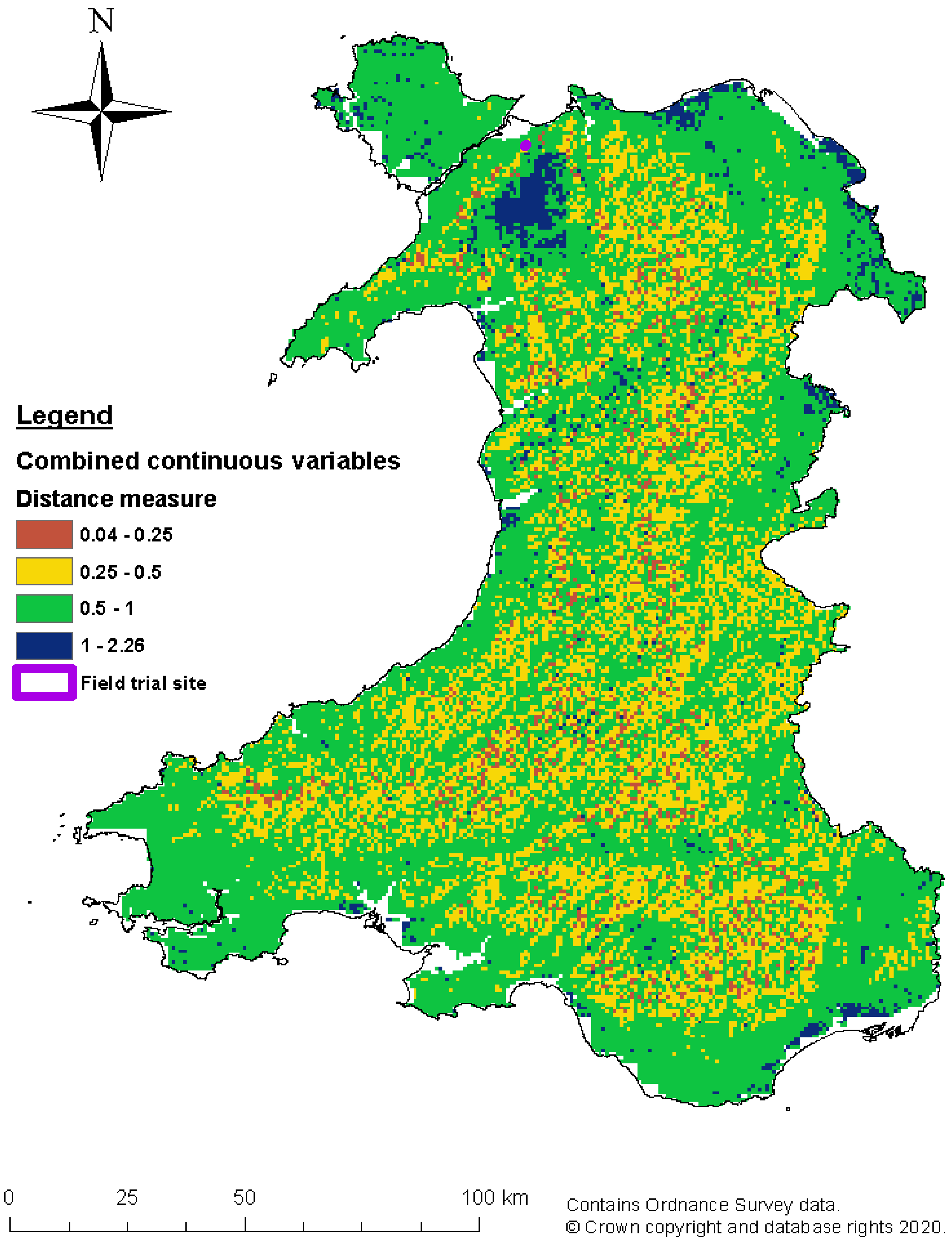

Figure 5.4. Final continuous distance layer ranging from 0.04 to 2.26 SD. The SD measure is not calculated for some areas around Wales' boundary due to lack of rainfall data for these areas.

5.3.3 Final selection

The final selection was expressed as a combination of the categorical and continuous selection. We can conclude that a total of 33,847 ha matched within 0.25 SD (3.4% of Wales' LFA and SDA areas), 205,857 ha within 0.5 SD (23.9% of Wales' LFA and SDA land areas) and 398,255 ha within 1.0 SD (46.2% of Wales' LFA and SDA land areas).

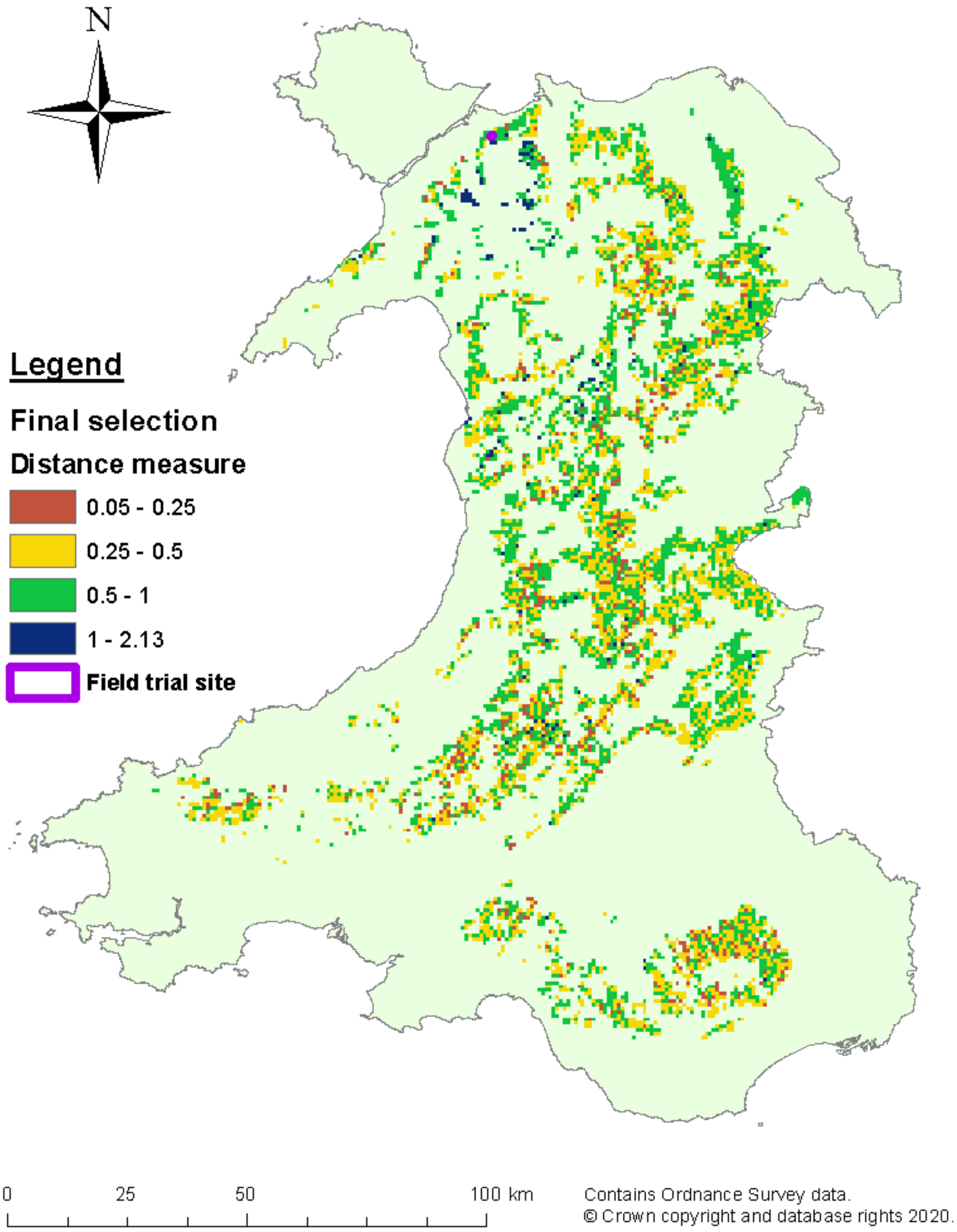

Figure 5.5. Final distance layer of the categorical and continuous selection combined ranging from 0.05 to 2.13 SD.

Table 5.3. Potential grass production and nitrous oxide emissions based on various Euclidean distance measures for the final selection (within 0.25 SD, within 0.5 SD, within 1.0 SD). Values are expressed as an annual total. Total N_2O emissions (N_2O-N converted to N_2O) is expressed as CO_2 equivalent, and used to calculate the proportion of Wales' agricultural GHG production.

Management	Total pasture production (Gg DM)			Total N_2O emissions (Mg N_2O-N)			Proportion of Wales' agricultural GHG production (%)		
	Within 0.25 SD	Within 0.5 SD	Within 1.0 SD	Within 0.25 SD	Within 0.5 SD	Within 1.0 SD	Within 0.25 SD	Within 0.5 SD	Within 1.0 SD
Unimproved	152.31	926.36	1,792.15	9.27	56.39	109.10	0.076	0.462	0.893
Lime and fertiliser input only	154.84	941.70	1,821.83	8.89	54.04	104.55	0.073	0.442	0.856
Rotovate, spring forage rape followed by autumn grass reseed, lime and fertiliser input	-	-	-	61.33	372.98	721.57	0.502	3.053	5.907
Plough, spring grass reseed, lime and fertiliser input	329.41	2,003.43	3,875.88	71.72	436.17	843.83	0.587	3.571	6.908
Rotovate, spring grass reseed, lime and fertiliser input	308.36	1,875.45	3,628.28	100.37	610.46	1,181.01	0.822	4.998	9.669

Table 5.4. Cattle liveweight gain based on the final selection (within 0.25 SD, within 0.5 SD, within 1.0 SD).

Management	Actual total cattle liveweight gain based on experimental results (Gg)			Potential total cattle liveweight gain based on potential pasture production (Gg)		
	Within 0.25 SD	Within 0.5 SD	Within 1.0 SD	Within 0.25 SD	Within 0.5 SD	Within 1.0 SD
Unimproved	31.04	188.76	365.19	-	-	-
Lime and fertiliser input only	26.81	163.08	315.49	31.55	191.89	371.24
Rotovate, spring forage rape followed by autumn grass reseed, lime and fertiliser input	-	-	-	-	-	-
Plough, spring grass reseed, lime and fertiliser input	-	-	-	67.12	408.24	789.79
Rotovate, spring grass reseed, lime and fertiliser input	-	-	-	62.84	382.16	739.34

Upscaling the experimental LCA results based on the cattle liveweight gain values for pasture receiving lime and fertiliser input in Table 5.4 led to a total of 297.63 Gg CO_2-eq (for the land area within 0.25 SD) 1,810.14 kg CO_2-eq (land area within 0.5 SD) and 3,501.94 Gg CO_2-eq (land area within 1.0 SD). For unimproved pasture, the LCA values equate to 254.50 Gg CO_2-eq (land area within 0.25 SD), 1,547.87 Gg CO_2-eq (land area within 0.5 SD) and 2,994.55 Gg CO_2-eq (land within 1.0 SD).

5.3.4 Spatial beef cattle density trends

Data provided by the Cattle Tracing System for the Agricultural Small Area Statistics for Wales (Welsh Government, 2018a) were used along with values of farmed areas from the Small Area Statistics to calculate the beef cow density across Wales on a per farmed hectare basis (Figure 5.6). The beef cow density was at its highest in 2018 in regions of north west Wales. When analysing the trend in beef cow density alongside the categorical selection in Figure 5.6, it is evident that a large proportion of land area selected for upscaling is highlighted as areas where beef cow density is low. Given this, the results suggest that the areas identified in the categorical selection (Figure 5.3) could be considered as key areas for increasing beef production in Wales.

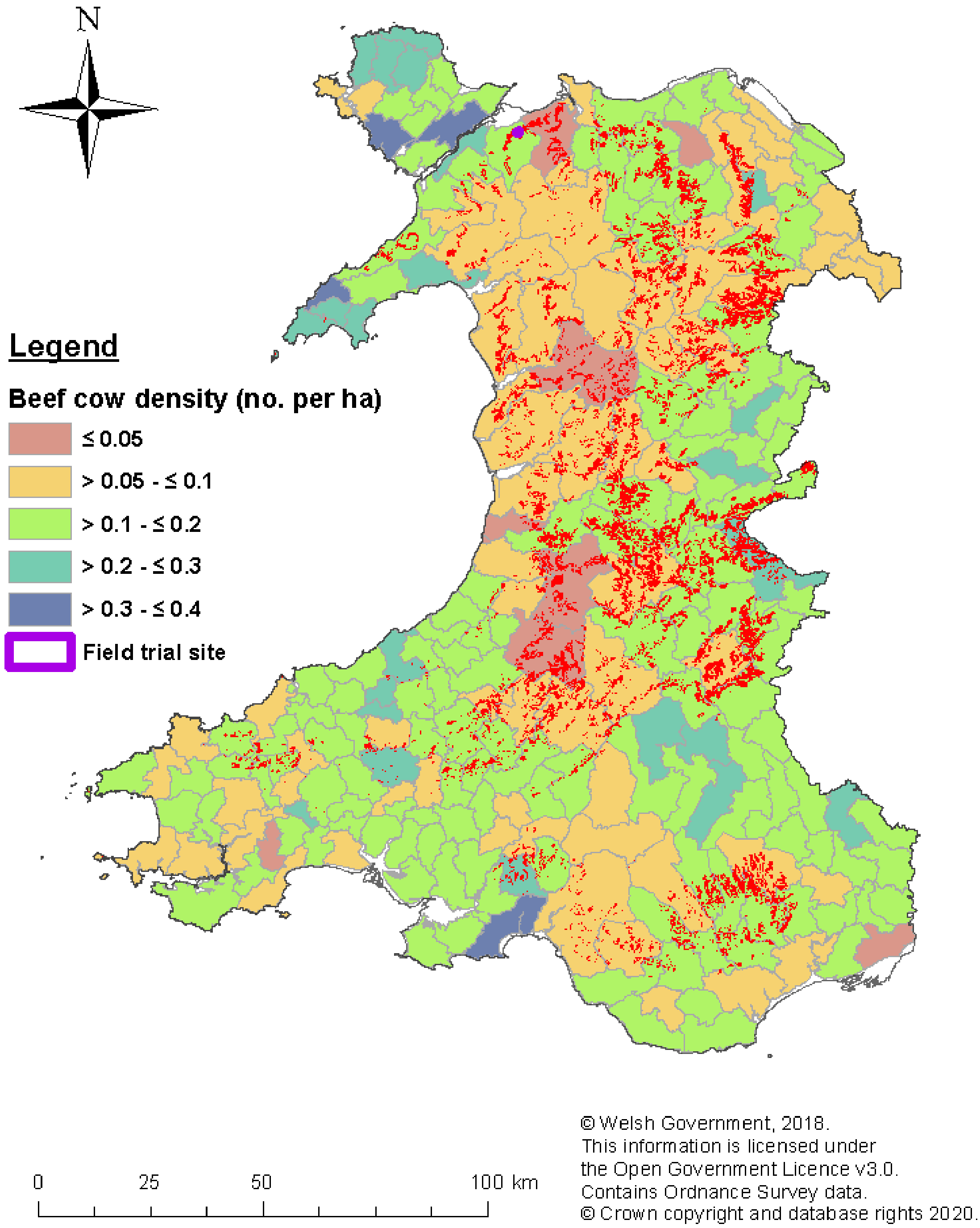

Figure 5.6. Spatial data on beef cow density within farmed areas of Wales (number of cows per farmed ha^{-1}) derived from the Agricultural Small Area Statistics for Wales, based on the Cattle Tracing System data (Welsh Government, 2018a). The red areas represent the categorical selection.

5.3.5 A spatial analysis of the current selection, current and future land for woodland

Figure 5.7 below shows the areas of Wales which are identified in the categorical selection that overlap with land currently managed as woodland, along with areas that have been identified as potential sites to be managed as woodland in the future. In 2018, the total area of forestry land use in Wales was 276,724 ha (Natural Resources Wales, 2018). The overlapping area where land was identified via categorical selection but managed as woodland in 2018 was 607 ha.

A total of 994,234 ha has been identified in the Glastir Woodland Creation Opportunities map as potential sites for new woodland creation in the future. The land is scored from 0-37, depending on the suitability for new woodland creation. When comparing these areas along with the land included in the categorical selection for upscaling, 34,395 ha (3.5% of the land identified in the Glastir Woodland Creation Opportunities map, 54.9% of the categorical selection land) is identified as land that would be suitable to improve for the expansion of livestock production or afforestation in the future. Consequently, this would suggest that there will be increasing competition for the land area that is suitable for both land use types in the future, provided that there is a demand for increased beef production. When assessing the overlapping areas in Figure 5.7, a total of 28,281 ha is surplus from the categorical selection that is not deemed suitable for afforestation. To the contrary, improving the overlapping areas for agricultural purposes, i.e. livestock production, would result in a total of 959,839 ha of land that could be considered for new woodland creation as of the Glastir Woodland Creation Opportunities map scoring.

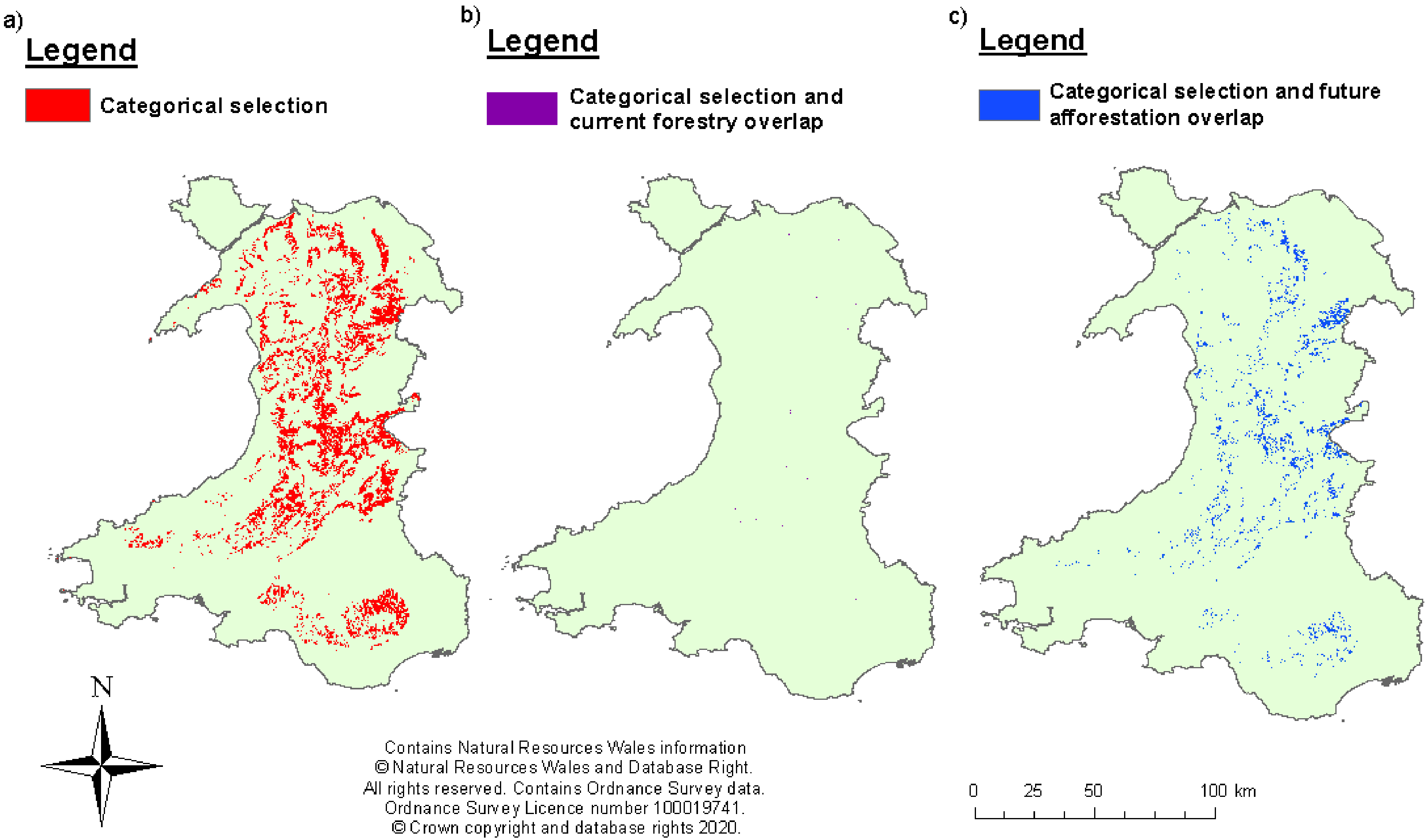

Figure 5.7. Extent of the land areas identified as a) the categorical selection for expansion of pasture and therefore livestock production along with distribution of land b) managed as woodland in 2018, and c) determined as potential sites for afforestation in the future.

5.4 Discussion

In this study, we captured the potential area for improving pasture and expanding upland beef production in the future. Many existing spatial analyses have considered the suitability of land area for afforestation in the future (Committee on Climate Change, 2020b). However, here we integrated both land uses, which are in fact the two dominant land uses in the uplands and may be in increased 'competition' in the future. Due to lack of a statutory definition for the uplands, research does not often focus on land use in these areas. The spatial heterogeneity along with the several levels of matching in the final selection distance layer is valuable when assessing various new woodland creation targets required in order to achieve net zero in the future.

Given that similarities occur in land area identified in the categorical selection distance layer and areas where beef cow density is low, it is important to consider how pasture improvement methods could lead to increased beef production from upland areas. Nevertheless, it should be emphasised that physical limitations may restrict the extent to which improvement methods such as ploughing and rotovating can be implemented in such areas. Whilst altering the management (e.g. reseeding, varying nutrient inputs and grazing regimes) can influence agricultural productivity in these areas, natural limitations (e.g., altitude and slope) are factors that cannot be altered to increase production.

The uplands are currently dominated by extensive livestock systems (National Assembly for Wales Commission, 2016). Based on the upscaling results in the final selection, there is potential to increase beef production from these areas by increasing pasture production. Assuming that demand for Welsh beef remains constant over the coming years (Hybu Cig Cymru, 2020), current beef production levels could be achieved from a reduced land area. Extensive upland livestock systems are currently heavily reliant on subsidies (Arnott et al., 2019), and removal of the Common Agricultural Policy Pillar 1 payments will highlight the vulnerability of such systems in their current state. Changes in agricultural subsidies in the future may lead to the implementation of the 'land sparing' concept, whereby the most productive land on farm would be intensified in terms of agricultural production, which would free up less productive land for other land uses, e.g. new woodland creation. This concept is increasingly discussed as a method of offsetting GHGs produced from agriculture (Cohn et al., 2014; de Oliveira Silva et al., 2016; Herrero et al., 2016; Lamb et al., 2016), e.g. those highlighted in Table 5.3. Sustainable agricultural intensification in some areas by means of reseeding, lime and fertiliser applications, coupled with efficient utilisation of the additional grass grown, as well as good husbandry to maximise cattle growth rates (e.g. via good management of animal health) (Wreford et al., 2015) could lead to a reduction in the carbon intensity per unit of beef produced, as opposed to inefficient production over a larger land mass (Soteriades et al., 2019). However, if it presents an opportunity to increase beef cattle stocking rates and therefore the total number of cattle, this would result in greater absolute GHG emissions produced from cattle systems, which goes against targets for reducing GHG emissions from agriculture. Improving beef production efficiency in this way would, however, require a reduced land area, offering opportunities for where the 'spare' land elsewhere could be managed for afforestation as a means of mitigating net GHG production (Styles et al., 2018). A desirable outcome for land used for cattle grazing systems in the future, however, would be to identify areas where beef production could be maximised but with fewer cattle. This could be achieved by good husbandry measures (improving animal health, optimised diets, and reducing mortality losses) which would enhance liveweight gain (thereby reducing days to slaughter) and/or heavier beef carcasses. Given that cattle performance is a key factor contributing to emission intensity (McAuliffe et al., 2018), maximising production from fewer stock in suitable areas would result in both reduced emission intensity of the product and the absolute GHG emissions, especially where 'spared' land can be managed to offset emissions.

Our results suggest that there will be increased competition for delivery of the two dominant upland land use in Wales in the future – agriculture in the form of livestock production and forestry. Fifteen percent of Wales' total land area is managed as woodland (Forestry Commission, 2016), which is significantly lower than many other European countries. As previously mentioned, it is advised that 152,000 ha of new woodland by planting conifer and broadleaf is required by 2050 to work towards Wales achieving net zero emissions (Committee on Climate Change, 2020b). When examining this along with the areas identified in our results, ca. 250,000 ha of the final selection distance layer could therefore be managed for livestock production.

As seen in Figure 5.3, the areas categorically selected as similar locations to the field trial sites are dispersed across Wales. The majority of these land areas are located in the eastern regions of the country. Much of the land identified in the east is also scored as appropriate woodland creation sites in the Glastir Woodland Creation Opportunities Map (see Appendix 5.3, Figure E). However, land identified in mid-Wales via categorical selection does not correspond to suitable areas for new woodland creation through the Glastir scheme. It is anticipated that land suitability for afforestation will shift geographically in the future (Environmental Systems, 2020). While a relatively large area of Wales has been identified for woodland creation, this land may not be suitable for all tree species. For example, a study working on the suitability of land to deliver the afforestation target for Wales identified that areas in east Wales will be largely unsuitable for sessile oak *(Quercus petraea)* and sitka spruce *(Picea sitchensis)* planting by 2050 under both medium and high emission scenarios (Environmental Systems, 2020). Many of these areas are currently managed as extensive livestock systems. Nevertheless, this does not indicate that these areas are fully suitable for the expansion and intensification of beef production either, as they may also be precluded from measures to improve agricultural productivity (such as nutrient inputs) due to e.g. environmental designations. Maintaining and restoring key habitats is a means of ecosystem services provision in the uplands. Previous studies have indicated an imbalance in ecosystem services and dis-services delivery from both agriculture and forestry upland land use in Wales (Hardaker et al., 2020). Key habitat and species priorities may influence land use management decisions in the future, e.g. increased grazing intensity in the long-term led to a decline in breeding density and egg-stage nest survival of meadow pipit in British uplands. However, a decline in fledglings, potentially as a result of increased predator numbers was harboured by nearby developing woodland (Malm et al., 2020). Careful consideration should be given to selecting areas for land use change, i.e. afforestation in order to avert from using land of high biodiversity value currently managed for high nature value farming (McCracken, 2004). The potential positive and negative interaction between grazing, afforestation, and biodiversity targets adds further nuance and complexity to the discussion as to what the uplands should be managed for.

Given that a significant proportion of land is privately rather than state owned, further research is required on ownership of the land identified in Figure 5.7 for potential afforestation. Private landowners are not sufficiently up-taking available incentives to achieve afforestation targets (Lawrence and Dandy, 2014), which could be a considerable challenge in achieving net zero targets in the future. Changes in land values and poor financial returns have been identified as barriers for afforestation (Eves et al., 2014). Previous research has determined that subsidies have a major impact on afforestation decisions (Eves et al., 2014; Glynn et al., 2012). Hardaker (2018) concluded that establishing woodland on upland farmland is not economically viable without government subsidies and grant incomes. It could therefore be argued that some form of financial support will be required to maintain both agricultural and forestry land use in Wales' uplands in the future, based on current projections. Recent consultations on Wales' future agricultural policy has largely emphasised the intended focus on sustainable food production and delivery and maintenance of fundamental environmental services (Welsh Government, 2019d). Given that payments for environmental outcomes is highlighted as an important aspect of financial support to Welsh farmers after leaving the European Union (Welsh Government, 2019d), the ability for farm business to derive an income from providing environmental outcomes will likely impact upland land use in the future.

We are faced with the challenge of maintaining upland beef production as well as increasing new woodland creation in order to achieve afforestation and net zero targets in the future. Some of the methods discussed and implemented in previous chapters, e.g. implementing pasture improvement methods in some areas to increase beef production in order to release land elsewhere for afforestation, provides one potential solution to this challenge. Upland areas that are currently regarded as limited or unsuitable for new woodland creation may be required for afforestation by 2080 due to increases more broadly in soil droughtiness (Environmental Systems, 2020). This brings uncertainty as to how this will impact on land area required for forestry in the future. Conversion of land use from agricultural production to afforestation may lead to a reduction in livestock output from these areas and require intensifying agricultural production elsewhere (Hardaker, 2018). These locations currently unsuitable for afforestation could therefore be considered for potential sustainable intensification (maintenance of overall yield by means of productivity gains in suitable areas). In doing so, land elsewhere could be converted for afforestation and delivery of other ecosystem services.

In the event of the spatial suitability of land area for afforestation expanding in the future, new woodland creation should not be at the expense of food production (Committee on Climate Change, 2020a). The dangers of "off-shoring" environmental impacts is increasingly recognised, and a reduction in self-sufficiency and increased beef imports should be avoided. For example, high food imports in Sweden reduces the country's land area requirement for food production by displacing production to high-yielding areas (Kastner et al., 2014). However, this results in 60% of the GHGs responsible for the country's food consumption total climate footprint produced overseas (Cederberg et al., 2019). This is not always an effective approach in "off-setting" GHGs, as areas of low environmental impacts elsewhere are often influenced i.e. land overseas converted for agricultural production, leading to increasing deforestation (Cuypers et al., 2013; Henders et al., 2015). The Committee on Climate Change have identified a 20% reduction in meat and dairy consumption would contribute to reducing GHGs derived from food, and would 'free up' land currently used for agriculture to be used for measures to off-set GHG emissions, predominantly via afforestation (Committee on Climate Change, 2020a). However, at present, there are few precedents for intervention to encourage this. Cattle numbers are already decreasing due to market forces and changes in agricultural support. A further reduction may leave the sector unviable due to reaching a pivotal point from an economic perspective, such as a critical mass needed to sustain other services in the sector (e.g. abattoirs). This emphasises the need to drive efficiency improvements in the uplands and this should be incorporated into future agricultural policy.

5.5 Conclusions

This study has shown that a substantial proportion of Wales' uplands identified for future potential to increase beef production overlaps with areas where afforestation has been proposed. Achieving a balance between these two land uses is fundamental in order to meet the demand for beef production and to achieve new woodland creation targets. Sustainable intensification (i.e. beef production in this chapter) of some upland areas, thereby releasing land for afforestation could be one method of offsetting GHGs from agriculture. The proposed afforestation targets for Wales under current food consumption patterns cannot be achieved without compromising beef production, unless there are significant beef productivity gains. Future policy should encourage the implementation of this on farm, which would also allow additional land for new woodland creation.

5.6 References

AHDB, 2017. Nutrient Management Guide (RB209). [Online]. Available at: https://ahdb.org.uk/nutrient-management-guide-rb209. Accessed 10th June, 2020.

Arnott, D., Chadwick, D.R., Wynne-Jones, S., Jones, D.L., 2019. Vulnerability of British Farms to post-Brexit subsidy removal, and implications for intensification, extensification and land sparing. *Land Use Policy*, 104154.

Cederberg, C., Persson, U.M., Schmidt, S., Hedenus, F., Wood, R., 2019. Beyond the borders–burdens of Swedish food consumption due to agrochemicals, greenhouse gases and land-use change. *Journal of Cleaner Production 214*, 644-652.

CEH, 2017. Land Cover Map 2015. [Online]. Available at: https://www.ceh.ac.uk/services/land-cover-map-2015. Accessed June 30th, 2020.

Cohn, A. S., Mosnier, A., Havlık, P., Valin, H., Herrero, M., Schmid, E., Obersteiner, M., 2014. Cattle ranching intensification in Brazil can reduce global greenhouse gas emissions by sparing land from deforestation. *Proceedings of the National Academy of Sciences 111*(20), 7236–7241.

Committee on Climate Change, 2020a. Land use: Policies for a Net Zero UK. [Online]. Available at: https://www.theccc.org.uk/wp-content/uploads/2020/01/Land-use-Policies-for-a-Net-Zero-UK.pdf. Accessed May 10th, 2020.

Committee on Climate Change, 2020b. Tree Suitability Modelling – Planting Opportunities for Sessile Oak and Sitka Spruce in Wales in a Changing Climate. [Online]. Available at: https://www.theccc.org.uk/wp-content/uploads/2020/01/Environment-Systems-Ltd-2020-Tree-Suitability-Modelling-%E2%80%93-Planting-Opportunities-for-Sessile-Oak-and-Sitka-Spruce-in-Wales-in-a-Changing-Climate.pdf. Accessed May 12th, 2020.

Committee on Climate Change, 2019. Net Zero – The UK's contribution to stopping global warming. [Online]. Available at: https://www.theccc.org.uk/wp-content/uploads/2019/05/Net-Zero-The-UKs-contribution-to-stopping-global-warming.pdf. Accessed May 10th, 2020.

Committee on Climate Change, 2018. Reducing UK emissions – 2018 Progress Report to Parliament. Technical Annex: The Smart Agriculture Inventory. [Online]. Available at: https://www.theccc.org.uk/wp-content/uploads/2018/08/PR18-Chapter-6-Annex-The-Smart-Agriculture-Inventory.pdf. Accessed June 30th, 2020.

Conant, R.T., Paustian, K., Elliott, E.T., 2001. Grassland management and conversion into grassland: effects on soil carbon. *Ecological Applications 11*, 343-355.

Cuypers, D., Geerken, T., Gorissen, L., Lust, A., Peters, G., Karstensen, J., Prieler, S., Fischer, G., Hizsnyik, E., Van Velthuizen, H., 2013. The impact of EU consumption on deforestation: Comprehensive analysis of the impact of EU consumption on deforestation.

de Oliveira Silva, R., Barioni, L. G., Hall, J. A. J., Folegatti Matsuura, M., Zanett Albertini, T., Fernandes, F. A., Moran, D., 2016. Increasing beef production could lower greenhouse gas emissions in Brazil if decoupled from deforestation. *Nature Climate Change 6*(5), 493–497.

Environmental Systems, 2020. Tree Suitability Modelling – Planting Opportunities for Sessile Oak and Sitka Spruce in Wales in a Changing Climate. [Online]. Available at: https://www.theccc.org.uk/wp-content/uploads/2020/01/Environment-Systems-Ltd-2020-Tree-Suitability-Modelling-%E2%80%93-Planting-Opportunities-for-Sessile-Oak-and-Sitka-Spruce-in-Wales-in-a-Changing-Climate.pdf. Accessed May 19th, 2020.

ESRI, 2020. ArcGIS Desktop: version 10.7.1. Redlands, California, USA: Environmental Systems Research Institute.

Eves, C., Johnson, M., Smith, S., Quick, T., Langley, E., Jenner, M., Richardson, W., Glynn, M., Anable, J., Crabtree, B., White, C., Black, J., MacDonald, C., Slee, B., 2014. Analysis of the Potential Effects of Various Infleunces and Interventions on Woodland Management and Creation Decisions, Using a Segmentation Model to Categorise Sub-Groups. Defra, London.

FAO, 2019. Key facts and findings. By the numbers: GHG emissions by livestock. [Online]. Available at: http://www.fao.org/news/story/en/item/197623/icode/. Accessed June 24th, 2020.

Forestry Commission, 2020. Provisional Woodland Statistics. 2020 Edition. [Online]. Available at: https://www.forestresearch.gov.uk/documents/7647/PWS_2020.pdf. Accessed June 24th, 2020.

Glynn, M., Richardson, W., Anable, J., Quick, T., Rowcroft, P., Smith, S., 2012. Independent Panel on Forestry Woodland Owner Survey. Final Report to the Independent Panel on Forestry. URS, London.

Goulding, K.W.T., Stockdale, E.A., Fortune, S., Watson, C., 2000. Nutrient cycling on organic forms. *Journal of the Royal Agricultural Society of England 161*, 65-75.

Hallett, S.H., Sakrabani, R., Keay, C.A., Hannam, J.A., 2017. Developments in land information systems: examples demonstrating land resource management capabilities and options. *Soil Use and Management 33*(4), 514-529.

Hardaker, A., Pagella, T., Rayment, M., 2020. Integrated assessment, valuation and mapping of ecosystem services and dis-services from upland land use in Wales. *Ecosystem Services 43*, 101098.

Hardaker, A., 2018. Is forestry really more profitable than upland farming? A historic and present day farm level economic comparison of upland sheep farming and forestry in the UK. *Land Use Policy 71*, 98-120.

Henders, S., Persson, U.M., Kastner, T., 2015. Trading forests: land-use change and carbon emissions embodied in production and exports of forest-risk commodities. *Environmental Research Letters 10*(12), 125012.

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

Hybu Cig Cymru, 2020. Industry Statistics. [Online]. https://meatpromotion.wales/en/industry-statistics. Accessed June 16th, 2020.

Illbery, B.W., Kidd, J., 1992. Adoption of the Farm Woodland Scheme in England. *Geography 77*(4), 363-367.

Kastner, T., Erb, K.H., Haberl, H., 2014. Rapid growth in agricultural trade: effects on global area efficiency and the role of management. *Environmental Research Letters 9*(3), 034015.

Lamb, A., Green, R., Bateman, I., Broadmeadow, M., Bruce, T., Burney, J., Carey, P., Chadwick, D., Crane, E., Field, R., Goulding, K., 2016. The potential for land sparing to offset greenhouse gas emissions from agriculture. *Nature Climate Change*, *6*(5), 488-492.

Lawrence, A., Dandy, N., 2014. Private landowners' approaches to planting and managing forests in the UK: what's the evidence? *Land Use Policy 36*, 351-360.

Malm, L.E., Pearce-Higgings, J.W., Littlewood, N.A., Karley, A.J., Karaszewska, E., Jaques, R., Pakeman, R.J., Redpath, S.M., Evans, D.M., 2020. Livestock grazing impacts components of the breeding productivity of a common upland insectivorous passerine: Results from a long-term experiment. *Journal of Applied Ecology,* 1-10.

Mansfield, L., 2011. Upland Agriculture and the Environment. Badger Press Ltd.

McCracken, D.I., 2004. Farmland and high nature value areas. *Background report prepared for ECNC/ETC/NPB input to European Environment Agency sub-report on Halting Biodiversity Loss. European Environment Agency, Copenhagen*.

National Assembly for Wales Commission, 2016. The Farming Sector in Wales. [Online]. Available at: https://senedd.wales/research%20documents/16-053-farming-sector-in-wales/16-053-web-english2.pdf. Accessed June 30th, 2020.

National Soils Resources Institute - Cranfield University, 2006. National Soil Map (NATMAP) Soilscapes. [Online]. Available at: http://www.landis.org.uk/soilscapes/about.cfm. Accessed June 30th, 2020.

National Soils Resources Institute - Cranfield University, 2002. Spatial Environmental Information System for Modelling the Impact of Chemicals (SEISMIC) Agroclimatic data. [Online]. Available at: http://www.landis.org.uk/services/seismic.cfm. Accessed June 30th, 2020.

Natural Resources Wales, 2018. National Inventory of Woodland and Trees. [Online]. Available at: https://libcat.naturalresources.wales/webview/?infile=details.gluandloid=116285andrs=202937andhitno=1andstraight_to_details=andtiarray=full. Accessed June 30th, 2020.

Ordnance Survey, 2009. Land-Form PROFILE DTM. [Online]. Available at: https://digimap.edina.ac.uk/webhelp/os/data_information/os_products/land-form_profile_dtm.htm. Accessed June 30th, 2020.

Oosthoek, J.W., 2000. The Logic of British Forestry Policy, 1919-1970. In: *Proceedings of the 3rd Biennal Conference of the European Society for Ecological Economics,* 2.

Pringle, D., 1994. The First 75 years: a brief account of the history of the Forestry Commission, 1919-1994. Forestry Commission: Edinburgh.

Reed, M.S., Bonn, A., Slee, W., Beharry-Borg, N., Birch, J., Brown, I., Burt, T.P., Chapman, D., Chapman, P.J., Clay, G.D., Cornell, S.J., 2009. The future of the uplands. *Land Use Policy 26*, S204-S216.

Soteriades, A.D., Foskolos, A., Styles, D., Gibbons, J.M., 2019. Diversification not specialization reduces global and local environmental burdens from livestock production. *Environment International 132*, 104837.

Soussana, J.F., Tallec, T., Blanfort, V., 2010. Mitigating the greenhouse gas balance of ruminant production systems through carbon sequestration in grasslands. *Animal 4*(3), 334-350.

Stapledon, R.G., 1951. Reclaiming Land for Agriculture. Crosby Lockwood and Son: London.

Styles, D., Gonzalez-Mejia, A., Moorby, J., Foskolos, A., Gibbons, J., 2018. Climate mitigation by dairy intensification depends on intensive use of spared grassland. *Global Change Biology 24*(2), 681-693.

Welsh Government, 2020a. Agricultural land classification: predictive map. [Online]. Available at: https://gov.wales/agricultural-land-classification-predictive-map. Accessed May 19th, 2020.

Welsh Government, 2020b. Draft Water Resources (Control of Agricultural Pollution) (Wales) Regulations 2020. [Online]. Available at: https://gov.wales/sites/default/files/publications/2020-04/water-resources-control-agricultural-pollution-wales-regulations-2020-draft-statutory-instrument_0.pdf. Accessed May 12th, 2020.

Welsh Government, 2020c. Glastir Woodland Creation – Opportunities Map. [Online]. Available at: http://lle.gov.wales/apps/woodlandopportunities/#lat=52.4342andlon=-3.8177andzoom=8. Accessed May 12th, 2020.

Welsh Government, 2019a. Agriculture in Wales. [Online]. Available at: https://gov.wales/sites/default/files/publications/2019-06/agriculture-in-wales-evidence.pdf. Accessed May 18th, 2020.

Welsh Government, 2019b. Predictive Agricultural Land Classification (ALC) Map 2. [Online]. Available at: http://lle.gov.wales/map/alc2. Accessed June 30th, 2020.

Welsh Government, 2019c. Prosperity for All: A Low Carbon Wales. [Online]. Available at: https://gov.wales/sites/default/files/publications/2019-06/low-carbon-delivery-plan_1.pdf. Accessed May 12th, 2020.

Welsh Government, 2019d. Sustainable Farming and Our Land. [Online]. Available at: https://gov.wales/sites/default/files/consultations/2019-07/brexit-consultation-document.pdf. Accessed June 17th, 2020.

Welsh Government, 2018a. Agricultural Small Area Statistics: 2002 to 2018. [Online]. Available at: https://gov.wales/agricultural-small-area-statistics-2002-2018. Accessed June 30th, 2020.

Welsh Government, 2018b. Glastir Woodland Creation Rules Booklet. Version 6 – February 2018. [Online]. Available at: https://gov.wales/sites/default/files/publications/2018-08/glastir-woodland-creation-window-6-february-2018-rules-booklet.pdf. Accessed June 16th, 2020.

Welsh Government, 2018c. Woodlands for Wales. The Welsh Government's Strategy for Woodlands and Trees. [Online]. Available at: https://gov.wales/sites/default/files/publications/2018-06/woodlands-for-wales-strategy_0.pdf. Accessed June 16th, 2020.

Wreford, A., Moran, D., Moxey, A., Andy Evans, K., Fox, N., Glenk, K., Hutchings, M., McCracken, D.I., McVittie, A., Mitchell, M., Topp, C.F., 2015. Estimating the costs and benefits of adapting agriculture to climate change. *EuroChoices 14*(2), 16-23.

6. Discussion

In this chapter, the key findings will be summarised according to the original research aims. Furthermore, study limitations will be identified along with suggested areas for further research.

6.1 Study limitations

While I am confident that the field sites designated for the research encompassed in the data chapters of this thesis are true reflections of typical agricultural upland sites in the UK, and specifically in Wales, I recognise that having a number of field sites situated across Wales' uplands could have captured differences between sites when conducting data collection for Chapters 3 and 4. These variations between sites could include altitude, slope, climate, soil, vegetation, nutrient input and management factors (Mansfield, 2011). However, due to the restricted time of a PhD, identifying and implementing field experiments over three years across multiple sites was beyond the remit of the present study. Due to extreme environmental conditions (i.e. a drought in 2018), the climate for the sampling years differed dramatically. This impacted the results and meant that despite having multi-year experiments, directly comparing the data from different sampling years was difficult. However, I believe that a large proportion of farmers/land managers will have experienced the same difficulties in terms of pasture and livestock production. Furthermore, results from an abnormal year may still be compared with other studies conducted during the same period.

Some results were omitted from the analysis, e.g. yield-scaled N_2O production was not calculated for 2018 in Chapter 3 due to the low soil N_2O emissions results. There are several controlling factors that influence soil processes i.e. N_2O emissions, including temperature, compaction, water-filled pore space, pH and N availability (Dobbie and Smith, 2001; Torbert and Wood, 1992). Additional sampling to measure these parameters, for instance by taking soil samples and bulk density cores from directly underneath each greenhouse gas chamber on every sampling day would have provided additional information about the effect of environmental conditions e.g. pH, electrical conductivity, soil moisture on soil N_2O emissions. However, excessive soil disturbance due to regular soil sampling within the same area would be unavoidable, and could also influence N_2O emissions. Furthermore, I restricted the sampling and analysis to ensure both field experiments (Chapter 3 and 4) could be managed concurrently within the PhD timeframe.

I made every effort to ensure that the grouping of the cattle (Chapter 4) matched for the different grazing treatments. Sourcing a herd of cattle that was consistent in terms of sex, age, breed and weight proved difficult. Nevertheless, I aimed to mitigate any differences in breed and weight by pairing the heifers together to ensure one grazing treatment was a reflection of the other. Traditionally, the majority of beef production systems in the uplands are managed as suckler systems. Due to the difficulty in monitoring suckler herd performance, I decided on monitoring beef liveweight gain of youngstock for this study. The traditional extensive nature of upland livestock systems means that many sites are managed as a series of open range grazing fields. This was applicable to the field sites used for this research and therefore a substantial amount of equipment was required i.e. fencing materials and water supply to set-up the treatments. For this reason, the stock were treated as replicates rather than having several plot replicates. Furthermore, accessibility to the field sites was difficult, and stocking rates were determined accordingly to ensure adequate pasture to satisfy stock requirements for the duration of the experimental period.

6.2 Key findings

As stated in Chapter 1, the research contained within this thesis aimed to provide answers on the effect of alternative pasture and grazing management options in upland cattle systems on production efficiencies, and their respective environmental trade-offs.

In Chapter 2, I describe the extensive research base that exists encompassing cattle upland systems in the UK, as well as considering Wales as a case study. It is evident that cattle numbers in these areas have fluctuated dramatically over the years due to the influence of factors such as changes in market demand and agricultural policy. However, agriculture and livestock production in particular has remained a dominant upland land use. This chapter described the influencing factors, along with the impact of various management options to consider in terms of upland pasture productivity; from grazing management, reseeding, fertiliser use, breed effect, and the incorporation of supplementary feeding into the system. Reviewing the current literature has improved my understanding of the environmental impacts associated with livestock grazing considerably. However, it became clear that most of the empirical research on cattle upland systems was conducted some time ago. This enabled me to identify gaps in the research whereby I developed the experimental work. These gaps included research on establishment, production and utilisation of particular seed mixtures on-farm as well as the implementation of upland cattle systems in terms of pasture production, grazing management and the environmental impact of such systems.

Here, I present the main findings and implications of the research in relation to the objectives stated in Chapter 1.

To investigate pasture productivity and quality in the uplands under different management options, along with the associated environmental impact.

While it is widely known that reseeding with a suitable grass variety, along with appropriate lime and fertiliser application can lead to greater grass production in grazing systems, few studies have focussed on the uplands. Furthermore, comparing the effect of different cultivation methods prior to sowing on pasture production as well as GHG emitted from the soil has been under-researched in the uplands. In Chapter 3, I demonstrated that cultivation methods i.e. ploughing and rotovating followed by reseeding with a suitable grass variety led to a significant increase in grass growth in comparison with the control and existing upland pasture that received lime and fertiliser application. While cultivation, reseeding and nutrient application led to greater grass growth, these land improvement methods also led to increased nitrous oxide (N_2O) emission from the soil in the year of establishment, with the highest N_2O fluxes and nitrogen (N) fertiliser emission factor (EF) following rotovation. Ploughing and rotovating did not result in greater carbon loss via CO_2 emissions. This finding somewhat contradicts with previous work which detected higher CO_2 emissions following ploughing (Willems et al., 2011), and emphasises the influence of other factors e.g. climate and different sward types on GHG loss from the soil. Altering fertiliser application in consecutive years led to pasture yield gains. However, in contrast to the establishment year, N_2O fluxes and EFs were low. As a major drought was experienced, our results strongly suggest that the low N_2O emissions from the soil was because of little plant N uptake due to lack of soil moisture. Understanding how environmental conditions influence pasture production and GHG emissions is very important when considering the cost-benefits of implementing land improvement measures.

To assess cattle liveweight gain from upland pasture, and to investigate the environmental impact of cattle grazing systems.

Livestock production is responsible for a considerable percentage of global anthropogenic GHG emissions (Gerber et al., 2013), with fertiliser use and livestock excretion resulting in emissions of N_2O. In Chapter 4, I developed the concept of improving upland pasture that was investigated in Chapter 3 by implementing this within a cattle grazing system. Recognising the environmental impact of measures to increase pasture production within livestock grazing systems is important when considering ways of mitigating GHG production from agriculture. Cattle liveweight gain was monitored from stock grazing improved (lime and fertiliser input) and unimproved pasture. Improving grass utilisation has been recognised as a mechanism of maximising both pasture and stock production from grassland (AHDB, 2016), and therefore varying stocking rates between the improved and unimproved pasture was explored. There was no evidence of pasture inputs influencing cattle daily liveweight gain. This implies that the improved pasture was not utilised to its full potential. Furthermore, soil N_2O emissions following lime and fertiliser application were monitored. Urine and dung samples were collected from cattle grazing both the improved and unimproved treatment and applied to the grazed pasture to determine soil N_2O emissions following excretion. I found that soil N_2O emissions were significantly greater following cattle urine excretion than the other treatments (control, lime and fertiliser, and dung deposition). This was particularly evident following urine deposition of the cattle grazing the unimproved pasture, and therefore suggesting that intake of fertilised pasture does not influence N_2O produced from cattle excretion.

The results were used to calculate a combined excretal N_2O EF along with an environmental footprint of both grazing systems per kg of liveweight, using a Life Cycle Assessment tool. The combined excretal N_2O EF was lower than both the IPCC default and the country-specific grazing excretal N_2O EF. Differences between IPCC default, country specific and this study’s combined excretal N_2O EF strengthens the argument for measuring EFs and more generally, the environmental impact at a system scale in order to consider feedback mechanisms linking factors e.g. soil, pasture and livestock (McAuliffe et al., 2020).

With increasing pressure on the agricultural sector to reduce its environmental burden, it is encouraging that the combined excretal N_2O EF from these systems is lower than both IPCC and country-specific default as this implies that upland cattle systems may not have a considerable impact on N_2O production via excretion. However, urine and dung excretion onto pasture is only one route of N_2O emissions from agriculture. Other sources include fertiliser N leaching, use of manure as fertiliser on-farm and the ploughing and cultivation of crop residues and legumes (Welsh Government, 2008). Furthermore, the environmental burden in the form of GHGs deriving both directly and indirectly from agricultural practices includes the production of carbon dioxide (CO_2) and methane (CH_4). While the research included in this chapter focussed on N_2O emissions, livestock enteric sources largely contribute to Wales’ total CH_4 emissions (Welsh Government, 2008).

To examine upland land use in the future and competition for land between agriculture and afforestation.

Delivering the dual ambitions of achieving afforestation targets and maintaining other fundamental land uses is a challenge for several countries. Afforestation has been identified as a process to drive emissions sequestration (Committee on Climate Change, 2020). Wales is among some of the countries working towards increasing new woodland creation, whilst not compromising its agricultural land. In

Chapter 5, Wales was used as a case study to identify upland areas that could be considered for sustainably intensifying beef production in the future. The results from Chapter 3 and 4 were extrapolated to model potential beef production from the identified areas. Furthermore, areas that will possibly be considered for afforestation were determined. I observed that a reasonable proportion of Wales' uplands could both be used for beef production or afforestation in the future. This implies that there will be increased competition between upland agriculture and forestry in the future, and this will largely depend on the availability and form of financial support in the form of subsidies (Hardaker, 2018). Due to the restricted time of a PhD preventing extrapolating of the research included in Chapters 3 and 4 to multiple field sites, an economic analysis based on data from a single field site was not deemed appropriate, due to variations between sites. I considered ways of achieving a balance between both land uses, including sustainable beef production from a smaller land area, releasing some land for afforestation, which would also offset GHG emissions. Currently, Welsh tree planting figures do not suffice for working towards achieving the proposed afforestation and GHG mitigation targets. This data provides clear scenarios that represent current beef production and afforestation projections that could be useful in informing policy and decision-makers.

6.3 Future developments

Given the limitations highlighted above, the following research would complement the results in this thesis:

- Further research on upland pasture productivity and the associated environmental impact. A series of field trials in locations across Wales' uplands to monitor pasture production, quality and GHG emissions following cultivation, reseeding, lime and fertiliser application.
- Further research into cattle liveweight gain from upland grazing systems. The experiment in Chapter 4 could be developed across multiple fields on farms. This would establish field replication as well as cattle replication. Furthermore, the use of additional fields would increase the cattle sample size, therefore statistical power. Another development to consider would be to repeat the experiment in Chapter 4 but with a herd of younger heifers in order to assess the effect of differences in growth rates stages on liveweight gain. This experiment could be established as a long-term study, with potential to vary cattle age and breed over different years.
- Different pasture types has shown to affect enteric CH_4 emissions from lambs, with higher emissions from lambs fed on ryegrass than permanent pasture (Fraser et al., 2015). Measuring enteric CH_4 emissions produced from cattle of varying breeds and pasture type would be a

valuable development of Chapter 4 in order to consider other (unmeasured in this thesis) environmental burdens from cattle. This could be further developed by calculating the carbon footprint of the product, which would then reflect the actual EF for N_2O and the amount of CH_4 generated when livestock are on different diets/pasture systems.

- In the event of some of the above developments being undertaken, an economic analysis of the costs involved with increasing the efficiency of upland cattle systems across several sites could be carried out.

6.4 Wider implications of the research

The findings of the studies encompassed within this thesis have indicated that there are opportunities to apply the concept of sustainable intensification to upland cattle systems in order to increase production efficiencies, and thereby reduce the environmental burden associated with livestock production. This will include implementing measures not studied here, such as improving animal health, genetic merit, and diets; whilst this work has shown the potential gains from improved pasture productivity and utilisation, and targeted nutrient application. Collectively, such measures could allow for intensifying beef production from less land area, which should reduce both total GHG emissions and the carbon intensity of beef produced. While this is not achievable from all of Wales' uplands, a significant proportion of land will be required for new woodland creation in the future to achieve afforestation targets, as previously discussed. Current agricultural policy in Wales prevails around agricultural production as well as environmental land management. Future upland policy will be largely driven by targets such as those to increase afforestation and mitigate climate change. Results of this study clearly highlights possible mechanisms to work towards this, and whilst extensive upland cattle systems will need to adapt accordingly, food production should not be compromised and the environmental impacts of food 'off-shored', given that the demand remains constant. This may mean that future upland policy could drive less efficient upland cattle systems from a production and environmental perspective away from livestock production, with the land converted for an alternative use. The purpose of this research is to contribute to the growing evidence base around the potential production and environmental impact of grazing cattle, but to also generate upland-specific data that could be of use to policy-makers in the future.

6.5 References

AHDB, 2016. Improve grass utilisation. [Online]. Available at: http://beefandlamb.ahdb.org.uk/improve-grass-utilisation/. Accessed June 30th, 2020.

Committee on Climate Change, 2020. Land use: Policies for a Net Zero UK. [Online]. Available at: https://www.theccc.org.uk/wp-content/uploads/2020/01/Land-use-Policies-for-a-Net-Zero-UK.pdf. Accessed June 10th, 2020.

Dobbie, K.E., Smith, K.A., 2001. The effects of temperature, water-filled pore space and land use on N_2O emissions from an imperfectly drained gleysol. *European Journal of Soil Science 52*(4), 667-673.

Fraser, M.D., Fleming, H.R., Theobald, V.J., Moorby, J.M., 2015. Effect of breed and pasture type on methane emissions from weaned lambs offered fresh forage. *The Journal of Agricultural Science 153*(6), 1128-1134.

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

Hardaker, A., 2018. Is forestry really more profitable than upland farming? A historic and present day farm level economic comparison of upland sheep farming and forestry in the UK. *Land Use Policy 71*, 98-120.

Mansfield, L., 2011. Upland Agriculture and the Environment. Badger Press, Bowness on Windermere.

McAuliffe, G.A., López-Aizpún, M., Blackwell, M.S.A., Castellano-Hinojosa, A., Darch, T., Evans, J., Horrocks, C., Le Cocq, K., Takahashi, T., Harris, P., Lee, M.R.F., 2020. Elucidating three-way interactions between soil, pasture and animals that regulate nitrous oxide emissions from temperate grazing systems. *Agriculture, Ecosystems and Environment 300*, 106978.

Torbert, H.A., Wood, C.W., 1992. Effects of soil compaction and water-filled pore space on soil microbial activity and N losses. *Communications in Soil Science and Plant Analysis 23*(11-12), 1321-1331.

Welsh Government, 2008. Climate Change and Agriculture: Options for Mitigation of Greenhouse Gas Emissions from Agricultural Activity in Wales. [Online]. Available at: https://gov.wales/sites/default/files/publications/2018-01/options-to-mitigate-greenhouse-gas-emissions-from-agricultural-activity.pdf. Accessed June 30th, 2020.

Willems, A.B., Augustenborg, C.A., Hepp, S., Lanigan, G., Hochstrasser, T., Kammann, C., Müller, C., 2011. Carbon dioxide emissions from spring ploughing of grassland in Ireland. *Agriculture, Ecosystems and Environment 144*(1), 347-351.

Appendices

Appendix 3.3 – Supplementary material to Chapter 3 Results

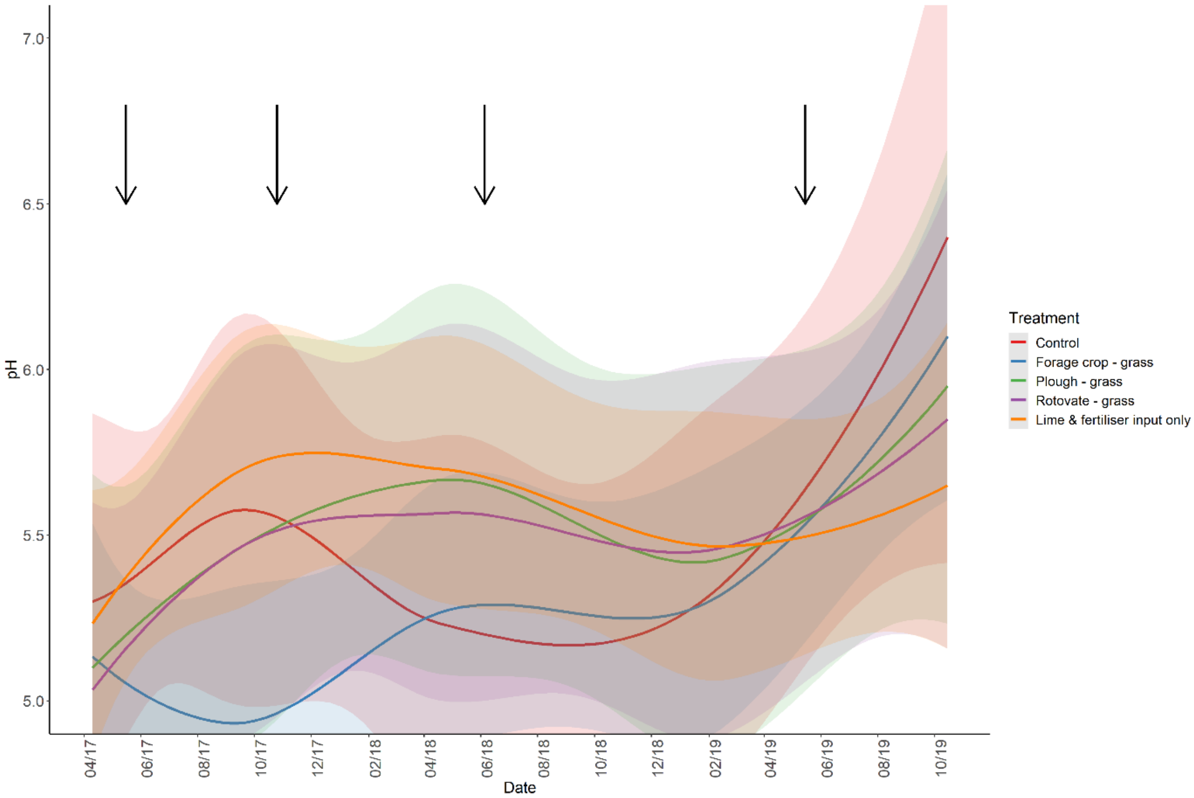

Figure A. Mean soil pH of the treatments for the sampling period (2017–2019). Coloured lines represent the treatment mean, shaded areas represent the upper and lower bounds of the SEM. Arrows represent the timing of lime applications.

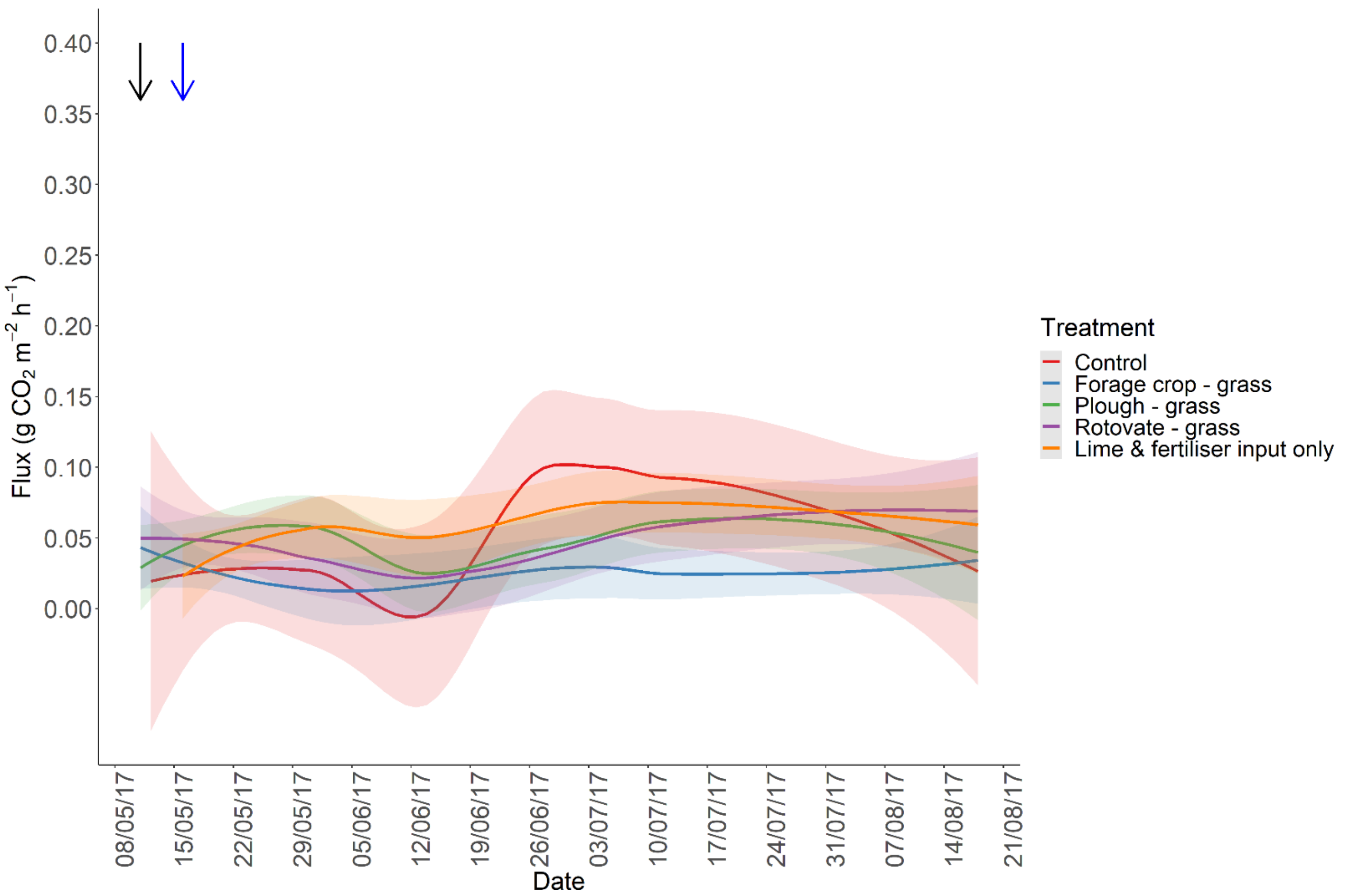

Figure B. Carbon dioxide (g CO_2 m^{-2} h^{-1}) emissions from the treatments in 2017. Coloured lines represent the treatment means (n = 3) and the shaded areas represent the upper and lower bounds of the SEM. Dates are in dd/mm/yy format. Amendments were made at the points of the arrows. The black arrow denotes the ploughing and rotovating and the blue arrow denotes the lime and fertiliser application. Shaded areas appear to exceed the minimum value on the *y*-axis due to the nature of the smoothing curve.

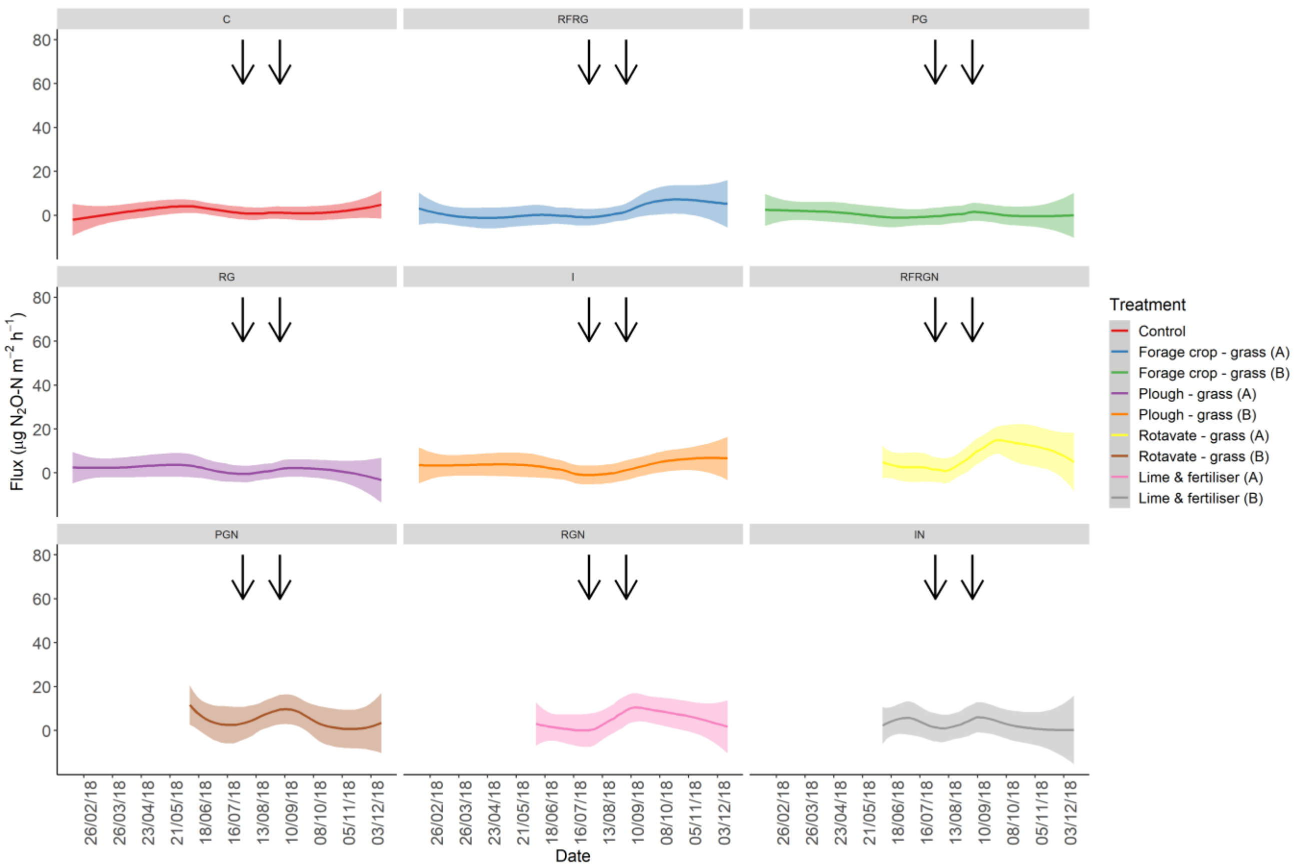

Figure C. Nitrous oxide (µg N_2O-N m^{-2} h^{-1}) emissions from the treatments in 2018. Arrows represent the timing of N fertiliser applications. N was applied to all treatments (control exempt) on 05/06/18. Second and third N applications were made to the required treatments on 31/07/18 and 05/09/18. Coloured lines represent the treatment means (n = 3) and the shaded areas represent the upper and lower bounds of the SEM. Variation in start date of curves is due to split-plotting.

Appendix 4.3 - Supplementary material to Chapter 4 Results

Table A. Daily liveweight gain (kg ha^{-1}) of the grazing treatments for the 2019 monitored period; excluding and including bracken areas.

Treatment	DLWG; excluding bracken areas (kg ha^{-1})	DLWG; including bracken areas (kg ha^{-1})
Improved	2.17	1.96
Unimproved	2.51	1.55

Appendix 5.3 - Supplementary material to Chapter 5 Results

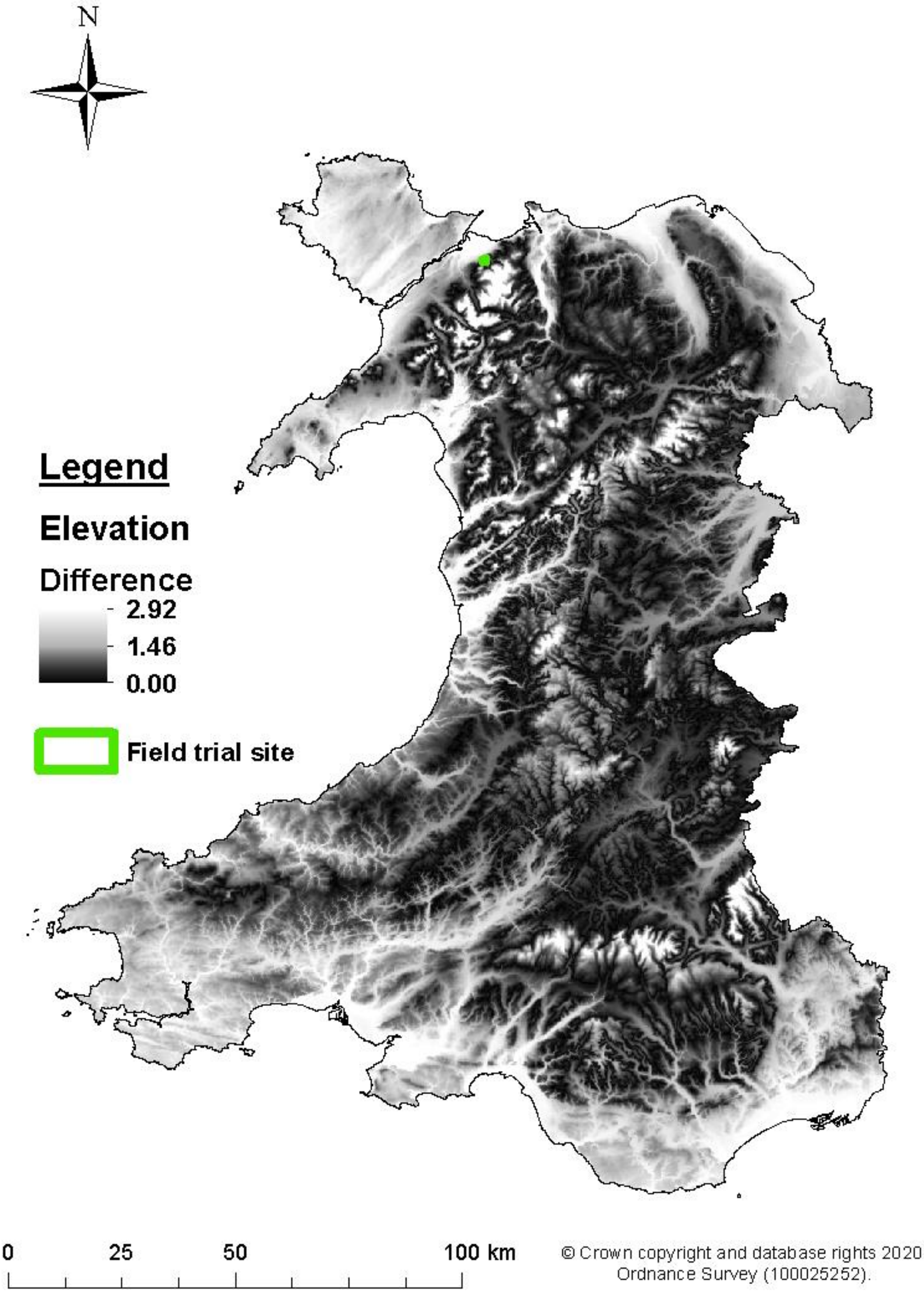

Figure A. Elevation difference, ranging from 0 (field site elevation) and 2.92.

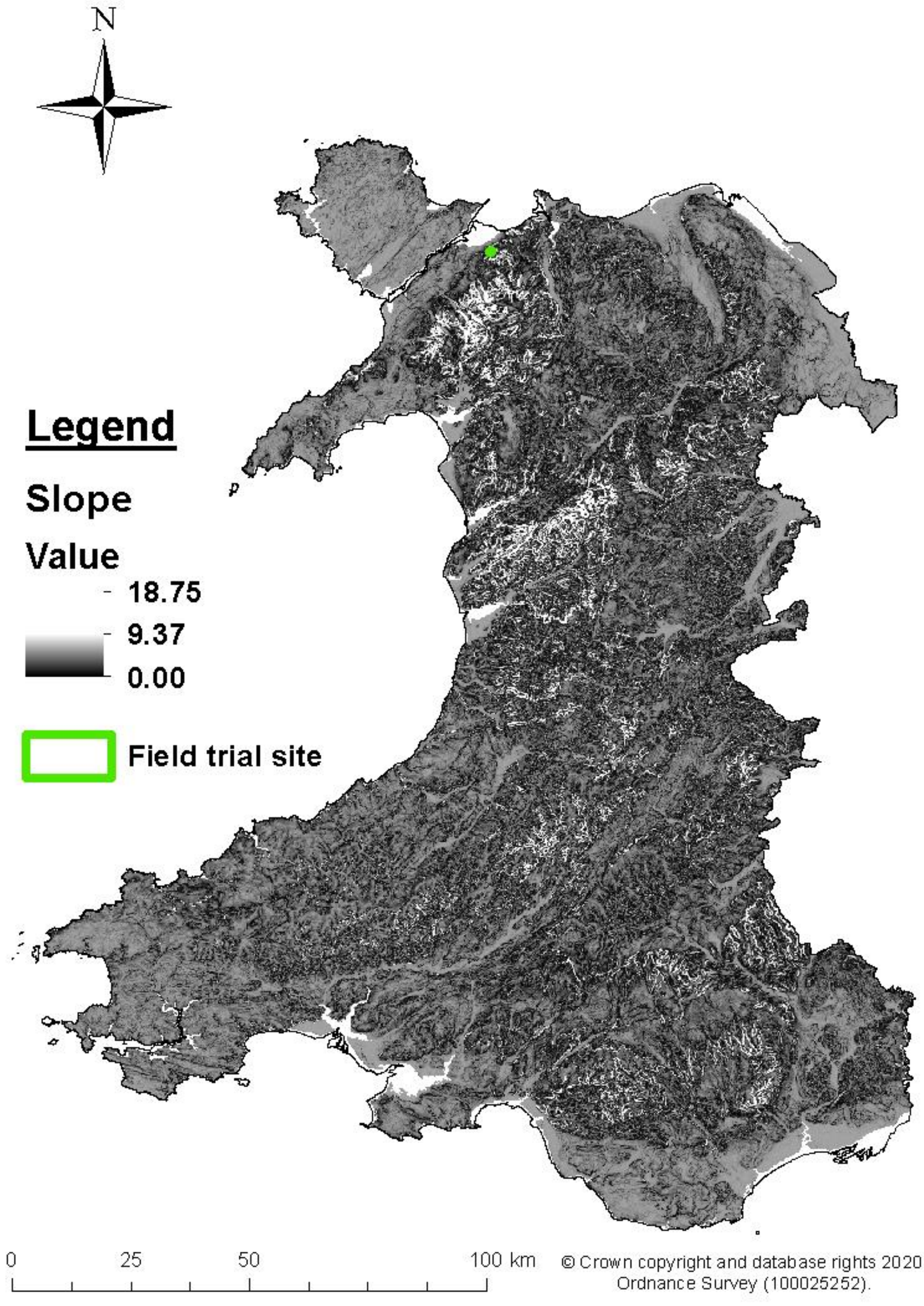

Figure B. Slope difference, ranging between 0 (field site slope) and 18.75.

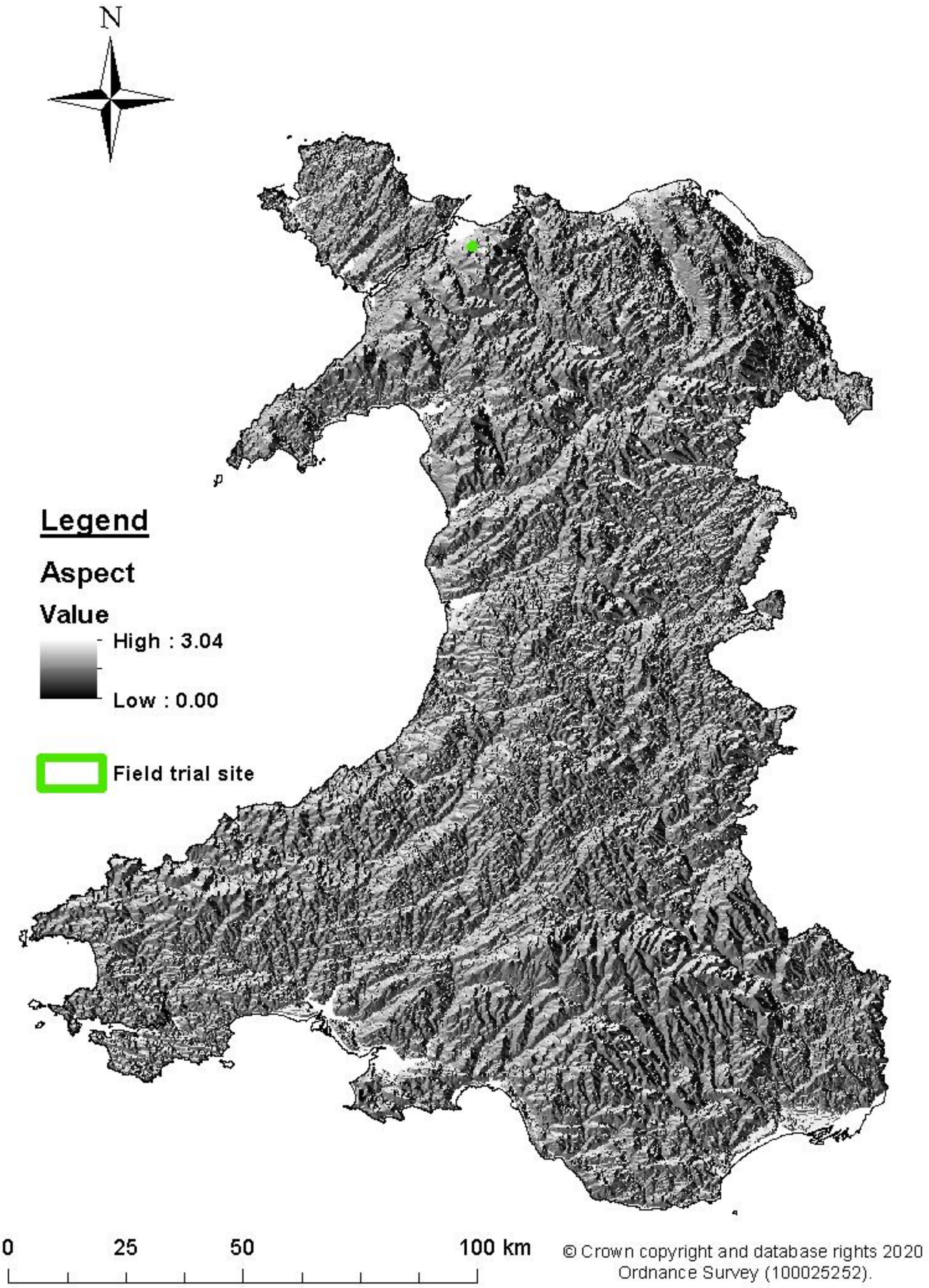

Figure C. Aspect difference, ranging between 0 (field site aspect) and 3.04.

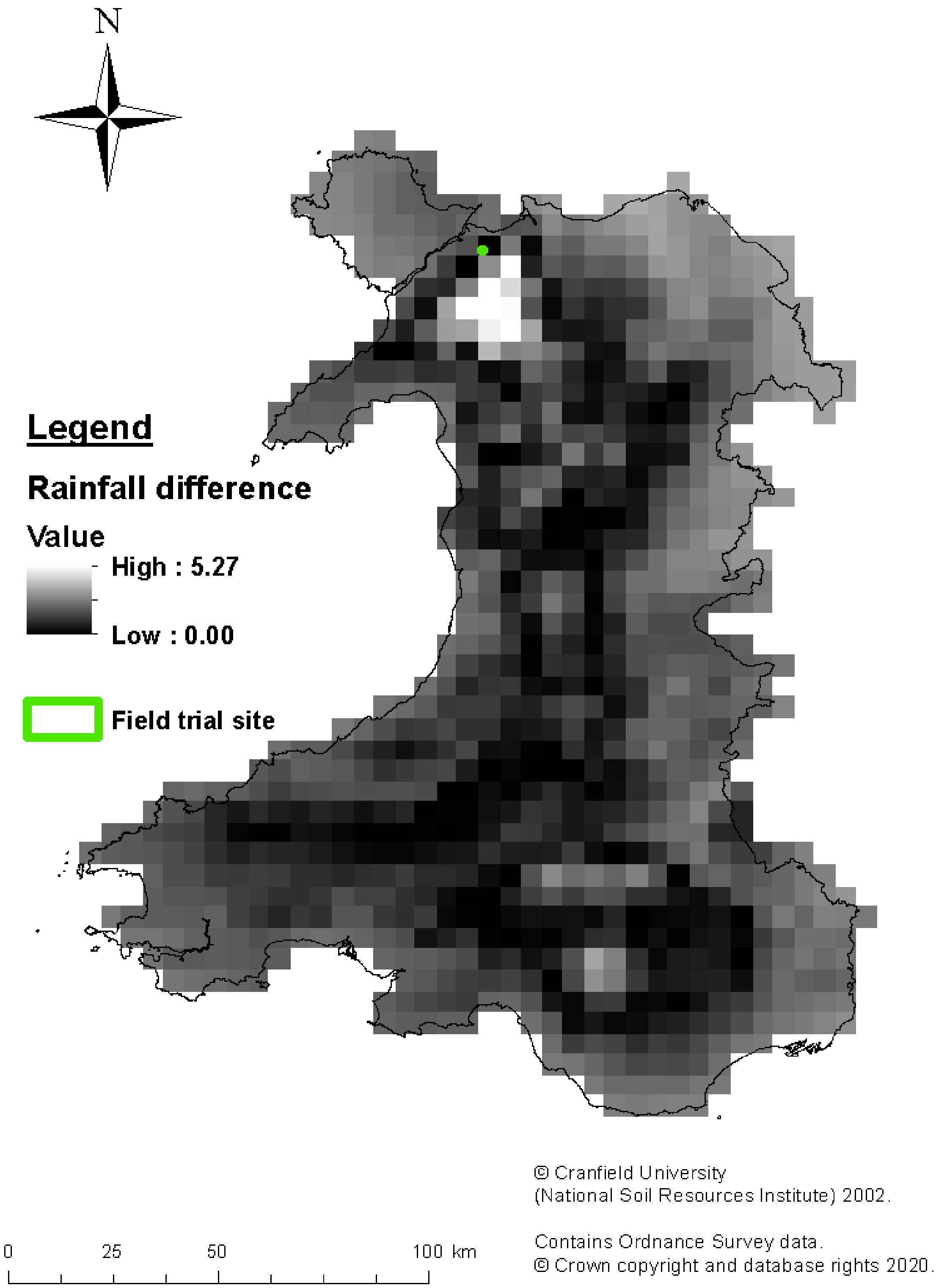

Figure D. Rainfall difference, ranging between 0 (field site rainfall) and 5.27. Rainfall difference is not calculated for some areas around Wales' boundary due to lack of data for these areas.

Table A. Potential grass production and nitrous oxide emissions based on various Euclidean distance measures for the continuous selection (Within 0.25 SD, within 0.5 SD, within 1.0 SD). Values are expressed as an annual total. Total N_2O emissions (N_2O-N converted to N_2O) is expressed as CO_2 equivalent, and used to calculate the proportion of Wales' agricultural GHG production.

Management	Total pasture production (Gg DM)			Total N_2O emissions (Mg N_2O-N)			Proportion of Wales' agricultural GHG production (%)		
	Within 0.25 SD	Within 0.5 SD	Within 1.0 SD	Within 0.25 SD	Within 0.5 SD	Within 1.0 SD	Within 0.25 SD	Within 0.5 SD	Within 1.0 SD
Unimproved	310.08	2,869.26	8,942.55	18.88	174.67	544.40	0.154	1.430	4.457
Lime and fertiliser input only	315.22	2,916.77	9,090.63	18.09	167.39	521.70	0.148	1.370	4.271
Rotovate, spring forage rape followed by autumn grass reseed, lime and fertiliser input	-	-	-	124.85	1,155.24	3,600.52	1.022	9.458	29.477
Plough, spring grass reseed, lime and fertiliser input	670.62	6,205.34	19,340.04	146.00	1,350.99	4,210.60	1.195	11.060	34.471
Rotovate, spring grass reseed, lime and fertiliser input	627.78	5,808.93	18,104.57	204.34	1,890.81	5,893.05	1.673	15.480	48.245

Table B. Cattle liveweight gain based on the continuous selection (Within 0.25 SD, within 0.5 SD, within 1.0 SD).

Management	Actual total cattle liveweight gain based on experimental results (Gg)			Potential total cattle liveweight gain based on potential pasture production (Gg)		
	Within 0.25 SD	Within 0.5 SD	Within 1.0 SD	Within 0.25 SD	Within 0.5 SD	Within 1.0 SD
Unimproved	63.19	584.67	1,822.24	-	-	-
Lime and fertiliser input only	54.59	505.1	1,574.25	64.23	594.36	1,852.41
Rotovate, spring forage rape followed by autumn grass reseed, lime and fertiliser input	-	-	-	-	-	-
Plough, spring grass reseed, lime and fertiliser input	-	-	-	136.65	1,264.47	3,940.96
Rotovate, spring grass reseed, lime and fertiliser input	-	-	-	127.92	1,183.70	3,689.20

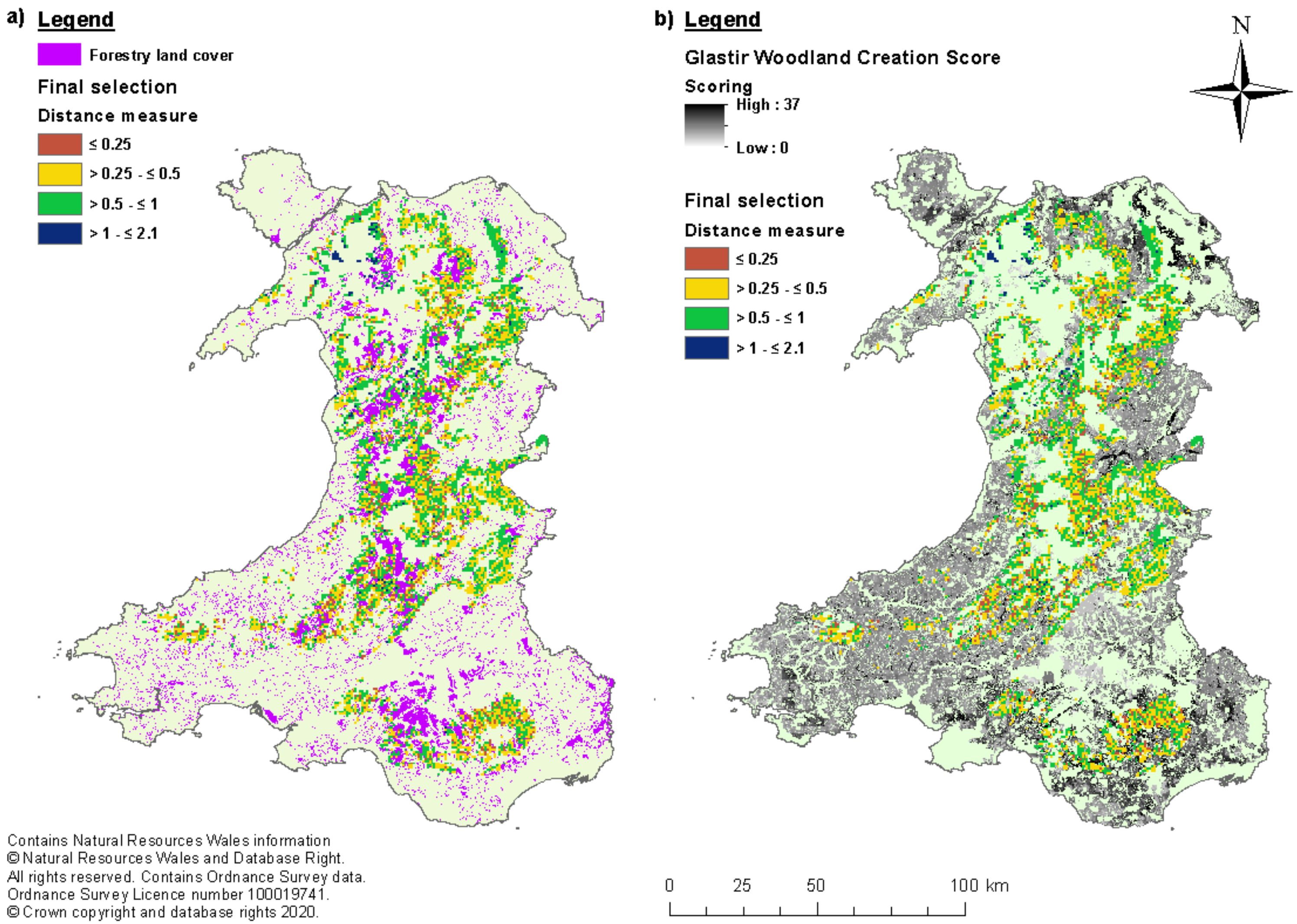

Figure E. Extent of the land areas identified as the final selection for expansion of pasture and therefore livestock production along with distribution of land a) currently managed as woodland and b) determined as potential sites for afforestation in the future.