Environment and Rural Affairs Monitoring & Modelling Programme (ERAMMP)

ERAMMP Report-68:
Review of GHG Emission Reduction and Carbon Sequestration in Agriculture to Inform Agriculture and Land Use Policy

Havard Prosser

Client Ref: Welsh Government / Contract C210/2016/2017
Version 1.0
Date: 21-June-2022

Funded by:
Llywodraeth Cymru
Welsh Government
Canolfan Ecoleg a Hydroleg y DU
UK Centre for Ecology & Hydrology

Version History

Version	Updated By	Date	Changes
1.0	Author Team	21/6/2022	Publication

Mae'r adroddiad hwn ar gael yn electronig yma / This report is available electronically at: **www.erammp.wales/68**

Neu trwy sganio'r cod QR a ddangosir / Or by scanning the QR code shown.

Mae'r ddogfen yma hefyd ar gael yn Gymraeg / This document is also available in Welsh

Series	Environment and Rural Affairs Monitoring & Modelling Programme (ERAMMP)
Title	ERAMMP Report-68: Review of GHG Emission Reduction and Carbon Sequestration in Agriculture to Inform Agricultural and Land Use Policy
Client	Welsh Government
Client reference	C210/2016/2017
Confidentiality, copyright and reproduction	© Crown Copyright 2022 This report is licensed under the Open Government Licence 3.0.
UKCEH contact details	Bronwen Williams UK Centre for Ecology & Hydrology (UKCEH) Environment Centre Wales, Deiniol Road, Bangor, Gwynedd, LL57 2UW 01248 374500 erammp@ceh.ac.uk
Corresponding author	Havard Prosser c/o erammp@ceh.ac.uk
Authors	Prosser, H.
Contributing authors & reviewers	**We acknowledge and are extremely grateful for the input provided by Ken Stebbings and several other external reviewers who provided comments and challenges to the text. This led to additional information and clarifications to the report, from which it has greatly benefited. Thanks to Hugh Martineau for discussion, Robert Matthews for input on forestry and Chris Evans for input on peat soils.**
How to cite (long)	Prosser, H. (2022). *Environment and Rural Affairs Monitoring & Modelling Programme (ERAMMP)* ERAMMP Report-68: Review of GHG Emission Reduction and Carbon Sequestration in Agriculture to Inform Agricultural and Land Use Policy. Report to Welsh Government (Contract C210/2016/2017)(UK Centre for Ecology & Hydrology Projects 06297 & 06810)
How to cite (short)	Prosser, H. (2022). ERAMMP Report-68: Review of GHG Emission Reduction and Carbon Sequestration in Agriculture to Inform Agricultural and Land Use Policy. Report to Welsh Government (Contract C210/2016/2017)(UKCEH 06297/06810)
Approved by	

Abbreviations Used in this Report

3-NOP	3-Nitroxypropanol
AN	Ammonium Nitrate
AHDB	Agriculture and Horticulture Development Board
BAU	Business As Usual
BCMS	British Cattle Movement Service
BNF	Biological N Fixation
C	Organic Carbon
CAN	Calcium Ammonium Nitrate
CARBINE	Forest Research Carbon Accounting model
CP	Crude Protein
CTS	Cattle Tracing Scheme
DCD	DiCyanDiamide
DEFRA	Department for Environment, Food and Rural Affairs
DMPP	3,4-dimethylpyrazole phosphate
ECOSSE	Estimating Carbon in Organic Soils – Sequestration & Emissions model
EF	Emission Factors
ERAMMP	Environment and Rural Affairs Monitoring & Modelling Programme
FYM	Farm Yard Manure
GHG	Greenhouse Gas
GMEP	Glastir Monitoring & Evaluation Programme
GWP	Global Warming Potential
HWP	Harvested Wood Products
IACS	Integrate Administration & Control System
IPCC	Intergovermental Panel on Climate Change
JAS	June Agriculture Survey
LCA	Life Cycle Analysis
LIDAR	Light Detection And Ranging
LULUCF	Land Use, Land-Use Change & Forestry
MACC	Marginal Abatement Cost Curve
MAFF	Ministry of Agriculture, Fisheries & Food
MORECS	Meteorological Office Rainfall & Evaporation Calculation model
NARSES	National Ammonia Reduction Strategy Evaluation System
NIAB	National Institute of Agricultural Botany
NBPT	N-(n-butyl) thiophosphoric triamide
NI	Nitrate Inhibitor
NRW	Natural Resources Wales
NSRI	National Soil Resources Institute
NUE	Nutrient Use Efficiency
REV	Relative Economic Value
RFT	Robust Farm Type
SOC	Soil Organic Carbon
SRUC	Scotland's Rural College
UKCCC	UK Climate Change Commission
UKCEH	UK Centre for Ecology & Hydrology
U	Urea
UI	Urease Inhibitor
WFPS	Water Filled Pore Space
WG	Welsh Government
WSC	Water Soluble Content
WTD	Water Table Depth
ZT	Zero Till

Abbreviations and some of the technical terms used in this report are expanded on in the programme glossaries: https://erammp.wales/en/glossary (English) and https://erammp.cymru/geirfa (Welsh)

Contents

1 Key Messages

A. International research on ways to reduce emissions of methane and nitrous oxide show that the potential reductions are mainly incremental and need to be combined to make significant inroads to emissions reduction**. For agriculture, the maximum technical potential for emission reduction is estimated to be 22.5% for nitrous oxide and 32.4% for methane by 2050**. Taking account of a reduction in agricultural engine emissions by **electric and hydrogen fuel, estimated total emissions from agriculture could decrease from 5603kt CO_2e in 2018 to 3802kt CO_2e by 2050 – a 32.1% reduction**, **based on maintaining the current levels of production.**

B. Expanding woodland expansion by 180,000 hectares by 2050 in line with UKCCC recommendations, coupled with peatland restoration could decrease emissions by a maximum of 2176kt CO_2e/year in 2050. **Forecast net emissions would be 1626 kt CO_2e/year in 2050 from implementing the mitigation measures at the maximum technical potential for agriculture and land use together.**

C. Reduction of methane emissions from ruminants focuses mainly on intensifying production from less animals by improving diet, livestock management and genetics. **But if production is intensified, this can lead to farmers increasing their animal numbers on their productive land depending on input costs.** In turn this leads to increasing emissions since the GHG inventory counts numbers of animals and uses standard emission factors which do not take account of lower emissions from any changes in livestock management which will reduce emissions.

D. Only two recently researched interventions have shown significant impacts in reducing emissions. 3-Nitroxypropanol (3-NOP) suppresses methane formation by up to 40% when included in ruminant diets, but its use is likely to be limited to the dairy sector and beef cattle housed during the winter months, when fed controlled rations. There are no practical ways to use 3-NOP for grazing animals. The use of 3-NOP requires veterinary approval and would require revised emission factors for the GHG Inventory. Without the use of 3-NOP emissions would be 160 kt CO_2e greater – mainly from the dairy herd.

E. The second intervention is the use of DiCyanDiamide (DCD) that suppresses the formation of nitrous oxide from the application of manures and fertilisers to crops and soil by up to 42%. DCD could be added to fertilisers for their application on arable crops and improved grassland – but not extensive grazing land. Regulatory approval would be required for its use in the environment. For the GHG inventory, use of DCD would require a specific emission factor, together with collection of activity data on its use.

F. More efficient use of manures and fertilisers to meet crop requirements, with better storage and application methods for manures can reduce emissions. Biological fixing of nitrogen with legumes and clovers also reduces emissions. In the UK there is a substantial excess of nitrogen input compared with outputs – showing the substantial loss of nutrients to the environment, and the substantial potential to continue reducing fertiliser use by a further 50%. This will be incentivised by the recent large increase in fertiliser prices. The reduction would be reflected in the GHG inventory.

G. Widespread and ongoing peat restoration has contributed to a reduction in total emissions, but the majority of restoration has taken place within modified upland bogs, with modest emissions. **Much greater reductions would be obtained from restoration of degraded lowland peats used for agriculture and plantation forestry.**

H. Farmers are looking to use carbon sequestration as a potential way of offsetting emissions**. However offsetting to achieve net-zero emissions is not possible. Claims that permanent grassland continues to sequester carbon are not substantiated by research evidence.** The only exception is where grassland has been established recently on arable land and on degraded soils. Emissions would also be reduced where lowland peats are restored.

I. In Wales, the high proportions of permanent grassland and soils already high in carbon severely limit the potential for further carbon sequestration. The essential requirement is to maintain existing carbon stocks in soils and peats to reduce carbon dioxide emissions. **The management methods would not be counted in the GHG inventory.**

J. It is important to recognise the other benefits of reducing GHG emissions. Measures to reduce emissions of nitrous oxide have multiple benefits in improving air and water quality and reducing eutrophication of habitats. Reduction of methane emissions also has a secondary benefit in reducing ozone formation in the lower atmosphere (up to 15km). Ozone itself is a greenhouse gas and causes human health problems as well as damage to vegetation.

K. **The recent research has shown the maximum technical potential to reduce emissions, but the practical delivery of these options by Welsh farmers depends crucially on the economic, regulatory and cultural framework to motivate uptake.**

2 Executive Summary

This report updates previous Welsh Government reviews in 2008 and 2010. This review has concentrated on the large amount of research that has been done in the last decade to find ways to reduce the GHG emissions from the main sectors – soils and land use, manures and fertilisers and livestock farming – specifically the dairy, beef and sheep sectors. **This report makes the assumption that existing production levels are maintained, but public health pressures to reduce the consumption of red meat and dairy products will challenge this assumption.** UKCCC 6th Report advocates a substantial reduction in the consumption of meat and dairy products (UKCCC, 2020).

Reduction of GHG emissions from agriculture and land use concentrates on:

- Reducing methane emissions from ruminants
- Reducing nitrous oxide emissions from the application of manures and fertilisers on soils
- Reducing emissions from manure storage, handling and application
- Maintaining carbon stocks in soils and peats,
- Expanding carbon sequestration in biomass and soils to offset emissions from other sources.

Ruminant methane emissions can only be reduced in the inventory by reducing the numbers of animals, reducing animal numbers while increasing individual animal productivity could maintain current production levels. But there is the natural tendency for farmers to use all of their land for production, whereas the freeing up of land for increasing biodiversity, sequestering carbon or flood mitigation would help to deliver public goods. The incentives to do this need careful consideration to avoid increased productivity leading to increased animal numbers and increased emissions.

Reduction of methane emissions from ruminants requires good quality diets high in digestible fibre. Historically, genetic improvement has improved productivity. Specific genetic improvement to reduce methane emissions in the rumen is still in the early stages of development. Much of the genetic development has focused on the dairy sector, with less development in the beef and sheep sectors. Other options for reducing methane emissions are based on a range of diet additives, of which 3-nitropropanol (3-NOP) is proving to be most effect in suppressing methane generation. One option is to develop ways to capture methane from cattle sheds. Work in Canada suggests that this may be a useful mitigation option. (Fedrizzi et al., 2018).

Livestock management on Welsh farms shows large variations between best and worst performing enterprises (Jones et al., 2014). On sheep farms, high lambing percentages and rates of weight gains are important to deliver emission reduction per unit of production.

The main way of reducing nitrous oxide emissions is to make smarter use of fertilisers calibrated to plant needs. Improving efficiency of use coupled with analysis of the contribution of manures to crop needs is essential. Since 2010, research on nitrate inhibitors applied with fertilisers has shown significant emission reduction can be achieved although results are variable. Reseeding of grasslands with legumes to fix nitrogen from the atmosphere is another way to reduce fertiliser use.

Emissions from manures can be reduced by covers over lagoons and storage heaps. Methane capture techniques are being developed for stores for subsequent conversion of the methane in combustion or biofiltration. Other techniques including acidification of slurries reduce methane emissions, used on Danish farms. Application of slurry manures by slit injection into soils is helpful in reducing emissions of both nitrous oxide and ammonia. Also the reduction has a secondary benefit of reducing the amount of mineral fertiliser required to meet crop needs. There are other important benefits in reducing air pollution from fine particulates formed from ammonia and reducing pollution from run-off into water courses. The Control of Agricultural Pollution Regulations (Wales) 2021 should also reduce GHG emissions and pollution incidents. Table 2.1 summarises the maximum technical potential for emission reduction from reviewing the options for managing livestock, manures and fertilisers.

Table 2.1 The Maximum Technical Potential reduction from the management of livestock, manures and fertilisers

Mitigation method	**Nitrous oxide reduction** (kt CO_2 e)	**Methane reduction** (kt CO_2 e)
Manures and Fertilisers		
N fertiliser and manure efficiency – 50% reduction of fertilisers	236	
N inhibitors on remaining fertilisers and all manures 40% reduction	138	
Reducing methane emissions from housing and stores by 50% by acidification, methane capture and/or anaerobic digestion -		205
Livestock management		
Diet, management, genetics and 3-NOP in dairy herd - 43% emissions reduction		341
Diet, management, genetics and 3-NOP in beef herd – 30% emissions reduction		356
Diet, management and genetics in sheep flock – 30% emissions reduction		215
Total Emissions (2018)	**1660**	**3449**
Total Emission Reduction	**374**	**1117**

Taking account of a potential reduction of 310 kt CO_2e from the agriculture vehicle sector by the use of electric and hydrogen or methane fuels (UKCCC, 2020), this would reduce agriculture inventory emissions by 1801 kt CO_2e.

Maintaining the high levels of carbon in soils and peats is important to reduce carbon dioxide emissions. For soils, the main emphasis is on maintaining the fertility and minimising the conversion of grassland to arable cropland. Although there are claims about the significant sequestration capability of grasslands, the research evidence is that the capability is limited to the first few decades after arable or degraded soils are converted to grassland. Grassland management has no clear benefits for sequestration. For permanent grassland, the research evidence is that sequestration is very small. Agroforestry and energy crops have potential benefits, recognising that soil disturbance at the time of initial planting will lead to some emissions of carbon dioxide.

For peats and organic soils, the main mitigation measure is to rewet existing areas to prevent carbon dioxide emissions. Increasing peatland areas has the danger of accelerating methane emissions in anaerobic conditions unless there is careful control of levels of the water table.

The main opportunity for carbon sequestration is to expand forestry, woodlands and hedgerows which sequester carbon in the vegetation and the soils. Modelling of expanding woodland by 100,000 hectares/year by 2050 has shown that a significant carbon sink in soils and vegetation could be obtained, but not enough to offset agricultural emissions. Welsh

Government has presented its target to plant 43,000 hectares of new woodland by 2030, and 180,000 hectares by 2050, aligning with the Balanced Pathway set out by the UKCCC (Welsh Government, 2021). Table 2.2 summarises how much emissions from the agriculture and LULUCF sectors could be reduced by 2050 if the forecast Maximum Technical Potential is attained.

Table 2.2 The Maximum Technical Potential from the Agriculture and Land Use, Land Use Change and Forestry (LULUCF) sectors

Sector	**GHG Emissions in 2018**	**Forecast Annual GHG Emissions in 2050**	
Agriculture	5603 kt CO_2e	3802 kt CO_2e	
LULUCF	-444 kt CO_2e	-1872 kt CO_2e (1)	-2176 kt CO_2e (2)
Total	5159 kt CO_2e	1930 kt CO_2e	1626 kt CO_2e

Note: (1) is based on woodland expansion of 100,000ha by 2050
(2) is based on woodland expansion by 180,000ha by 2050

Changes to farming and land use practices will have economic costs and benefits. To assess these, Marginal Abatement Cost Curves (MACC) have been estimated. Recent work for England (Moran et al, 2020) has shown that the most cost-effective options are

1. Improved crop management to make effective use of manures and fertilisers by precision farming, with improved crop varieties, high sugar grasses and legumes to improve biological nitrogen fixation.
2. Improved livestock management and genetics to improve productivity per animal and to reduce methane emissions.
3. Improved manure management

Use of nitrate inhibitors added to fertilisers incurs significant costs per tonne of emissions saved and the use of 3-NOP feed additive also incurs a cost. Similar conclusions have been found in Scotland (Eory et al. 2020).

Apart from the economic aspects, important factors for uptake of measures by English farmers included the self-identity of the farmer and social norms, the management of risk, the acquisition of knowledge, and the long-term planning constraints due to lock-in effects of assets and practices. In addition the role that current practices have on determining how new ones fit, family life cycle factors, the entry of newer farmers into the industry, and institutional factors such as supply chain constraints have an influence in implementing change.

From a survey of farmer attitudes in England in 2020, 18% of farmers reported that it was “very important” to consider GHGs when making decisions relating to their land, crops and livestock and a further 46% thought it “fairly important”. However 30% of respondents placed little or no importance on considering GHGs when making decisions, or thought their farm did not produce GHG emissions. This situation in England in 2020 may be similar in Wales, but requires confirmation.

The thinking of farmers in England has been reflected in the slow progress in reducing emissions set out in the England Action Plan. Although the target was to reduce agricultural production emissions by 3 $MtCO_2$e by 2020 compared to a 2007 baseline, only a 0.9 $MtCO_2$e reduction had been achieved.

Clearly there is a substantial training programme required with a range of regulatory and financial incentives. Access to practical advice through Farming Connect and the Levy Boards is essential to increase uptake of mitigation measures. Financial and regulatory incentives will be required to make emissions savings that approach the Maximum Technical Potential.

The data compiled for the GHG inventory will require a better understanding of manure and fertiliser management systems across farms. Livestock management methods will also need to be estimated in greater detail. More accurate data on emission factors for different systems and the collection of farm activity data will be needed to reflect actual emissions. This would improve the inventory and reward individual farmers for any actions they take to reduce their own farm emissions.

3 GHG Inventory and Methodology

The UK and Wales GHG emission Inventories use Global Warming Potentials based on the Fourth Assessment Report (IPCC, 2007). Values for methane are 25 and 298 for nitrous oxide based on a 100-year horizon. The GWPs are slightly different from the ones used in the Welsh Government 2008-2010 reports. In this review the baseline considered was the 2018 GHG inventory expressed as kilotonnes of Carbon dioxide equivalent ($ktCO_2e$). Table 3.1 summarises the GHG inventory for Wales in 2018 (Welsh Government 2021)

Table 3.1 GHG Inventory for Wales – Agriculture and Land Use Sectors – 2018 (in units of kilotonnes of Carbon Dioxide Equivalent - kt CO2e)

Emission Category	Emission Source	CO_2	CH_4	N_2O	Total
Agricultural Engines	Mobile	469		52	521
	Stationary	25		3	28
Enteric	Dairy		792		792
	Other cattle		1188		1188
	Sheep		981		981
	Others		43		43
Manures	Dairy		243	47	290
	Other cattle		166	153	319
	Sheep		25	6	31
	Other livestock		11	72	83
Agricultural soils	Fertiliser application			472	472
	Manure/sewage application			110	110
	Grazing – direct deposition			145	145
	Crop residues			69	69
	Mineralisation from loss/gain of Soil Organic Material			34	34
	Cultivated organic soils			296	296
	Indirect from leached N compounds			92	92
	Indirect from atmospheric N deposition from agriculture			43	43
	Liming			58	58
	Urea application			8	8
Sub-totals		**494**	**3449**	**1660**	**5603**
Land Use, Land Use Change and Forestry					**-444**
Energy Supply		11929			11929
Public Institutions Energy Supply		326			324
Residential		3699			3699
Transport		6171			6171
Waste Management			1244		1244
Industrial Processes		1867			1867
Business		8500			8500
Grand Total					**38891**

Emission estimates are based on multiplying activity data – e.g. number of livestock - by Emission Factors (EFs). Defra and devolved administrations funded the Agricultural UK GHG Platform to refine the EFs to be used for livestock and management systems. Each EF needs to be multiplied by the activity e.g. number if dairy cows, number of followers, manure management systems, etc. Activity data come from a variety of sources, primarily the June Agriculture Survey for agricultural emissions and Countryside Survey and Forest Research for land use data. Details of the methodologies are available from UK GHG Inventory reports (Brown et al. 2021).

Activity data for Wales is often not disaggregated for England and Wales or UK data. Various initiatives are in progress to use routine agriculture data on crops and livestock movements to improve activity data (H Martineau – personal communication). It is important to recognise that as the methods of analysis become more refined they are applied both to the latest estimate AND to the baseline year (1990). For example, reducing an EF by 50% may not have much overall effect when comparing the emissions for the current and baseline year.

For agriculture, the UK GHG Platform project (DEFRA AC0114, AC0115 and AC0116) (DEFRA, 2013) has enabled a move from standard IPCC Emission Factors – designated Tier 1 to UK specific EFs for groups of animals, soil types, manure storage methods and manure application methods – Tiers 2 and 3. Use of specific EFs for particular management practices requires detailed collection of activity data to cover the following categories of sources

- Enteric CH_4,
- Manure management CH_4 and N_2O emissions from Dairy, Beef, Sheep, Swine, Poultry, Goats, Horses and Deer;
- Direct and indirect N_2O emissions from synthetic N fertiliser, organic N (e.g., animal manure, sewage sludge and digestate) applied to grassland and arable crops, and crop residues;
- N_2O emissions from urine and dung N deposited on pasture from Dairy, Beef, Sheep, Swine, Poultry, Goats, Horses and Deer.
- N_2O emissions from mineralisation of soils and CO_2 emissions from liming and application of urea to soils

Data are collated and calculations are performed at a 10 x 10 km grid cell resolution.

For the LULUCF inventory, the major changes are:

- The CARBINE model has been developed for forests and woodlands (Matthews, 2020), and for forest soils using the ECOSSE soils model. The driving data comes from the National Forestry Inventory and planting grants/licences.
- For other land uses and changes, the carbon stock for non-organic soils is based on the database of soil carbon density to 1m depth (NSRI). Carbon changes come from the initial and final land use. Biomass carbon stock change comes from a Land-use change matrix. N_2O emissions for non-organic soils come from IPCC Tier 1 EFs. Classification of soils comes from the Agriculture and Horticulture Development Board (AHDB) RB209 soil classification. Activity data comes from Agriculture census data, land cover map 2015, and Countryside Survey. Cranfield University provides the soil emissions data for non-organic soils.
- For organic soils e.g. peats and wetlands, emissions are based on the report by Evans et al. 2017. This covers EFs and area assessments, with scenarios on future peatland areas and their management.

Within the agriculture sector there has been debate that the emissions of methane should be treated differently because the lifetime of methane in the atmosphere is shorter (about 12 years) (Chen et al. IPCC 6th Report, 2021) than for carbon dioxide and nitrous oxide. CO_2 emissions last in the atmosphere for a long time and are cumulative as observed by the increasing concentration in the atmosphere (IPCC 6th Report, 2021). In contrast, because it breaks down rapidly, methane emissions do not act cumulatively. For a constant rate of methane emissions, one molecule in effect replaces a previously emitted one that has since broken down into carbon dioxide. Reduced methane emissions would lead to the atmospheric concentration falling relatively quickly.

Nonetheless, CH_4 concentrations have increased from 729 ppb in 1750 to 1866 ppb in 2019, with a 63 ppb rise in atmospheric CH_4 concentrations between 2011 and 2019. The oil and gas industries are major sources of methane whilst agriculture emits 47% of methane emissions in the UK.

To overcome the problem of methane being short-lived, some researchers have proposed alternative metrics to replace carbon dioxide equivalents (CO_2e) using the 100-year Global Warming Potential (GWP100) (Allen et al, 2018, Cain et al. 2019). GWP* has been proposed based on the *change in rate* of methane emissions, compared to the *total amount* of CO_2. GWP*, allows emissions of short-lived and long-lived climate pollutants to be more consistently expressed within a single metric by equating a change in the emission rate of a short-lived climate pollutant as equivalent to a single emissions pulse of a long-lived pollutant. Using GWP*, changing the rate of methane emissions is assigned *a much higher CO_2-equivalence* than it is under GWP100; as such it better captures the risks of increasing and the benefits of decreasing methane emission rates.

At present the UN Framework Convention on Climate Change (UNFCC) retains the GWP – 100 year assessment method for methane (UNFCC, 2019). The short lifetime of methane means that reducing methane emissions would have a rapid effect in reducing climate change to meet the target of limiting global average temperature rise to 1.5C. COP26 has agreed a target for reducing methane emissions to contribute to the limiting of global temperature rise.

The major challenge is to estimate the changes in GHG emissions using these more sophisticated models instead of the simple Tier 1 EFs used in 2008-2010. Also the availability of the activity data and its sensitivity to change is a challenge e.g. Countryside Survey and Land Cover data to monitor changes of land use, changes of hedgerows, changes to manure management etc.

4 Land Use, Land Use Change and Forestry (LULUCF)

4.1 Introduction

This chapter describes the role of LULUCF as a source of GHG emissions, and as a sink for GHG emissions. The emissions are reviewed, together with mitigation options and scenarios for emissions to 2050.

4.1.1 Carbon sequestration and carbon storage in soils

It is important to distinguish between carbon sequestration and carbon storage. Carbon sequestration is the process of transferring CO_2 from the atmosphere into the soil, through plants, plant residues and other organic solids which are stored or retained as part of the soil organic matter (humus). The retention time of sequestered carbon in the soil (terrestrial pool) can range from short-term (released back to atmosphere) to long-term storage over millennia. The sequestration rate diminishes to zero over a period of decades as soils reach a new state of carbon equilibrium, and gains can be lost if soils are disturbed by ploughing and cultivation. Conversely conversion of arable to grassland can lead to sequestration rates of 1.01 t C/ha/yr until a new equilibrium is attained (Conant et al, 2001).

Carbon storage is broader as it is the increase in SOC stocks in the soils, not necessarily associated with a net removal of CO_2 from the atmosphere. For example adding the available manure resources on a given agricultural field may locally increase SOC stocks (where manure has been added), but not increase the associated CO_2 removal from the atmosphere at the landscape scale.

While storing organic carbon for long times is preferable in terms of GHG mitigation, labile fractions of SOC (e.g. with residence times of months to years) are essential in terms of soil fertility, soil physical condition (aggregate stability largely depends on labile carbon) and of soil biodiversity - organic matter being the trophic resource of organisms. Hence it is desirable to increase stocks of both labile and stable forms of organic matter.

Carbon is incorporated into the plant and some of this carbon will be in its above ground biomass (stem, leaves, flowers, seeds), and some in its root structure. When plants and other forms of biomass die and decay most of this carbon is emitted back to the atmosphere as CO_2 over a period of weeks or months, and the net effect on atmospheric CO_2 concentrations is therefore small. But some will be stabilised in the soil.

Inputs of carbon to the soil include dissolved organic matter, root products, and transported particulates from the surface. Whereas litter and living root inputs could supply two distinct SOC formation pathways, Sokul et al. (2018) found that living roots were the overall dominant conduit of carbon to a mineral soil. Relevant root traits that may affect soil carbon formation include root branching intensity, root depth distribution and specific root length (Poirer et al, 2018).

Most soil carbon derives from below-ground inputs and is transformed, through oxidation by microorganisms, into the substances found in the soil (Bradford et al, 2013). The persistence of organic matter in soil is largely due to complex interactions between organic matter and its environment, dependent on compound chemistry, reactive mineral surfaces, climate, water availability, soil acidity, soil redox state and the presence of potential degraders in the immediate microenvironment (Six et al, 2002).

Using data from long term studies, Smith (2014) suggested that soil C stocks may reach a relatively stable equilibrium around 100 years after a land use change, but the main changes occur within the first 40 years ((Conant et al, 2001). Twenty years is often chosen as the time for changes in soil carbon to cease following a change of land use by IPCC (Eggleston et al., 2006).

4.1.2 Soils in Wales

The soils and biomass of Wales provide a valuable sink in sequestering carbon dioxide, but managed poorly, they can become a source of emissions. The allocation of land use in Wales (Welsh Government, 2020) is shown in Table 4.1, based on the June Agriculture Survey, and the Countryside Survey Land Cover Map. Both are for the 2015.

Table 4.1. Land allocation in Wales in 2015

Category	**Area – June Ag Survey (ha)**	**Area – Land Cover Map (ha)**
Arable	90,144	99,777
Temporary grass	157,501	-
Permanent grass	1,068,814	984,555
Rough grazing - sole rights	257,264	339,965
Commons	180,306	180,305
Woodland on farms	77,961	83,004
Other land	11,998	-
Total Area of Farms	1,843,988	1,687,606
Forest/woodland	310,000	220,716

There are differences in the area estimates, because the June Agriculture Survey is based on farmers' interpretation of the management of their farms, whilst the Land Cover Map is based on interpretation of remote sensing data.

LULUCF activities can result in net emissions and removals of GHG, and changes in carbon stocks in the pools associated with LULUCF. LULUCF is divided into six land use types: Forest Land, Cropland, Grassland, Wetlands, Settlements, and Other Land. Carbon stock changes for Harvested Wood Products are reported as an additional category.

For Wales, the GHG Inventory 2018 reports a net sink of 443.73 kt CO_2e, dominated by the carbon sequestration provided by existing forestry – a sink of 1209.74 kt CO_2e – see Table 4.2 (Welsh Government, 2021)

Table 4.2 Main Sources and Sinks - Wales GHG LULUCF Inventory – 2018

Source	Emissions (2018) (kt CO2e)
Indirect N_2O	22.74
Forest remaining Forest	-1209.74
Other land categories converted to forest	17.95
Cropland remaining cropland	412.52
Grassland converted to cropland	463.62
Grassland remaining grassland	-278.96
Forest converted to grassland	100.69
Cropland converted to grassland	-253.91
Wetlands and land converted to wetlands	-
Settlements converted to grassland	-98.13
Settlements remaining settlements	289.89
Forest converted to settlements	22.29
Cropland converted to settlements	24.49
Grassland converted to settlements	363.09
Harvested Wood Products	-296.00
Total	-444

Note: Minor emissions have been omitted

4.2 Forestry

4.2.1 Carbon Stocks

This section is based on the report of Matthews (2020), on forestry and woodland in Wales. Woodlands and forests are major reservoirs of carbon both in the soil and the biomass. Results from the BioSoil study of 166 woodland sites in Great Britain produced total soil carbon stocks for seven different broad soil types with mean values ranging from 108 tC/ha to 539 tC/ha down to 1m depth (Vanguelova et al. 2013). Values of carbon stock in the above ground biomass in a woodland are typically 50 – 170 tC/ha meaning that the soil carbon stock can be at least as large as that in the above ground biomass, and in some cases considerably more. In the Biosoil survey, mineral soil types had a carbon stock of 108 – 173 tC/ha, (so broadly similar to the stock in trees) but organo-mineral and organic soils had considerably higher stocks (mean of 36 peaty gleys = 362 tC/ha and 14 deep peats = 539 tC/ha (Vanguelova et al. 2013).

Consequently, the management of the soil carbon stock can have an important impact on the overall woodland carbon balance, particularly for organo-mineral and organic soils. In addition to this soil carbon stock, the litter layers (including both the true litter layer, and the fermentation or F layer, (consisting of partially decomposed matter), may contain an additional 12-20 tC/ha (the mean for the BioSoil survey sites is 16 tC/ha).

For Wales, Vanguelova et al. (2013) assessed the area occupied by different soil types and under coniferous and broadleaved woodland. Based on the BioSoil measurements of carbon stock per area, the estimated total woodland soil carbon stock was 51 MtC. The majority of the existing woodland area (in 2003) was on brown earths, podsols and peaty gleys/podsols. The areas of coniferous woodland on peaty gleys/podsols and deep peats, although only 21% of the area, contributed 42% of the woodland soil carbon stock because of the high carbon stock of these organo-mineral and organic soil types. The additional carbon stock in the litter layer in woodlands was estimated for Wales to be 4.6 MtC (Morison et al. 2012)

4.2.2 Mitigation Options

There are a number of specific forest management activities that can contribute to climate change mitigation:

- Creation of new woodland areas (afforestation)
- Forest management including prevention of woodland loss (avoidance of deforestation), conservation or enhancement of carbon in existing woodlands.
- Enhancement of production, e.g. through increased harvesting in existing woodlands, to achieve substitution/displacement impacts in other sectors.

4.2.2.1 Afforestation

Matthews (2020) used the CARBINE model using the ECOSSE soil model (Smith et al) to forecast the carbon sink to 2100, for planting at the rate of 4000 ha per year from 2018 until 2040, then 1000 ha per year thereafter (Stretch scenario of approximately 100,000ha expansion). Initially the GHG balance is dominated by net CO_2 emissions from loss of soil carbon stocks (2.0 tCO_2/ha/yr), which occur as a result of site preparation and the time involved in the transition between the loss of pre-existing vegetation on the site and the full establishment of the trees.

For conifers, over a time horizon to 2050, although CO_2 emissions from loss of soil carbon stocks remain significant, these emissions are more than balanced by carbon sequestration in the living biomass of trees, soil biomass and in deadwood and litter (-3.8 tCO_2/ha/yr, as the trees grow in their full-vigour phase. There is also a modest contribution to carbon sequestration in the form of carbon stock increases in wood products from thinnings (-0.4 tCO_2/ha/yr). Across all relevant management regimes, coniferous tree species consistently exhibit significant net CO_2 sequestration and GHG emissions reductions.

For new broad leaf woodland, taking oak as an example, in the period 2020-2050 CO_2 emissions arising from losses of carbon stocks in soil almost completely offset carbon sequestration in trees, deadwood and litter. Losses of soil carbon stocks are similar for conifers and broadleaf trees but carbon sequestration in the broadleaf trees takes longer to reach the full vigour phase. Consequently, annualised total net carbon sequestration in the oak woodland is almost negligible for the period 2020-2050, but net carbon sequestration over longer time horizons is sustained. Results for different broadleaf tree species are variable. The best sequestration rates are for birch trees because of a higher tree growth rate.

Once wood production comes on stream, product substitution for other building materials and wood fuel makes a sustained contribution to GHG emissions reductions of about -3.5 tCO_2/ha/yr. Across all relevant management regimes, coniferous tree species consistently exhibit significant net CO_2 sequestration. The fact that they reach maturity about 60 years after planting means that they can be harvested for wood products, and a new tree crop can be planted, to start a new cycle of initial emissions and then sequestration.

Relatively high climate change mitigation potentials can be obtained from the introduction of tree species or varieties with superior growth rates (e.g. genetically improved Sitka spruce), at 3 tCO_2e/ha/yr over the period 2020 to 2050 and 5.5 tCO_2e/ha/yr over the period 2050 to 2100 (the latter estimate including a significant contribution from substitution of timber for high energy material such as steel and concrete).

4.2.2.2 Forest management

A quantitative assessment by Matthews (2020) gave the following broad estimates for the climate change mitigation potentials of forest management activities:

- The creation of short rotation forestry plantations (for raw biomass rather than timber production) can mitigate between 1 and 1.5 tCO_2e/ha/yr as a fuel substitute over the period 2020 to 2100.
- The avoidance of woodland loss can mitigate between 55 and 120 t CO_2e/ha where opportunities exist to halt or reduce activities that involve deforestation.
- Adjustments to the management of existing woodlands to conserve or enhance woodland carbon stocks and sequestration can mitigate between 1 and 2.5 t CO_2e/ha/yr over the period 2020 to 2050 and between about 0 and 2 t CO_2e/ha/yr over the period 2020 to 2100.
- Adjustments to the species composition and growth rates of existing woodlands, to enhance wood production whilst maintaining carbon stocks, give variable outcomes. The overall growth rates of trees in diversified woodlands need to increase to improve sequestration.

4.2.2.3 Production enhancement

Matthews (2020) concluded that certain adjustments to the management of existing woodlands may contribute towards maintaining carbon stocks. The main interventions are:

- Deferring final harvest (clear-felling) in even-aged commercial woodlands, by extending rotations
- Transformation of woodlands from even-aged management to continuous cover management, generally by avoiding large-scale clear-felling and maintaining tree cover by developing an uneven-aged structure in woodlands
- Restricting or avoiding tree harvesting in woodlands, with the aim of maximising the accumulation of carbon stocks in trees and soil, possibly requiring transformation of woodlands to be composed of enduring tree species.
- Conservation of long-established woodlands with high carbon stocks.

4.2.3 Summary

The expansion of woodlands represents a major way that GHG emissions can be reduced by sequestration in the soils and biomass. After initial emissions due to soil disturbance and loss of existing vegetation, there is a time lag between planting and maximum sequestration - a shorter period for conifers than for broad leaf trees. Conifers have a typical life cycle of about 60 years between planting and maturity whereas broadleaf species such as oak will grow for 100 years or more. In the short term to 2050, conifers would provide the largest carbon sink, whereas broadleaf trees sequester carbon in the period 2050-2100 and beyond.

Woodland management can make minor contributions to sequestration.

In addition, harvested wood products retain (i.e. effectively sequester) carbon in the woody biomass from which they are made. Wood products have relatively low inputs of energy and other non-renewable resources in their manufacture. Hence, the GHG emissions involved in manufacturing wood products can be relatively low, compared with equivalent products made from concrete and steel.

Harvested wood biomass can also be used as a fuel to replace fossil fuels. However, there is a large variation of results from Life Cycle Analyses on the overall benefits for GHG emissions.

4.3 Agricultural Soils

The ERAMMP report (Alison et al. 2019) on soil carbon management has been used to provide information for this section.

4.3.1 Sequestration Options

The role of agriculture soils as a carbon sink linked to its sequestering capacity is conflicting with a range of studies based on

- measurements of changes in soil carbon stocks and
- direct measurements of emissions of carbon dioxide from soils

An important factor determining sinks and sources is the high soil organic content of Welsh soils, mainly associated with permanent grassland and the uplands. Although the Welsh soil carbon stock is estimated to be 409 Mt carbon (1,499 Mt CO_2 e) (Bradley et al. 2005), estimates of soil carbon reserves were heavily reliant on the quality of soil maps (degree of ground truthing, map scale, classification type) and on algorithms describing the carbon density in soils. Consequently, estimates of national soil carbon storage from different mapping approaches give a range of 340-530 Mt carbon (1,246-1,943 MtCO_2e) and a mean of 436 ± 27 Mt carbon (1,598 ± 99 Mt CO_2e) estimated from 7 different datasets / national soil maps.

Approximately half of the total soil carbon stock is located within an area of 492,721 ha or 23.4% of the land surface of Wales, predominantly in upland areas and/or areas of permanent grassland. The remaining 76.6% of Wales is covered primarily by mineral soils with low carbon content. Typical soil carbon contents are

- Arable 47.3 tC/ha
- Grass 67.2-68.6 tC/ha
- Acid grass 90.6 tC/ha
- Natural vegetation 82.8-89.9 tC/ha

(Countryside Survey 2007 - Emmett et al, 2010)

Agricultural management systems can strongly influence soil processes such as carbon sequestration and erosion. Physical disturbance (e.g. ploughing) breaks up soil aggregates, enhancing oxygenation and allows microbial access. Drainage of and cultivation of waterlogged organic soils, leading to aeration, increases microbial decay and an increase in CO_2 emissions, but decreases in N_2O emissions. Intensive arable use of mineral soils can enhance N_2O emissions due to the increased rate of de-nitrification associated with excess fertiliser applications. Increasing the soil carbon content can only occur either by increasing carbon input, decreasing carbon emissions or by a combination of the two through improved management.

4.3.1.1 Grassland – grazing

Garnett et al. (2017) reviewed the literature on grazing options. Grazing animals potentially aid the sequestration of carbon in soils by stimulating plant growth (through grazing, and through nutrient cycling) including, importantly, root growth and by helping carbon to be moved from above ground (in the atmosphere, in vegetation) to below ground (buried manure, plant roots) where it can be less easily disturbed. Carbon cycling occurs as animals eat the vegetation, removing some of its carbon in the process. Much of the carbon they ingest is subsequently lost from the grassland system in the form of CO_2 (through microbial respiration) and methane (through enteric fermentation) and is embedded in the animal carcass or in milk. Some of the carbon is returned to the soil as dung. If this dung ends up being incorporated into the soil and the carbon is converted into more stable forms, this can cause soils to gain carbon.

Linked to this carbon cycling, nitrogen cycling also occurs between vegetation, animals, and soils. Urine and dung from animals can stimulate soil processes and plant growth. This soil nitrogen from dung and urine can also favour soil organic matter decomposition rates, resulting in less stable carbon stocks, with the carbon ultimately lost to the atmosphere as CO_2 (Neff et al, 2002). The forage that the animals do not eat will also die and decay, releasing its nitrogen back into the soil. All these re-allocations of nitrogen can boost carbon uptake, but they can also increase soil carbon release into the atmosphere.

The main value of appropriate grazing for sequestration is the effect on root growth. High root growth is needed to support high rates of net pasture growth. If plants respond to the grazing stimulus by putting down new roots, then the carbon is already underground and has a better chance of being retained there where it may eventually be converted into more stable forms.

If grazing is too heavy – that is, if the ‘offtake’ rate is higher than the capacity of the leaves to photosynthesise and create more leaves and tillers (new plant shoots) – the plants die, which means that their roots also die and, of course, grazing can no longer be supported. The sward cannot simply recover itself fast enough, meaning the plants are no longer photosynthesising and taking carbon out of the atmosphere.

Light to moderate intensity grazing is more likely to maintain soil carbon stocks and has greater potential to foster sequestration (on lands where this is possible) than continuously heavy grazing, which is usually damaging and reduces soil carbon. There needs to be just enough perturbation to stimulate plant growth, but not so much as to overwhelm it. Grasses vary, however, in their ability to withstand grazing pressure. Good grazing management at the right stocking rate helps to maintain soil carbon stocks, as compared with poor grazing practice or conversion to cropland. Where soils are degraded, there is more scope for improved grazing management to build soil carbon than where soils are already in good condition.

Conant et al. (2001) did a meta-analysis of management on grasslands including improvement by fertilisation, improved grazing management, and conversion to pasture from native and cultivated lands. Average sequestration rates were 0.30-0.35t C/ha/yr for fertilisation and improved grazing. Of the studies examining different intensities of grazing, soil C increased by an average of 0.19t C/ha for studies comparing different grazing intensities. Conant et al. (2017) updated their analysis and concluded that management practices characterised as “improved” tended to lead to increased soil C stocks, with the average across all studies of 0.47 t C/ha/yr, but this included studies where cropland was converted to grassland. Sequestration rates for grazing on established pastures (0.28 t C/ha/yr) were lower. The few studies of rotational grazing were inconclusive (Godde et al, 2020).

Smith et al. (2014) updated a meta-analysis from 2008 (Smith et al, 2008) and reported potential carbon sequestration rates of 0.22 t C/ha/yr in the cool-moist (temperate) bio-climatic region as a result of improved grassland management – a value close to that measured by Soussana et al. (2010). Soussanna et al. (2007) found evidence from multiple European grassland sites to show that soil C sequestration rates are in a large range of 0.05+/-0.30 t C/ha/yr, based on measuring soil organic C stocks. Sequestration of 0.22 t

C/ha/yr was found from measuring emission rates of carbon dioxide. European carbon balance sites (Schulze et al., 2009) indicated that on average, soils under grasslands are net C sinks of 0.57±34 t C/ha/yr showing the large variation of results. Based on Countryside Survey 2007 soil monitoring (Emmett et al. 2010), sequestration rates of 0.05t C/ha/yr were estimated for improved grassland. Henderson et al. (2015) estimated the average sequestration of 0.014 t C/ha/yr to be achievable through changes in grazing practices which maximise forage production. The large variation of sequestration rates is caused partly by the variation of management of the soils prior to measurements.

Using repeat soils samples, Bellamy et al. (2005) examined soil carbon change in topsoils in England and Wales by resampling after 20 years. They found no evidence of an increase in topsoil C in grasslands; in fact the four grassland categories (rotational grass, permanent grass, rough grazing and upland grass) showed small to moderate C losses. The losses were attributed to climate change but this was challenged by Smith et al. (2007), postulating that the observed declines may be due to changes of farming practices.

Chamberlain et al. (2010) found no evidence of an overall change in Soil Organic Carbon (SOC) across all land uses and management regimes. However, in Wales, apparent declines have occurred in C concentration in upland habitats between 2007 and 2016 (Frogbrook et al. 2009, Emmett and the GMEP team, 2017), and the drivers of this trend are under investigation. One consistent finding is that where SOC declines occur, they can occur disproportionately in regions with higher starting C stocks (Bellamy et al., 2005). SOC trends are generally derived from the top 0-30cm of soil, so their representativeness of total stock of SOC is uncertain (Buckingham et al., 2013). In some systems, the C stored below 1m equates to >50% of the C stored above 1m (Jobbagy and Jackson, 2000).
Schrumpf et al. (2011), reviewing nine studies (some of which were also included by Soussana et al., 2010), showed increases in SOC in four, decreases in two, and mixed findings or no difference in four studies. There was no consistent evidence in repeat sampling studies that grasslands are gaining in topsoil C. Schrumpf et al. (2011) emphasised the need for precise determinations to monitor soil carbon stocks and changes. . Other studies have also challenged the ability of soils to sequester carbon on a continuing basis (Poulton et al, 2018: Powlson et al, 2011: Schlesinger et al, 2019: Hopkins et al, 2009: White et al, 2018).

There is little consensus the effects of specific types of grazing management on SOC levels due to the range of grazing settings and practices (Schils et al., 2005). For example, effects of “mob grazing” on SOC are unresolved (Buckingham et al., 2013). Mob grazing involves grazing land for shorter periods of time (typically for 1 day) at higher stocking densities. Evidence for sequestration benefits of holistic, adaptive, and other variants of rotational grazing is contradictory (Nordborg 2016). One of the significant challenges in assessing the sequestration potential of grazing practices lies in the complexity of the interactions between soils, vegetation, grazing animals, and human interventions which are difficult to capture in the farming management categories usually assessed in the scientific literature.

Regenerative or mob grazing along with deep rooted herbal leys have been advocated as a way of increasing soil carbon, but the evidence is small and contradictory (Garnett et al, 2017). For ‘conventional’ rotational grazing (animals are moved between paddocks either according to calendar dates or after a certain percentage of the sward has been eaten), controlled grazing experiments which have deliberately sought to exclude all variables so as to isolate the effect of the rotation itself, have not found rotational grazing systems to offer carbon sequestration or other advantages over continuous grazing (Briske et al, 2008).

A meta-analysis of the effects of grazing on grassland soil carbon confirmed the importance of the site-specific variables on sequestration (McSherry and Ritchie (2013). No easy judgements could be made about the relationship between grazing intensity and any single factor such as rainfall or soil type. They noted that the effects of grazing management on

SOC can be large, with equally distributed gains or losses of about 5.5 t CO_2/ha/yr (1.5 t C/ha/yr), with variability over time.

In summary there is conflicting evidence about the effect of grazing on carbon sequestration although light-moderate grazing may be advantageous. Various studies conclude that sequestration rates are in the range minus 0.25 to plus 0.35 t C/ha/yr (mean 0.05 t C/ha/yr) with large errors. Much depends on the previous management of the grassland.

The indication of sequestration from soil organic carbon (SOC) levels is also conflicting with increases and decreases being reported for grassland sites. Studies of rotational grazing practices have shown no consistent trends in soil carbon levels or sequestration. There is a lack of evidence about changes in soil organic carbon levels in subsoils.

4.3.1.2 Effects of Fertilisers

It is unclear to what extent nitrogen stimulates sequestration. Lu et al. (2011) concluded from their meta-analysis that N stimulation of SOC storage primarily occurred in plant pools and less in soil pools. The small magnitude of the effect of N addition on SOC stocks was explained by the higher stimulation of above-ground biomass production than that of below ground biomass. Furthermore, the dataset gathered by Lu et al. (2011) showed that N addition stimulated soil organic matter mineralization to release carbon dioxide. This was consistent with results by Neff et al. (2002).

Manures transfer existing organic carbon to the soil pool (Chenu et al., 2019). Additions of organic materials may also improve crop primary productivity via increased nutrient availability and labile C fractions. This represents a secondary pathway by which this measure can influence net atmospheric C removal. However net sequestration depends on the added carbon becomes locked into the soil.

Fertiliser (particularly N) input has the potential to maximise yield and SOC. (Alvarez 2005). But it is important to look at overall GHG emissions, since N fertilisers and manures will increase nitrous oxide emissions. Henderson et al. (2015) found that N_2O emissions from N fertilisation exceeded soil C sequestration in all global regions. In W Europe, sequestration rates were low, ranging from 0.001 to 0.002 t CO_2 /ha/yr. Almost all (99%) of the potential sequestration was realised with low rates of N inputs and estimated sequestration rates (and amounts) changed minimally with increasing N inputs. They considered that this approach often caused losses either because the plants apportioned their growth into their above-ground biomass rather than their roots, or because the nitrogen accelerated carbon decomposition.

Fornara et al. (2015) showed from 43 years of data from a permanent grassland experiment that soil C stocks have increased across all nutrient treatments between 1972 and 2013 with soil C accumulation rates ranging between 0.35 t C/ha/y in control-unfertilised soils to 0.86 t/ha/y under highest applications of cattle liquid manures. In a subsequent assessment across 126 sites in Northern Ireland, Fornara et al. (2020) found evidence that significant changes in soil C stocks mainly occurred in the soil top 20 cm (not in deeper soils) and only between ‘extreme’ nutrient treatments (i.e. unfertilised vs. highly fertilised soils).

Ammann et al. (2007) investigated the C budget of a temperate grassland, which was newly converted from arable, for three years. They found that SOC storage was 2 tC/ha/yr higher under “intensive” management, in which manures and N fertilisers were applied, than “extensive” treatment, in which no manure or fertiliser was applied

A review to inform LULUCF inventories (Buckingham et al., 2013) found positive changes in grassland soil C stocks brought about through slurry or manure applications (0.7 to 15 t C /ha). Jones et al. (2006) reported C storage of 15.7-48.3 t C/ha following application of manure for six years. Smith et al. (2008b) reported CO_2 mitigation potential of -0.62 – 6.20 t CO_2 /ha/yr for the application of manure or biosolids in cool moist regions, although this includes both cropland and grassland. Manure application effects for SOC are sometimes presented in combination with other interventions. A meta-analysis by Conant et al. (2001) reports that fertilisation can increase SOC by 0.3 t C/ha/yr.

Optimising soil pH generally through the application of alkaline calcium or magnesium carbonates or oxides (lime) improves soil nutrient availability, increasing primary productivity and Organic Matter input to soil. There is some evidence that liming can decrease nitrous oxide emissions, but dependent on soil moisture content (Clough et al., 2004)

In summary applying fertilisers can increase carbon incorporation, but also has the potential to increase decomposition of soil carbon. Fertiliser application has the disbenefit of increasing N_2O emissions, so that overall emissions can be increased. Manure applications increase SOC particularly in the top 20cm of soil, but this accumulation does not equate to carbon sequestration unless the carbon is locked into the soil carbon pool.

4.3.1.3 Summary - grassland

Smith (2014) concluded that it is untenable that grasslands act as a perpetual carbon sink, and the most likely explanation for observed grassland carbon sinks over short periods is legacy effects of land use and land management prior to the beginning of flux measurement periods. Simply having grassland does not result is a carbon sink, but judicious management of previously poorly managed grasslands can increase the sink capacity. Given that grasslands are a large store of carbon, and that it is easier and faster for soils to lose carbon that it is for them to gain carbon, it is an important management target to maintain these stocks. Management of previously poorly managed grasslands can increase the sink capacity (though this will decrease over time).

It is important to emphasise that carbon sinks are reversible – what can be done, can be undone. Soil carbon stocks can increase through good soil management, but also be lost through bad management. Practices can change to meet economic conditions, for example moving grassland to arable.

The research literature indicates that the link between grazing management and carbon sequestration is extremely variable with time and previous management, making it difficult to set a sequestration rate which can be measured in an inventory. Rotational or mob grazing regimes show little clear evidence of promoting sequestration.

The main conclusions are that

- **Carbon stocks in soils need to be conserved.**
- **Conversion from arable to grassland leads to increasing sequestration.**
- **Opportunities to increase soil carbon occur mainly for mineral soils of Wales, particularly for improved pasture which has been previously in crop rotation with arable land.**
- **Fertilisers can increase sequestration rates but have accompanying increases in nitrous oxide emissions.**

- **Management options for the 363,000ha of semi-natural grasslands on mineral soils is more difficult as very little is done in management interventions. The land does provide opportunities for expanding woodland.**

4.3.1.4 Arable

Countryside Survey 2007 quoted soil carbon stocks in Wales to be 56.0 t C/ha for arable, and 68.1 t C/ha for improved grassland – emphasising the reduction of soil carbon stocks by emissions of carbon dioxide on converting grassland to arable (Emmett et al, 2010). The lower soil carbon stock in arable soils reflects the fact that soil disturbance accelerates top soil organic C decomposition (Conant et al., 2001).

Carbon is lost more rapidly than it is gained after a change in land use (Soussana et al., 2010). As a result of periodic tillage and re-sowing, short-duration grasslands tend to have a potential for soil C storage intermediate between crops and permanent grasslands. The C storage increases in line with less frequent ploughing (Soussana et al., 2004). SOC stock changes do not occur instantaneously but over a period of years.

The strongest negative effect on SOC reported by Guo & Gifford (2002) came from conversion of pasture to crops. Freibauer et al. (2004) reported changes of -1.0 to -1.7 t C/ ha/yr. Another study shows that converting a permanent grassland to an annual crop can decrease SOC at a rate of -0.96 t C/ha/yr over a 20-year period Soussana et al. (2004).

FAO (2017) emphasised that the decrease of carbon stocks in arable soils is affected strongly by the return of crop residues to the field, the application of organic manure and the degree of tillage intensity. FAO (2017) estimated significant and long-lasting depletion of soil organic carbon on arable land – the average carbon loss was about 0.4tC/ha/yr for conventional agriculture. Countryside Survey 2007 soil monitoring studies (Emmett et al. 2010) concluded that there was an average loss of 0.19t C/ha/yr for arable land.

Research on soil carbon in arable soils has examined various management techniques for maintaining soil carbon and reducing emissions of carbon dioxide. Powleson et al. (2014) reviewed published evidence that zero-till is beneficial for the functioning and quality of soil in many situations. The resultant soil conditions offer potential for improved crop growth and increased resilience to weather variability and likely impacts of climate change. Consequently in some environments zero-till can be regarded as a contribution to climate change adaptation. But published data on the magnitude of climate change mitigation from no-till through sequestration of organic carbon (C) in soil is much more equivocal.

Manley et al., (2005) concluded 'Our statistical analyses of more than 100 studies and some 900 estimates suggest that Zero Till (ZT) seems to sequester too little carbon at too high a cost to make this means of mitigating climate change an attractive alternative to emissions reduction'. Nevertheless, even in locations where ZT can increase soil C, it needs to be maintained. This can be difficult if ZT leads to build up of weeds which require control by cultivation or application of herbicides.

A comparison of no-till and conventional till, where soil had been sampled to at least 60 cm depth, showed no overall increase in SOC stock under no-till (Poirier et al., 2009). Larger stocks in the top 20 cm layer compared with conventional tillage were counteracted by smaller quantities in the 20–60 cm layer under no-till.

Carbon stocks are defined in terms of amount of carbon for a given weight of soil, but converting that figure to the amount of soil carbon for an area basis requires measurements of density as well as concentration. This approach requires measurement of soil bulk density in addition to SOC concentration, because bulk density is frequently altered by a change to no-till: crop residues are not mixed in the topsoil layer as occurs with ploughing or discing, so organic matter concentrates near the soil surface. This can lead to decreased soil density in the surface 5 cm compared with conventional tillage but much of the soil profile under no-till till almost invariably has increased bulk density due to the absence of disturbance.

The extra carbon under no-till is predominantly in labile forms that would be decomposed if no-till practices ceased and a farmer reverted to conventional tillage. Soil organic carbon does not continue to increase indefinitely and annual rates of accumulation decline as the soil approaches a new equilibrium, which can take from 25 to 100+ years depending on climate and soil type. (Smith, 2016) Hence, rapid rates of SOC accumulation sometimes measured in the early years after a change in management, such as a shift to no-tillage, cannot be extrapolated indefinitely. In northwest Europe, periodic ploughing is practised to control the perennial weeds and soil compaction which result from no-till practices – hence negating the benefit of no-till.

A promising option to sequester carbon in agricultural soils is the inclusion of cover crops in cropping systems. The advantage of cover crops as compared to other management practices that increase soil organic carbon (SOC) is that they neither cause a decline in yields, like extensification, nor carbon losses in other systems, like organic manure applications may do. Poeplau and Don (2015) conducted a meta-analysis to derive a carbon response function describing SOC stock changes as a function of time. Data from 139 plots at 37 different sites were compiled. In total, the cover crop treatments had a significantly higher SOC stock than the reference croplands. The SOC stock increased with the length of time since introduction of cover crops in crop rotations. The annual rate of change was 0.32 ± 0.08 t C/ha/yr at a mean soil depth of 22 cm over during the period of up to 54 years.

Another intervention that is being researched is the development of crops with larger, deeper root systems, hence increasing plant C inputs and soil C sinks (Kell 2012). Increasing root biomass and selecting for root architectures that store more C in soils has not previously been an objective for crop breeders, although most crops have sufficient ability to alter root characteristics substantially. Selection aimed at improved root adaptation to soil acidity and nutrient limitations could yield greater root C inputs as well as increased crop yields. Greater root C input is well recognised as a main reason for the higher soil C stocks maintained under perennial grasses compared to annual crops (Kell (2012). Much of the carbon and most of the measurements are restricted to the top 1 m of soil, and developing plants with deeper roots could sequester considerably more C than occurs presently.

A summary of the main changes in land use for agriculture is shown in Table 4.3

Table 4.3 Soil carbon changes and sequestrations/losses of land use changes

Land Use Change	Soil Carbon (tC/ha)	Carbon Sequestration (tC/ha/yr)
Grassland to continuous arable	From 58.7-83.8 to 43.2-56.0	- 0.96 to -2.02 -5 to -16 in first 2 years
Arable to rotational grassland	From 43.2 to 58.7	+0.1 to +0.53 *
Arable remaining arable	Range of 43.2-56.0	-0.19 to -0.4 **
Undisturbed grassland	Range of 68.1-83.8	-0.25 to + 0.35 mean value of +0.05

*Notes *Depending on rotation between grassland and arable: ** Depending on how many years continuous cropping has occurred and the initial SOC of the soil.*

In summary, for arable land in Wales, sequestering gains are limited apart from when arable is converted to grassland or woodland. Cover crops appear to be the main way to increase soil organic carbon. The current LULUCF Inventory approach would capture the impacts of land use and land use change, but not detailed land management practices.

4.3.1.5 Agro-forestry

Agroforestry is the practice of growing trees in crop or livestock systems. It can be applied to intercropped systems (e.g. alley cropping), fallow management, wind or shelter belts and grazing. In addition to C sequestration in above ground tree biomass, with ongoing transfer to the soil C pool, tree roots improve the quality and quantity of below ground C inputs, and recover nutrients and moisture from lower soil horizons (Cardinael et al, 2017). In the silvo-arable systems, the mean organic carbon stock accumulation rate in the soil was 0.24 (0.09–0.46) tC/ha/yr at a depth of 30 cm and 0.65 (0.004–1.85) t C/ha/yr in the tree biomass. Arable soils have lower soil carbon levels than grassland soils making below ground sequestration from trees to be more significant.

Meta-analyses have shown that tree planting on pasture, at least in the short term, can lead to declines in soil organic carbon (Shi et al., 2013). Such losses tend to be exacerbated in areas of high rainfall.

Upson et al. (2016) showed for experimental plots established 14 years before, the above ground storage of the woodland trees was 35.9 t C/ha , equivalent to an annual increment of 2.56 t C/ha/yr . Hence although woodland planting increased carbon sequestration in the above ground biomass during the first 14 years, the SOC losses in the 0–10 cm layer would alone offset 37% of the above ground gain. There was also a loss of SOC stock in the surface layer (0-10 cm) of the silvopastoral trees of 6.1 t C/ha, equivalent to a mean loss of 0.44 t C/ha/yr over the 14 years.

Over the 14 year time span, the above ground carbon storage of the silvopastoral trees was higher than for woodland, equivalent to 99.4 t C/ha (7.1 t C/ha/yr) when expressed in relation to the tree spacing of the silvopasture. Hence in this case the SOC losses in the surface soil layer would reduce the net carbon gain within the area of the silvopastoral trees by 6%. These results indicate that the silvo-pastoral system could store more carbon than equivalent areas of trees and pasture in separate blocks.

In summary, the evidence for carbon sequestration benefits from agroforestry appears significant for silvo-arable systems, but less so for silvopastoral systems which already have large carbon stocks in the soil. It is important to take account of soil C

loss during the establishment of trees.

4.3.1.6 Hedgerows

There is scope for expanding the length and width of hedgerows to sequester carbon in the biomass and soils. Recent studies in the UK suggest that hedgerow biomass could sequester 0.4–1.25 tCO_2 /ha/year, depending on hedgerow type, dimensions and vegetation density (Falloon et al, 2010). However this estimate was based on converting arable land with a lower carbon content than pasture to hedges. Black et al. (2014) estimated biomass sequestration rates of hedgerows in Ireland to be 0.66– 3.3 t CO_2 /ha/year). Based on their assumption of an average sequestration rate of 0.7 t CO_2 /ha/year, hedgerow trees in the Irish landscape could represent a sink of 200,000 t CO_2 /year.

Axe et al. (2017) studied the carbon stocks in biomass and soils in lowland hedges in England. They estimated that the Above Ground Biomass (AGB) for 3.5 m tall hedges was 42.0 ± 3.78 t C/ha. In the soil, the C stocks were 38.2 ± 3.66 t C/ha. When trimmed to 2.7 m high, and subsequently 1.9 m high, AGB C stocks were reduced to 40.6 ± 4.47 t C/ha and 32.2 ± 2.76 t C/ha respectively.

Countryside Survey 2007 showed that the length of hedges in Wales in 1984 was 71,800km, but had decreased to 54,000km by 2007. Based on the results by Axe et al. (2017), reinstatement of 17,800 km of hedges with 2m wide hedges would increase carbon stocks by 136,000t C below ground and by 114,500-149,520 t C AGB for hedges between 1.9m and 3.5m high. Doubling the widths to 4m would double the carbon stocks. **However it should be noted that the soil carbon stocks in pasture is already high, implying that the gains in soil carbon may be smaller than found by Axe et al.**

Doubling the width of **the existing 54,000km of hedges** from about 2m to 4m may sequester an additional 413,000t C below ground and an additional 347,000-454,000 t C AGB in the biomass for hedges 1.9m-3.5m high. In practice farmers would wish to trim hedges to keep them shorter than 3.5m. Trimming and regrowth would maintain some carbon sequestration in the above ground biomass.

Based on the Irish estimate of average sequestration rate of 0.7 t CO_2 /ha/year (Black et al. (2014), for the 54,000km of hedges in Wales at 2m width, sequestration is about 7,560 t CO_2 e/year, increasing to 10,050 t CO_2 e/year if hedges are increased to 71,800km. The total length of woody linear features in Wales in 2007 was 106,000km, **which may be sequestering about 15,000 t CO_2 e/year. The estimated sequestration rates have large errors from the study by Black et al. (2014), and actual sequestration rates could be up to 5 times higher – up to 75,000 t CO_2 e/year.**

These estimates of hedgerow[1] C stocks have identified the need for a more comprehensive biomass inventory of hedgerows to strengthen the national carbon accounting of agro-ecosystems in the UK. Black et al. (2014) conducted a (Light Detecting and Ranging) LIDAR technique to develop a hedgerow classification and sampling system to assess biomass and carbon sequestration. Direct modelling of LIDAR metrics was used to accurately estimate hedgerow and non-forest woodland biomass.

In summary, there are opportunities to sequester carbon by up to 75,000t CO_2e/yr by increasing the length of hedges in Wales – possibly up to levels in 1984 and before.

1 https://www.sciencedirect.com/topics/earth-and-planetary-sciences/hedgerow

Increasing the width and height of hedges would increase the carbon in biomass, but could have disadvantages in achieving effective trimming unless hedges are allowed to grow and are then laid – an expensive and labour intensive task. Soil carbon is also increased around the perimeters of arable fields. Around grassland fields the existing high soil carbon levels of grasslands may lead to smaller benefits. Measurement methods to assess hedges have been demonstrated and need to be implemented for inclusion in the national GHG inventories.

4.3.1.7 Bioenergy crops

McCalmont et al. (2017a) reviewed soil carbon changes from miscanthus based on 14 comparison sites. For arable land converted to miscanthus, 11 sites showed overall increases in SOC over their total sample depths with suggested accumulation rates ranging from 0.42 to 3.8 tC/ha/yr. The grassland to miscanthus comparisons showed three increases, three decreases, and one no change in soil C stocks. The findings were complicated by the miscanthus being planted on former arable, arable/fallow, or grassland, whereas all comparisons to arable were planted on former arable land. The range of gains and losses was relatively small, 1 to 0.94 tC/ha/yr with only the increase of 0.94 tC/ha/yr shown to be significant (Hansen et al., 2004).

McCalmont et al. (2017b) have studied the conversion from semi-improved grassland to miscanthus for biomass production. Using a simple mass balance approach, above-ground and below-ground biomass production were combined with CO_2 fluxes to estimate short-term carbon changes across individual years. Years one and two both ended with the site as a net source of carbon following cultivation disturbances. The site became a cumulative net sink for carbon in the third growing season and remained so for the rest of that year. Carbon gains were primarily found in biomass pools, and SOC losses were limited to years one and two. Year three saw recoupment of soil carbon at 0.74 tC/ ha/yr with a further estimate of 0.78 t C/ha/yr incorporated through litter inputs over the 3 years, suggesting a net loss of SOC at 3.7 tC/ha/yr from an initial concentration of 78.61 +/-3.28 t C/ha in the 0-30cm deep layer. Assuming this sequestration rate as a minimum would suggest replacement of cultivation losses of SOC by year 8 of a potential 15-year to 20-year crop.

In summary, bioenergy crops have overall benefits but must take account of soil C loss during the establishment of plants. Overall, bioenergy crops are important for sequestering carbon and also the burning of bioenergy crops is recorded as a zero emission compared with combustion of fossil fuels for energy.

4.3.1.8 Biochar

Biochar addition to soil is advocated for the long-term stabilisation of carbon. Biochar (charcoal) is produced from the pyrolysis of organic materials. If buried in soil it can act as a long-term soil carbon store (>500 years). Although biochar could be produced on farms, the volumes of biomass available are probably insufficient to meet demand, unless there was collective effort between farmers, e.g. for poultry farm manure. Biochar could be produced from large volume waste materials (e.g. green waste, biosolids, forest residues) and subsequently ploughed into agricultural fields. Typical application rates of biochar to arable fields range from 8-100 t/ha. After a few years of biochar applications this would effectively double the amount of carbon stored in soil organic matter in the topsoil.

There are also potential GHG emission benefits in the combustion of biomass waste if it recovers the energy for replacing fossil fuels and reduces methane emissions from landfilled material. The combustion of biomass waste reduces the natural degradation which would lead to methane emissions (Wolff et al, 2010). Before adoption of biochar, full LCA is required to quantify emissions associated with biochar production, transport and application (Quinn et al, 2020).

Biochar has been shown to have varying effects on soil qualities such as fertility and water holding capacity (Jeffrey et al, 2011). They found from a statistical meta-analysis of global studies that there was an overall small, but statistically significant benefit of biochar application to soils on crop productivity, with a grand mean increase of 10%. However, the mean results for each analysis performed within the meta-analysis covered a wide range of crop productivity (from −28% to 39%). The greatest (positive) effects were seen in tropical soils. Two of the main mechanisms for yield increase may be a liming effect and an improved water holding capacity of the soil, along with improved crop nutrient availability. The greatest positive result was seen in biochar applications at a rate of 100 t/ha (39%). Of the biochar feedstocks considered and in relation to crop productivity, poultry litter showed the strongest (significant) positive effect (28%).

For soils in temperate areas, Jeffrey et al, (2017) found biochar additions led to significantly lower crop yields, averaging approximately 3% at a median biochar application rate of 30 t/ha. For field experiments, there was no effect of biochar on crop yield. The conclusion was that to a large extent, the yield-stimulating property of biochar derives from raising the pH of soils by the soil liming effect.

In summary, biochar may contribute to reduced fertiliser and liming costs, if applied periodically at low application rates. The best methods of incorporation of Biochar into grassland have not been identified. It is likely that a one-off application could occur during reseeding where the Biochar is incorporated into the topsoil. Biochar application to soil needs to be adopted with caution since it is not reversible, making it difficult to reduce any negative effects.

4.4 Peatlands

This section is based on the report by Evans et al. (2017). It describes the area of peat, the emissions from peat, and mitigation measures, with scenarios of emission reduction from the mitigation measures. As well as modifying greenhouse gas emissions, there is evidence that peatlands absorb atmospheric ammonia emissions (Daniels et al, 2012).

4.4.1 Peat area

The extent of peat areas for Wales was developed in a previous project for the Welsh Government (Evans et al., 2014). This was mapped from a combination of peat areas recorded by 1:50,000 BGS superficial geology dataset, and a range of survey data held by Natural Resources Wales (NRW), comprising the Lowland Peat Survey, peat-associated habitat categories recorded in the Phase I Habitats Survey, and soil surveys undertaken by the former Forestry Commission Wales. The total peat area in 1990 was 90,050 ha. The land use for peat is summarised in Table 4.3.

Table 4.4 Peat areas from Phase 1 Habitat Survey 1990, and Emission Factors

Peat Category	Area (hectares)	Total Emission Factor (tCO2 e/ha)
Forest drained	9520	9.91
Cropland drained	102	38.98
Drained eroded modified bog	19	4.85
Undrained eroded modified bog	206	3.55
Drained heather dominated modified bog	1588	3.40
Undrained heather dominated modified bog	6237	2.08
Drained grass dominated modified bog	1588	3.40
Undrained grass dominated modified bog	29000	2.08
Extensive grassland	8993	19.02
Intensive grassland	6577	29.89
Near natural bog	23548	0.01
Near natural fen	2674	-0.61
Total	**90050**	**-**

The emission factors in Table 4.4 are from the report by Evans et al. (2017). Forest, cropland and grassland on peats have very high emission factors, making them important in emission reduction plans.

Between 1990 and 2013, 5563 ha was rewetted, mainly from extensive grassland, grass dominated modified bog and heather dominated modified bog. During this period 76 ha of peatland was afforested, and 331 ha was deforested.

4.4.2 GHG emissions from peat

Although around 30% of Wales' peat area is near natural, the remainder is in a modified state, ranging from relatively minor changes to vegetation cover and hydrology, through to the complete replacement of wetland vegetation by arable and horticultural crops, agricultural grasses and non-native conifers, with accompanying deep drainage. This has led to drying of the peat, loss of peat-forming species and erosion, converting these areas into net GHG sources.

The overall GHG emissions need to take account of rates of CO_2 sequestration by peat balanced by emissions of methane and nitrous oxide. Emissions of CH_4 are mainly associated with undrained areas, with the largest emissions coming from near-natural and modified bogs. Each of these categories contributes 8% to total UK GHG emissions from peatlands, but it is important to note that both fluxes can be considered predominantly natural, and that CH_4 emissions from near-natural bogs are cancelled out in CO_2-equivalent terms by CO_2 uptake. For drained land-uses on peat, including cropland, forestry and agricultural grassland, the majority of CH_4 emissions are derived from drainage ditches.

Nitrous oxide emissions occur as a result of oxidation of ammonium ions to nitrite and subsequently to nitrate, and anaerobic denitrification of nitrate. All N_2O emissions estimates were considered to have a high uncertainty. This is a particular issue for cropland and intensive grassland on peats, due to local variations in fertiliser and drainage regimes, as well as the intrinsic spatial and temporal heterogeneity of N_2O emissions. Cropland N_2O emissions from peat account for 8% of UK GHG emissions from peatlands, and N_2O emissions from intensive grassland on peat contribute a further 3%.

In the current GHG inventory, peats in areas that have been recently converted to forest act as a small emission source, but peat under mature forest ('forest remaining forest') is predicted to act as a net GHG sink because as the forest matures, litter inputs to soil increase and the model suggests that these eventually outweigh lost from soil due to oxidation. The net GHG sink in peat under mature forests modelled by CARBINE is sufficient to make peatlands in Wales approximately GHG-neutral.

This area of natural bog and fen is believed to be continuing to act as a significant net sink for CO_2, but this is counterbalanced by similar emissions of methane when its greater 100-year Global Warming Potential is taken into account. This makes near-natural peatlands close to carbon neutral. For natural fens, CO_2 uptake exceeds CH_4 emission on a CO_2-equivalent basis making them a very small net GHG sink.

Although the emissions per unit area of modified peatland are relatively low, their great extent makes them significant contributors to overall peatland GHG emissions. As a result, peatlands in the UK have transitioned from modest historical net GHG sinks (an estimated pre-anthropogenic sink in the region of 0.25 Mt CO_2e/yr) into large emission sources (exceeding 23 Mt CO_2e/yr).

It is important to emphasise the uncertainties in the emission estimates. 24% of the current estimate of total emissions from the UK's peatlands can be estimated with fairly high confidence, and a further 15% with moderate confidence, but the majority of the estimate (61%) can only be estimated with a low level of confidence. The GHG Inventory is based on Land Cover Map 2015 with collation of data from other sources, resulting in likely inconsistencies in classification and year of data collection. For such a mapping approach to be of value for peat activity mapping, a sufficient level of classification detail will be needed to differentiate different condition categories, particularly the relatively subtle changes in semi-natural species composition that influence emissions from modified bogs and fens. Mapping of the extent of drainage also presents particular difficulties.

4.4.3 Mitigation options

The Emission Factors for peats linked to peat category are shown in Table 4.4. These take account of direct CO_2 emissions, CO_2 emissions from Dissolved organic Carbon and from Particulate Organic Carbon, direct CH_4 emissions, indirect CH_4 emissions from ditches, direct N_2O emissions, and indirect N_2O emissions from drainage waters. Focusing on the restoration of the peat categories with the highest Emissions Factors would be most beneficial in reducing emissions. **Taking account of area, restoration of extensive and intensive grassland and forestry on peatlands would provide the most benefit.**

The main mitigation method is to re-wet existing peats by raising the water table, for example by blocking drainage ditches (often referred to as 'grips'), to restore the function of the peat as a net sink of CO_2 and a semi-permanent carbon store.

The evidence base for quantifying GHG gas emissions and carbon storage in peat after re-wetting is poor. In particular there is a need to determine the long-term rates of CH_4 flux from both virgin, drained and rewetted peats in Wales.

Evans et al. (2021) have investigated the relationship between emissions of CO_2 and CH_4 with water table, based on a range of sites across the UK and Ireland. The data showed maximum emissions of around 0.2 t C/ha/yr from CH_4 when the Water Table Depth (WTD) was close to zero (that is, at the peat surface), and predominantly near-zero emissions when WTD > 30 cm indicative of complete oxidation of CH_4 in the aerobic peat layer. A 10-cm reduction in WTD within the range of observations on average, decreased CO_2 emissions by around 3t CO_2e/ha/yr. Reducing WTD from 30 cm to 0 cm would increase CH_4 emissions by approximately 7t CO_2 e/ha/yr. Combining the two gases, maintaining a WTD of 5 cm to 13 cm ensures that the cooling effect of CO_2 sequestration exceeds the warming impact of CH_4 emissions.

Evans et al. (2017) have modelled scenarios for Peatland Management to 2050, based on a set of mitigation measures applied to differing areas of peat. Table 4.5 summarises the forecast changes in emissions.

Table 4.5. Scenarios for Emissions reduction from Peatlands in Wales

Scenario	Mitigation	Change of Annual GHG emissions (kt CO2e) in 2050
High – further degradation of existing peat	25% of near natural bog becomes modified, forest and extensive grassland	+54
Baseline – policies in 2009	Current areas for each peat condition unchanged	0
Central – Business as Usual	No restoration of lowland or upland peat	0
Low – policy aspirations post 2021	25% restoration of degraded lowland peat 50% restoration of degraded upland peat	-339
Stretched – higher aspirations	50% restoration of degraded lowland peat 75% restoration of degraded upland peats 50% restoration of forest area planted on peat since 1980	-685

In summary, the modelled results show that the focus of peatland management should be to reduce current high emissions. It is unlikely that so-called 'negative emissions' from peat formation will be able to offset emissions from other sectors. Widespread and ongoing peat restoration has contributed to a reduction in total emissions, but to date the majority of restoration has taken place within modified upland bogs, which produce modest emissions sources per unit area, rather than categories with higher emission factors per unit area such as grassland and plantation forestry in lowland areas.

Addressing continued emissions from these areas could provide a high degree of emission abatement but would face significant logistical and socioeconomic barriers particularly for lowland peat restoration. In the meantime, the continued restoration of modified upland bogs, notably higher-emitting categories such as actively eroding areas may represent more tractable options for emissions reduction.

4.5 Scenarios

Mitigation projection for land use in Wales have been estimated by Thomson et al. (2020). Two key scenarios are (1) Central – essentially business as usual taking account of current policies, and (2) Stretch. The assumptions of the Stretch scenario were

4.5.1.1 Forest

Conifers = 16% of future planting – based on current practiceIncrease of forest from 346,000ha to 440,000ha by 2050, at 4000ha/year over 2018-2040, and 1000ha/year for the period 2040-2050Note: Forest Research used a baseline figure of 346,000ha but there is a range of estimates.

4.5.1.2 Grassland

Extend hedges by 40% at a linear rate until 2050.

4.5.1.3 Cropland

Levels of activity remain constant but assumed Grass to Crop average rotation rate of 6.2kha/y Emissions are based on assumed manure and fertiliser additions, tillage methods and crop residues removal.

4.5.1.4 Settlement

Conversion to Settlement to meet housing demand according to the evolution of household projections published by ONS in 2018. Initial change of rate smoothed over 15 years; land from deforestation first, then split between grassland/cropland based on their relative share over 2008-2017.

4.5.1.5 HWP (Harvested Wood Products)

Allocation based on the average allocations for the period 2008-2017

The effect of the two scenarios on emissions is shown in Table 4.5. For Forestry initially, the GHG balance is dominated by net CO_2 emissions from loss of soil carbon stocks (2.0 tCO_2 e/ha/yr), which occur as a result of site preparation and the time involved in the transition occurring between the loss of pre-existing vegetation on the site and the full establishment of the trees. Consequently the carbon sequestration is less in the early stages for the higher planting rates, and the main benefits occur from 2040 onwards, as new trees grow rapidly.

Table 4.5 – Wales Annual Emissions Projections – Stretch scenario compared with Central scenario (equivalent to BAU)

Land Use	Annual emissions – Kt CO2 e								
	2020	2030		2040		2050		2060	
		Central	Stretch	Central	Stretch	Central	Stretch	Central	Stretch
Forest	-1146	-1328	-1236	-1609	-1618	-1628	-2008	-1313	-2039
Cropland	946	1461	1462	1763	1759	1940	1935	1167	1167
Grassland	-643	-834	-908	-1083	-1157	-1236	-1312	-975	-1051
Wetland	0	0	0	0	0	0	0	0	0
Settlement	653	497	437	377	304	289	218	163	123
Harvested wood (HWP)	-318	-147	-178	-33	-55	-30	-51	-105	-133
Total	**-483**	**-322**	**-391**	**-552**	**-731**	**-634**	**-1187**	**-1050**	**-1920**

Note. Wetland emissions are estimated for peat extraction only. This is zero in Wales.

Combined with reduction of emissions from the peat management scenarios (Table 4.4) by 685 kt CO_2e, the stretch scenario could reduce emissions by 1872 kt CO_2e/yr by 2050 compared with the Central scenario.
Recent Welsh Government proposals to expand woodland by 180,000ha by 2050 could lead to additional LULUCF emission reductions of up to 304 kt CO_2e in 2050 if the planting rate was in line with the planting profile for the 100,000ha expansion. In total, the emission reduction is forecast to be 2176 kt CO_2 e/year in 2050. In the period up to 2050 carbon

sequestration would be maximised from conifer woodlands, whereas broadleaf woodland increases its carbon sequestration in the period 2050-2100 and beyond.

4.6 Summary

The main conclusions are that:

- **Carbon stocks in soils need to be conserved. The capacity of soils to sequester carbon is limited to arable soils and other soils that have been degraded – for example brownfield sites which have been reclaimed. For permanent grassland, the soil carbon levels in Wales are already large. There is a limit to the capacity to increase carbon stocks, unless new deep rooting crops are adopted.**
- **The focus of peatland management should be to reduce current high emissions. Widespread and ongoing peat restoration has been mainly taken place within modified upland bogs, which produce modest emissions sources per unit area, whereas restoration of grassland and plantation forestry in lowland areas would achieve more substantial reductions of emissions.**
- **Livestock do not sequester carbon. They contribute to accumulation in some compartments (reservoirs) along the cycle: in soils, or in plant and animal biomass. Not all organic matter that enters the soil is converted into long term, stable soil carbon, since it leaves the system within a short period when ingested and respired by soil organisms.**
- **Grazing management shows variable effects on carbon sequestration. Studies of rotational or mob grazing regimes show little evidence of promoting sequestration.**
- **For arable soils, studies show that no-till led to no overall increase in soil organic carbon compared with conventional till, where soil had been sampled to at least 40 cm depth. Larger stocks in the top 20 cm compared with conventional tillage were counteracted by smaller quantities in the 20–40 cm layer under no-till.**
- **Cover crops used for cropland have a significantly higher SOC stock than reference croplands. The SOC stock increases at a rate of 0.32 ± 0.08 t C/ha/yr at a mean soil depth of 22 cm.**
- **Carbon sequestration benefits from agroforestry and bioenergy crops can be significant. Benefits must take account of soil C loss during the establishment of crops.**
- **Biochar is valuable in locking carbon into soils but its incorporation into grassland is difficult without disturbing the soil and potentially reducing the existing soil carbon. There is negligible effect on fertility for temperate soils.**
- **Fertilisers can increase sequestration rates but have accompanying increases in nitrous oxide emissions.**
- **Opportunities to increase soil carbon occur mainly for mineral soils of Wales under improved pasture. Management options for the 363,000ha of semi-natural grasslands on mineral soils are more difficult as very little is done in terms of management interventions. The land does provide opportunities for expanding woodland.**
- **Expansion of woodland is the most effective way to increase carbon sequestration. The stretch scenario for the LULUCF sector forecasts the potential to reduce GHG emissions by 1872 kt CO_2e/yr in 2050 if woodland is expanded by 100,000 ha, combined with substantial peatland conservation. Expansion of woodland by 180,000 ha by 2050 would reduce emissions by a further 304 kt CO_2e/yr in 2050 - a reduction of 2176 kt CO_2e/yr**

- **These forecasts are based on future planting of 16% conifers and 84% broadleaf. In the period to 2050, increasing the proportion of conifers would increase sequestration rates.**

5 Manures and fertilisers

5.1 Introduction

Manures and fertilisers are the main source of nitrous oxide representing 29.6% of total agriculture emissions for Wales in 2018. The handling and storage of manures is also the source of methane emissions – accounting for 7.9% of total agriculture emissions in 2018.

5.1.1 Nitrous oxide

The main sources of nitrous oxide are direct emissions (Butterbach et al, 2013).

- soils after application of inorganic and organic forms of nitrogen (N) as synthetic fertilisers, crop residues, manures or composts.
- nitrogen-fixing crops, such as clover and legumes which introduce large quantities of N into soils
- animal housing and manure storage,
- urine and faeces deposited on soils during grazing.

In addition to the direct sources of N_2O are also indirect sources that include

- nitrogen deposited onto land surfaces following ammonia and NOx volatilization, and
- nitrate leached from agricultural land in drainage water which, on passing into aquifers or into surface waters and their sediments, can be partially transformed to N_2O.

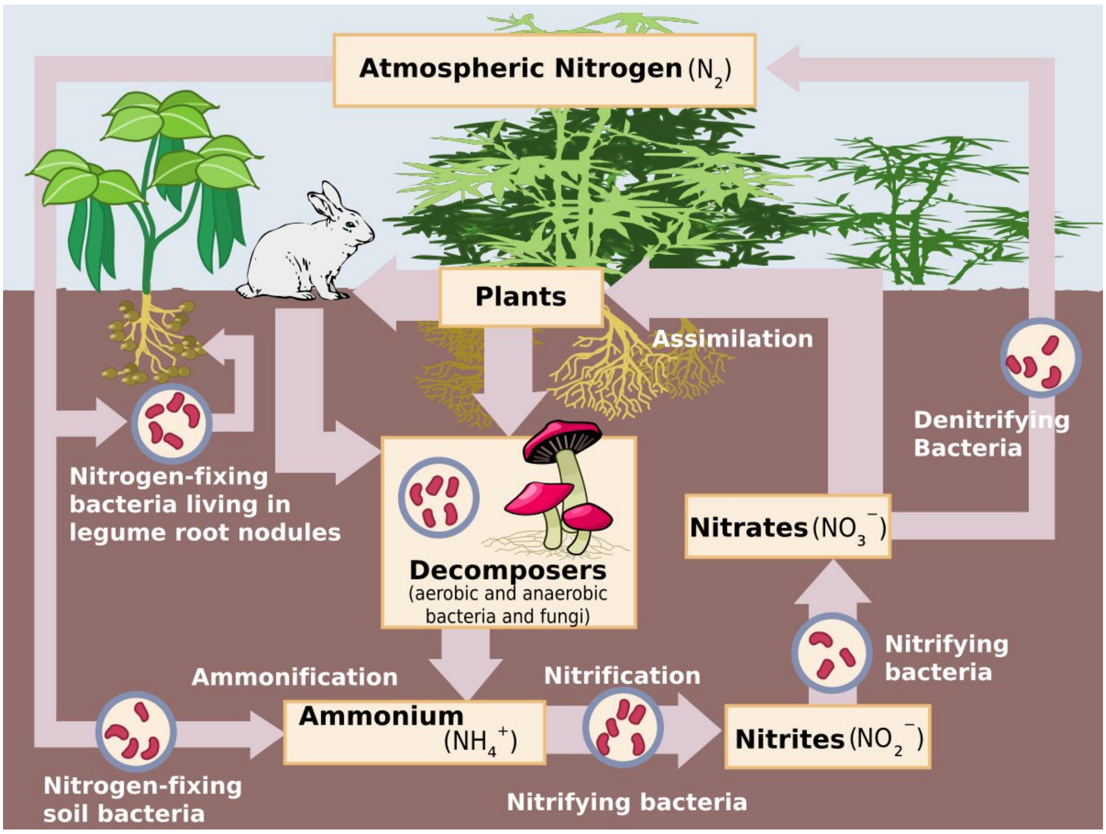

Figure 5.1 The Nitrogen Cycle between the atmosphere, plants and soils.
*(Cicle_del_nitrogen_de.svg: *Cicle_del_nitrogen_ca.svg: Johann Dréo (User:Nojhan), traduction de Joanjoc d'après Image:Cycle azote fr.svg.derivative work: Burkhard (talk)Nitrogen_Cycle.jpg: Environmental Protection Agencyderivative work: Raeky, CC BY-SA 3.0 <https://creativecommons.org/licenses/by-sa/3.0>, via Wikimedia Commons)*

The nitrogen cycle is shown in Figure 5.1. Nitrous oxide is produced from the combined nitrification-denitrification process that uses the nitrogen compounds in manure. The majority of nitrogen in manure is in ammonium (NH_4^+) form. The processes are

- Nitrification occurs aerobically by microbial oxidation of ammonium ions to nitrite with the release of nitrous oxide. Further oxidation of nitrite to nitrate occurs.
- Denitrification occurs anaerobically, and converts the nitrate to N_2O and nitrogen

In addition, another source of N_2O is the chemical reduction of nitrite ions by compounds such as amines present in soil organic matter, and by inorganic ions (Fe^{2+}, Cu^{2+}), particularly in subsoils. It is less important as a source of N_2O from agricultural soils.

Mitigation measures for nitrous oxide need to take account of the pollution swapping with ammonia. Measures to reduce nitrous oxide can increase ammonia and vice versa. In some cases, the interventions can be complementary. It is important to recognise that ammonia emissions can be deposited on land to be converted into nitrous oxide (2.6% of total N_2O emissions). Also leached N compounds end up in water ways and indirectly cause nitrous oxide emissions (5.5% of total N_2O emissions), which are counted in the GHG inventory.

5.1.2 Methane

Livestock manure is primarily composed of organic material and water. Under anaerobic conditions, the organic material is decomposed by bacteria. The end products of anaerobic decomposition are methane, carbon dioxide, and stabilised organic material.

The methane production potential of manure depends on the specific composition of the manure, which in turn depends on the composition and digestibility of the animal diet (see Chapter 6). Optimal conditions for methane production include an anaerobic, water-based environment, a high level of nutrients for bacterial growth, and warm conditions.

The sources of GHGs emissions and the options for mitigating them are reviewed in the next sections. The key determinant is animal numbers – both ruminants and other animals – particularly pigs and poultry. Any methods described in Chapter 6 for reducing animal numbers whilst maintaining productivity are important. But it must be recognised that manures are key to maintaining fertility of soils in particular their role of substituting for manufactured N fertilisers.

5.2 Animal management

5.2.1 Diet

Variation in dietary N intake affects excretion of urinary N, which is much more vulnerable to losses than is faecal N (Dijksta et al, 2013). Urinary N excretion, in particular that of urea N, is decreased by the reduction of dietary N intake or an increase in the supply of energy to the rumen microorganisms and to the host animal itself. A meta-analysis of the effects of dietary protein concentration and degradability on milk protein yield, and efficiency of utilisation of dietary N for milk protein synthesis, concluded that the Crude Protein (CP) concentration of the diet is the most important dietary factor influencing milk N efficiency, and that reducing dietary CP is the most significant means to increase efficiency of dietary protein utilisation (Huhtanen and Hristov, 2009). In some cases excess crude protein is fed as Rumen Undegraded Protein which is important for high yielding dairy cows. The practical effect of reducing crude protein was shown for dairy cows fed a 14% CP diet. The cows excreted 45%

more urinary N for a 19% CP diet compared with excretion from cows fed a 14% CP diet (Misselbrook et al., 2005a). There was also a small decrease in faeces N at the lower CP diet.

Most of the N in urine (from 50% to well over 90%) is present in the form of urea. Following its deposition on pastures or in animal sheds, microorganisms in soil and waters transform urinary N components into ammonium (NH_4^+), and thereafter into NO_3^- and ultimately into N_2 accompanied with the release of N_2O.

For grazing animals, nitrous oxide emissions are substantial. Urine patches from cattle on pastures represent substantial, highly localised additions of N of up to 1000 kg N/ha. Cardenas et al. (2016) showed variation of emissions with season - much higher in spring than in the autumn. Chadwick et al. (2019) showed that the average urine and dung N_2O Emission Factors were 0.69% and 0.19%, respectively.

Feeding high-sugar grass varieties with high Water Soluble Content (WSC) to pastured cattle has been proposed as a means to reduce N excretion in urine and faeces (Miller et al, 2001). The principle of the high-sugar grass mitigation strategy is that for a high-producing animal on pasture, rumen microbes lack adequate supply of energy (carbon) for microbial growth relative to the large amount of available protein N in the diet. This nutrient mismatch results in large amounts of N being lost from the rumen as ammonia instead of being incorporated into microbial protein. Most of the N lost is via excretion as urea in the urine. Increasing the sugar content of grasses could rectify the imbalance of carbon and N being delivered to the rumen microbes, thereby making N utilisation by the microbes more efficient, decreasing N loss from the animal and increasing the supply of CP to the ruminant.

In practical studies, the greatest benefits in terms of N utilisation ratio and urine N levels were seen when the WSC of grass increased at the expense of Crude Protein. Foskolos and Moorby, (2017) found nitrogen excretion in urine was reduced by 26% although milk yields were not increased. But although high-WSC grasses may increase the efficiency of N utilisation and milk yield, Ellis et al. (2011) concluded that the benefits depend on the diet under consideration.

The difficulty in quantifying the mitigation from diet change is that although specific experiments have shown quantified emission reductions, current protein evaluation systems are unable to predict marginal urinary N output in response to changes in diet composition (Dijksta et al, 2013). This makes it difficult to standardise efficient factors linked to CP.

In summary it is important to obtain and use accurate feed formulations to deliver the most appropriate balance between energy and protein nutrition. Protein not used by the animal is excreted in urine and faeces, becoming the source of nitrous oxide in the soil. There is evidence that grazing high sugar grasses are effective in reducing nitrogen compounds in excreted urine and faeces.

5.2.2 Animal housing

Abatement techniques for livestock housing focus on limiting the factors giving rise to the emissions, most often NH_3 and to a lesser extent, CH_4 and N_2O (Loyon et, 2016). Emissions arise from the manure in the building as well as enteric fermentation from ruminants. Ammonia comes principally from the urine which contains the majority of the volatile N excreted, whilst the dung is more likely to be the source of CH_4 production, and to some extent, N_2O (Chadwick et al., 2011). The generation of these three gases is influenced by the floor type, the ventilation system, the building temperature and the manure characteristics (Chadwick et al., 2011).

Slurry systems create the ideal anaerobic environment for methane production. The slurry/faeces/urine remains in a predominantly anaerobic state with little opportunity for the NH_4^+ to be nitrified. As a result, little or no N_2O emissions are likely to occur from such systems (Chadwick et al. 2011). N_2O from slurry based systems is very small (Thormans et al, 2007).

In contrast, in housing systems using straw and other bedding materials, cattle foul the litter in the surface layer with fresh urine and faeces. Depending on the type and amount of litter added, oxygen diffuses into the porous surface layer, and fermentation processes increase the temperature and induce an upward current of air containing NH_3, N_2O and N_2 (Rom and Henriksen, 2000). The N_2O is likely to be formed at the interface of oxidised and reduced conditions, where nitrification and denitrification processes may occur side by side. Extremely high N_2O emissions of 10% for deep litter systems have been reported (Oenema et al., 2005).

Emissions from housing are reduced by the frequent removal, and/or drying of manure), by maintaining good conditions in the building (adequate ventilation and temperature) and the use of air filters for pig and poultry houses. Air filtering is not applicable for cattle sheds because these are usually naturally ventilated, in contrast to pig and poultry systems that are often closed with forced ventilation. Biofilters can be used to treat the emissions, but these are mainly for absorbing methane (see Chapter 6) and are not applicable to N_2O.

In collecting yards, there are few studies, but emissions of nitrous oxide and methane are considered to be small.

In summary, solid manures emit nitrous oxide and some methane, whereas liquid manures are mainly a source of methane. Switching to liquid systems is useful if there are ways to capture the methane for biofiltration or for combustion. This could be a mitigation option for beef farming which characteristically uses straw or other solids bedding. However it would require a major investment in storage lagoons because beef farms tend to rely on solid manure systems typically comprising manure dumps in yards or fields.

5.3 Manure storage and handling

5.3.1 Slurry Covers

For slurry lagoons and tanks, the presence of a natural crust or layer of floating porous material will greatly increase the diffusion distance, slowing the rate of transfer of ammonia to the air (Misselbrook et al, 2005b). This leads to the retention of more ammonium in the slurry solution. In practice the crusting is unpredictable as a mitigation measure – depending on rainfall, temperature, rates of filling the store, straw content and agitation of the slurry. High rainfall in Wales is likely to make crusting difficult.

Kupper et al. (2020) reported on a comprehensive literature review of the effect of slurry covers used for cattle and pig slurry. For N_2O, an increase in emissions is observed in many cases. But reduced emissions occur as well. However, the number of records providing emission changes from slurry storage due to store covers is sparse and the effects are statistically insignificant. CH_4 emissions were lower by approximately 10% to 60% for impermeable covers (lid and plastic film), plastic tiles and vegetable oil compared to uncovered storage. For plastic fabrics, expanded polystyrene and peat, the emissions were higher by 2% to 33%. The other cover types (expanded clay, straw and organic materials such as corn stalks or wood chips) show both increases and reductions in CH_4 emissions

In summary, CH_4 emissions were lower by approximately 10% to 60% for impermeable covers (lid and plastic film), plastic tiles and vegetable oil compared to uncovered storage, but for other permeable materials, there were both increases and reductions in CH_4 emissions. For N_2O, both increases and decreases in emissions were found but the number of records is sparse and the effects are statistically insignificant.

Recently, gas sealed or partially sealed slurry stores have been developed, linked to equipment to capture the methane for energy. This may prove useful, depending on the costs of the stores and equipment[2] . Slurry bags have also been developed to reduce emissions[3,3] .

5.3.2 Solid manure heaps

Manure heaps are a source of N_2O and methane. When fresh manure is added daily on top of a heap, there is a constant source of fresh urea, but there is little opportunity for nitrifiers to develop in the anaerobic environment. Reported emissions are in the range of 0.1% to 0.5% of the N in the manure, but these estimates are based on few measurements (Oenema et al., 2005). Chadwick (2005) showed that covering and compacting cattle manure heaps has the potential to markedly reduce N_2O and NH_3 emissions by 30% and 90% respectively.

5.3.3 Solid-liquid separation

This can be achieved by a reduction in slurry dry matter and easily degradable organic matter content. Chadwick et al, (2011) commented that it was difficult to conclude if separation increased or decreased CH_4 emissions. Whether methane emissions are reduced depends on the storage conditions of the fractions, and the composition of the manure. For N_2O, the solid fraction behaves as untreated solid manure showing higher emissions in storage (Hansen et al. 2006). Kupper et al. (2020) also concluded that N_2O emissions were increased.

The process can lead to aeration which induces nitrification and denitrification to nitrite/nitrate with the aim of a complete denitrification to N_2. If the process is not properly controlled, aeration can produce substantial amounts of NH_3 and N_2O (Loyon et al., 2007). A reduction of CH_4 by ca. 50% to almost 100% emissions was observed by Amon et al. (2006) if slurry aeration was applied.

In practical terms, the on-farm solid-liquid separation is a complicating operation with the risk of producing additional NH_3 and N_2O emissions even if CH_4 is reduced.

---

[2]https://www.fwi.co.uk/machinery/technology/cornish-farm-creates-kit-to-turn-slurry-into-fuel

[3] https://www.albersalligator.com/slurry-storage-covers/alligator-slurry-bag/?lang=en

[3] www.quberenewables.co.uk

5.3.4 Anaerobic digestion

Anaerobic digestion promotes a proportion of organic N to be converted to NH_4^+, but with little opportunity for N_2O emissions (Bernet et al., 2000). Digestion increases the methane production for use as a fuel - this increase leading to reduction of the potential for methane emissions in the subsequent storage of the digestate.

Anaerobic digestion reduces the amount of degradable carbon (organic matter) applied in a single application to the soil and hence tends to decrease N_2O emissions relative to untreated manure (Montes et al., 2013). Ammonium ions and ammonia in the digestate solution are transferred to the soil when the digestate is land applied (Bernet et al., 2000). Anaerobic digestion increases plant availability of N and provides less energy to support the growth of N_2O -forming microorganisms, reducing the potential of N_2O emissions when applied to soil.

Because the low energy density of liquid manure reduces the generation of methane, farmers add additional biodegradable materials such as maize. In principle there is also the option of adding biodegradable food waste (Banks et al, 2011), increasing methane generation by a factor of 2-3. But recovery of food waste through anaerobic digestion is subject to the Animal By-products Regulation (ABPR) (EC 1774/2002), which is designed to protect both animal and human health by preventing the spread of animal disease.

One concern is pollution from spreading of digestate on land. The solution has only about 5% solids and can run off land unless injected. The fertilisers (N,P,K) are soluble and can easily be lost through run-off. Spreading on growing crops is essential to reduce losses, taking account of the crop needs. Winter spreading when crops are dormant needs to be avoided.

Although the reduction of manure organic matter content is expected to reduce N_2O emissions from manure-amended soils Thomsen et al. (2010) reported higher N_2O emissions when treated manure was applied in a wet spring season. Petersen and Sommer (2011) concluded that prediction of N_2O emissions from manure-amended soil depends on manure composition and soil conditions. Masse et al. (2011) noted high variability of N_2O emissions.

In summary anaerobic digestion is of considerable value in capturing methane although the subsequent effects of the digestate in reducing nitrous oxide emissions on soil are variable. From a farmer's perspective, the key issue is the cost of equipment, the possible need for supplementary feed stock to increase gas yield, and the operational control of the plant – both digester and energy recovery. Herd size generally needs to be large to generate sufficient slurry, with the need to maintain a continuous supply throughout the year. This is difficult to achieve if animals are grazing in the summer months.

5.3.5 Acidification

Acidification is another technique that has been applied to reduce ammonia emissions by favouring the formation of ammonium ions in the slurry instead of ammonia gas which is released into the atmosphere. Misselbrook et al. (2016) found that acidification of cattle slurry to pH 5.5 reduced CH_4 and NH_3 emissions by 61 and 75%, respectively. Nitrous oxide emissions were also monitored but were insignificant. Cao et al. (2020) showed manure acidification to pH5 and pH6 reduced CH_4 emissions by ~20% and reduced NH_3 emissions by 70%, mineral acids such as sulphuric acid being most effective. Some of the variability may be explained by the degree of aeration of the slurry.

The review by Kupper et al. (2020) also found that acidification reduces NH_3 and CH_4 emissions during storage while an increase occurs for N_2O and a minor change for CO_2 as compared to untreated slurry. For example methane emissions reduced by 61% for cattle slurry and 96% for pig slurry. Ammonia emissions were 71% and 77% less respectively but led to a 4% increase in nitrous oxide emissions for cattle slurry and 39% for pig slurry.

Fangueiro et al. (2015) found that acidified slurry had other benefits when applied to soils. A delay of ammonium N nitrification was observed in soils amended with acidified slurries, relative to non-acidified ones. This delay lasted for about 20 days, for both pig and cattle slurry. Furthermore, for more than 60 days, the NH_4^+ concentration in soil amended with acidified slurry or the liquid fraction of slurry remained significantly higher than in soil amended with the raw materials. N fertilisation is easier to manage with acidified slurry, since the NH_4^+ content is more constant relative to non-acidified slurry due to minimal NH_3 losses.

Roboredo et al. (2012) observed a significant effect of acidification on the Phosphorus (P) availability in soil as well as its evolution with time. Slurry acidification increased the most labile fraction of P. Petersen et al. (2013) also reported an increase of P availability in soils amended with acidified slurry, relative to non-acidified slurry.

In Denmark, acidification of slurry has come into commercial use for treating 20% of slurries (Jacobsen, 2017), primarily as a way of reducing ammonia emissions. The total acid consumption depended on slurry characteristics. A Danish study used acidification at up to 4 litres/tonne of slurry (Vestergaard, A.V., 2014). The average price per treated tonne manure was 6.5 DKK for field acidification with an acid consumption of 1 litre of acid per tonne manure, while for storage acidification the cost was 8 DKK per tonne of slurry using 2 litres of acid per tonne (currently 1DKK = £0.11) equivalent to £0.72/tonne manure, £0.88/tonne of slurry. Costs for applying 30 tonnes/ha were from £22-£44 per hectare. This was based on acid price of between 1.97 to 4.00 DKK per litre and the application cost of 1-5 DKK per tonne of slurry.

The benefits of reduced emissions show in the nitrogen effect of field and storage acidification – a benefit of between 7-18 kg N per ha. The grain yield increase attributed to the extra nitrogen was 15 kg per kg nitrogen. In addition the sulphur in the sulphuric acid has a fertiliser effect. But it may be necessary to lime soils to prevent reduction of pH.

In summary, acidification has the potential to reduce methane and ammonia emissions substantially from slurry stores – up to 96% and up to 77% respectively, but nitrous oxide emissions are increased. Reduction of ammonia emissions is important in reducing indirect emissions following leaching and run-off. Acidification also makes slurry N and P more available in soils. There are economic benefits in reducing fertiliser requirements, which may counterbalance the additional costs, particularly as fertiliser prices rise.

5.4 Manure and Fertiliser spreading

N_2O emissions from manure following application to land depend on the application timing and method of incorporation, manure composition, soil type, temperature, and rainfall (Chadwick et al., 2011). For slurry spreading, many of the studies focus on ammonia reduction. The main options are surface spreading, trailing hose, trailing shoe or shallow injection. During spreading, NH_3 emissions are less with the trailing shoe (65%) or with an open-slot injection (70-80%) than with trailing hose (35%) when compared to a standard splash plate system (Webb et al., 2010). However, the same authors note a large variation in the span of reported data.

Duncan et al. (2016) also showed relative NH_3 emissions were dramatically lower on injection-applied plots treated with cattle slurry. Injecting manure reduced emissions by up to 98% compared with broadcast slurries. N_2O emissions increased for injected plots by up to 2.5 times because soil anaerobic conditions that are favourable for denitrification can result in increased N_2O emissions compared with surface broadcasting. But under aerobic soil conditions slurry injection has the potential to reduce NH_3 emissions without increasing N_2O emissions.

These conclusions were supported by a review of N_2O mitigation options for manure management by Montes et al., (2013) who reported that whilst incorporating or injecting manures into the soil is very effective in reducing NH_3 emissions, it can leave more N susceptible to loss as N_2O, a finding confirmed by Velthof et al. (2003). Organic material added to soil through manure application can stimulate denitrification and also speed up soil respiration, depleting oxygen in soil pores, and triggering denitrification and release of N_2O (Lazcano et al., 2016).

Thorman et al. (2020) compared slurries broadcast on the soil surface with application using bandspreading techniques (trailing hose and trailing shoe) to minimise NH_3 losses. For the spring application, mean NH_3 losses were lower from the bandspread slurry treatments compared to the surface broadcast slurry. The higher N pool in the soil would be expected to lead to higher N_2O emissions, but this was found only for spring applications, possibly when soils are saturated and anaerobic.

For arable land, where manures, both liquid and solid, are applied to land, immediate incorporation by plough is the most effective option to reduce emissions. In their review, Webb et al. (2010) reported that rapid incorporation of slurries or solid manures by ploughing within 4-6 hours is an effective abatement technique, reducing NH_3 emissions by up to 90%. Similar incorporation of solid manure has also been reported to reduce subsequent emissions of N_2O (Webb et al., 2004).

In a modelling study of the impacts of NH_3 abatement measures, on nitrate leaching and N_2O emissions using the National Ammonia Reduction Strategy Evaluation System (NARSES) model, Webb et al. (2006) found that the increase in N_2O emissions was never more than 2% of the NH_3 -N conserved. In total, N_2O emissions were changed little by the adoption of NH_3 abatement techniques. Some NH_3 abatement methods also decrease emissions of both nitrate and N_2O. Obviously the reduction in ammonia emissions means that greater fertilising effect is retained and this can reduce the amount of additional fertiliser required by the crop. This can have benefit in reducing N_2O emissions.

In summary, the incorporation of manure below the surface of the soil reduces ammonia emissions markedly by up to 98%, and whilst the amount of the N reservoir in the soil is increased, nitrous oxide emissions can be small, and are reduced if the soil is well drained and aerobic. Reduction of ammonia emissions to the air reduces the indirect N_2O emissions, and improves air quality. Chadwick et al. (2011) concluded that soil and environmental conditions that give rise to N_2O production and emission (e.g., warm and wet soils) can be more important than the application method in controlling N_2O emissions. This places an emphasis on applying manure in dry weather conditions to growing crops which can utilise the manure immediately. A secondary benefit is that the reduction of ammonia emissions means that less N fertiliser needs to be applied to meet crop needs – reducing N_2O emissions from the fertiliser application.

5.4.1 Nitrification inhibitors for manures and fertilisers

Nitrification inhibitors (NIs) are compounds that delay the bacterial oxidation of ammonium to nitrite by depressing the activity of Nitrosomonas bacteria in the soil. NIs deactivate the enzyme responsible for the first step of nitrification, the oxidation of NH_4^+ to NO_2 (Subbarao et al., 2006). As nitrate is the substrate initially required for denitrification, the use of a NI decreases N_2O emissions from both processes; nitrification and denitrification. The inclusion of an NI with any NH_4^+ -based N fertiliser (including urea-based or other organic fertilisers which subsequently convert to NH_4^+ will therefore retain the N in the soil in the NH_4^+ form for longer. The most common NIs used in fertilisers are dicyandiamide (DCD) and 3, 4-dimethylpyrazole phosphate (DMPP) (Liu et al., 2013).

Many studies have demonstrated that NIs, such as dicyandiamide (DCD), can reduce N_2O emissions Gilsanz et al., 2016; Chadwick et al., 2018), but their efficacy in practice has been shown to depend on a number of external factors such as temperature, moisture, and soil properties including clay and organic matter content (Montes et al., 2013; McGeough et al., 2016). A global meta-analysis found that NIs were effective in reducing N_2O emissions by 42% and 40%, for DCD and 3,4-dimethylpyrazole phosphate (DMPP), respectively (Gilsanz et al., 2016).

In a study at 14 English sites, Misselbrook et al. (2014) examined the effectiveness of DCD on direct nitrous oxide emissions from the application of ammonium nitrate, cattle urine and cattle slurry. Mean reduction efficiencies of 39, 69 and 70% for ammonium nitrate, urea and cattle urine, respectively were observed. When included with cattle slurry a mean, non-significant reduction of 56% was observed. There were no N_2O emission reductions observed from the limited assessments of the other nitrification inhibitors. Generally, there were no impacts of the nitrification inhibitors on NH_3 volatilization, nitrate leaching, crop yield or crop N offtake. Use of DCD could give up to 20% reduction in N_2O emissions from UK agriculture (Misselbrook et al, 2014).

Trials in New Zealand reported that DCD reduced N_2O losses from field-deposited urine and faeces by an average of 50% Gillingham et al. (2012 and Chadwick et al. (2018) found that DCD reduced the N_2O Emission Factors from urine patches by an average of 46% under UK conditions. Luo et al. (2008b) reported up to 45% reduction in N_2O emissions from dairy cow urine applied to various soils in New Zealand by DCD but pointed out that the effectiveness of these compounds may be reduced under heavy rainfall. National trials in New Zealand reported an average N_2O reduction by DCD of 50% (Gillingham et al., 2012).

The effect of seasonal conditions was shown by Cardenas et al. (2016). DCD was only effective in reducing N_2O emissions for the spring application of urine and dung. Barneze et al. (2015) found that the use of DCD in reducing N_2O emissions from UK grazing in summer conditions was not significant. Under the conditions of these experiments, there was a very fast degradation of DCD in the soil, with a half-life of about 10 days, probably due to the elevated soil temperature. Other studies have also shown that as the temperature increases above 10 C, there is a linear decrease in the effectiveness; at temperatures of 25 C, the inhibitory effect lasts only 2–3 weeks (Zerulla et al., 2001). McGeough et al. (2016) also found that the efficacy of DCD was adversely affected by high temperatures and in soils with high clay and high organic matter content. The % inhibition in net Nitrate production and N_2O emissions by DCD was significantly lower in grassland soils than in arable soils because of the higher organic content of grasslands.

The review by Gilsanz et al. (2016), highlighted the paucity of studies regarding DMPP efficacy in reducing N_2O emissions, in comparison to DCD. This is especially true of urine-influenced soils. Marsden et al. (2018) tested DMPP applied at a rate of 1 kg/ha but found that it had no effect on overall cumulative N_2O emissions.

An alternative inhibitor for urea fertilisers is the urease inhibitor (UI), N-(n-butyl) thiophosphoric triamide (NBPT). Urea hydrolyses in the soil by the action of microbial urease to generate ammonia which can be lost by volatilization or oxidised to nitrate by microbial nitrifiers. NBPT occupies the active sites in urease and is the basis of commercial products that are applied together with urea fertilisers (Sigurdarson et al. 2018).

NBPT is reported to delay the hydrolysis of urea fertiliser by 7 to 10 days (Zaman et al. 2008), resulting in a smaller pH increase around the urea granule than for urea alone, and hence lower ammonia volatilization losses. In a large UK study, Chambers and Dampney (2009) reported a mean ammonia emission reduction of 70% (range 25–100%) from the use of NBPT with urea, and on average the use of NBPT increased crop N recovery compared with urea alone. However, differences at an individual site were not always significant and this may indicate the lack of effectiveness of the UI due to rapid breakdown in soil under certain conditions.

Smith et al. (2006) found some evidence that N_2O emissions from urea were smaller than from Ammonium Nitrate or Calcium Ammonium Nitrate, but when this was adjusted for loss of ammonia by volatilization, there was generally little difference between different forms of N. Emissions from urea modified by the addition of NBPT were lower than corresponding emissions from nitrate forms, suggesting that the use of a urease inhibitor can provide some mitigation of N_2O, as well as NH_3 emissions. Carswell et al. (2018) also found that urea with NBPT treatment reduced NH_3 emissions by 48 – 65% relative to the urea treatment.

In a meta analysis, Maaz et al. (2021) found the use of urease inhibitors, nitrification inhibitors, or polymer coated urea reduced N_2O emissions. The current finding of a reduction of 24% falls within the range reported by other meta-analyses, where nitrification inhibitors alone or combined with urease inhibitors reduced N_2O emissions by 8%–100%.

Issues have been raised when using DCD, as traces of DCD were found in milk when DCD was directly fed to animals (Welten et al., 2016), and grass leaves from grasslands to which DCD had been applied have been reported to contain traces of DCD (Pal et al., 2016). The fate of DCD in the environment is unclear, although its rapid degradation particularly at higher ambient temperatures is important in reducing persistent residues. Kamal et al. (2021) concluded that for the use of NIs in livestock grazed systems, research is needed to establish acceptable maximum residue level of NIs in soil, plant, and animal products.

Lam et al. (2017) caution the need for an integrated assessment of emissions. Though nitrification inhibitors show promise in decreasing nitrous oxide emissions, they may be much less effective than previously thought when both direct and indirect emissions are taken into account. Whilst nitrification inhibitors are effective at decreasing direct N_2O emissions and nitrate leaching, limited studies suggest that they may increase ammonia volatilization and, subsequently, indirect N_2O emissions.

There are some additional costs associated with using NIs. Subbarao et al. (2006) estimated that DCD added about 25-30% to the cost of fertiliser N. Trenkel (2010) estimated the additional costs to be 1.3-1.6 times the cost of standard fertilisers but that was based on data from 2005. Scaled up production may reduce the cost, although the study by Misselbrook et al. (2014) did not find any counterbalancing benefits in terms of increased crop yields.

In NIs were effective in reducing N_2O emissions by 42% and 40%, for DCD and DMPP, respectively (Gilsanz et al., 2016). The study by Maaz et al. (2021), showed that there is much variability in the effectiveness of these NIs. Effectiveness is reduced by higher temperatures and soil characteristics. Given the variability of observed reductions, it may be cautious to apply the conclusion of Misselbrook et al. (2014) that an average 20% reduction in nitrous oxide emissions can be achieved in UK agriculture by using inhibitors. Larger reductions up to 50% may apply under optimal conditions.

5.4.2 Fertiliser type

Within England and Wales, the main fertiliser types by % amounts are Ammonium Nitrate (AN) (39.1%), urea (U) (8%), Calcium Ammonium Nitrate (CAN) (1.6%) and Urea Ammonium Nitrate (UAN) (14.2%) (British Survey of Fertiliser Practice 2019). Other fertilisers include compound fertilisers containing Phosphorus (P) and Potassium (K). However, U can be susceptible to large NH_3 losses, which can be in the range 5 – 68% of the fertiliser-N applied (Chambers and Dampney 2009), compared with <3% for an AN. However AN may be subject to greater nitrous oxide emissions particularly when applied in wet conditions (Smith et al. 2012). In a data synthesis, Stehfest and Bouwman (2006) also concluded there was no significant difference among most fertiliser types in terms of N_2O emissions.

However from a meta analysis of 21 studies in UK and Ireland Cowan et al. (2020) showed that AN and CAN are the largest emitting fertiliser types by mass; however, emissions from AN applications were significantly lower for applications to arable fields than to grasslands. Emissions associated with urea were significantly lower than AN for grasslands but slightly higher for arable fields – showing the difficulty of establishing clear universal trends.

From the perspective of N_2O mitigation, the effectiveness of polymer-coated fertilisers (PCFs) has been tested in several studies (Akiyama et al., 2010). A comprehensive analysis showed that PCFs significantly reduced N_2O emissions by 14%-58% with an average of 35%. However, most of the existing studies were conducted in cropland. Slow release fertilisers would incur significant extra costs for their supply.

In summary, although urea seems a better choice in terms of N_2O emissions, ammonia emissions are higher. DEFRA project NT 26 (Smith et al, 2006) reported that ammonia emissions from granular urea were <u>on average</u> 27% (grassland) and 22% (arable) of the total N applied, compared with about 2% from N applied as AN. When indirect emissions of N_2O following volatilisation and re-deposition of ammonia are taken into account, the total N_2O emissions from AN and urea are similar.

5.4.3 Efficiency of N fertiliser and manure use

N fertiliser and manures that are not taken up by the crop or immobilised in soil organic N pools-which include both microbial biomass and soil organic matter—are vulnerable to losses from volatilization, denitrification, and leaching. N management to increase Nitrogen Use Efficiency (NUE) has been recognised as an effective way to mitigate N_2O emissions from agriculture (EU Nitrogen Expert Panel (2015). NUE is defined as

N output / N input, where

- N output is the total N measured in the harvested grass, and
- N input is the amount of fertiliser/manure N applied in the year plus atmospheric N deposition.

The overall NUE of a cropping system can therefore be increased by achieving greater uptake efficiency from applied N inputs, by reducing the amount of N lost from soil organic and inorganic N pools, or both.

Literature reviewed on improving NUE in crop-production systems (Cassman et al., 2002) has emphasised the need for greater synchronisation between crop N demand and the N supply from all sources throughout the growing season. This approach explicitly recognises the need to efficiently utilise both indigenous and applied N and is justified by the fact that losses from all N-loss mechanisms increase in proportion to the amount of available N present in the soil profile at any given time. Hence, uptake efficiency from a single N-fertiliser application typically decreases in proportion to the amount of N-fertiliser applied. The same principle applies to available N derived from organic N sources such as legume green manures, cover crops and animal manures.

Among the many practices of fertiliser N management, fertiliser N rate is most strongly linked with affecting soil N_2O emission. For Wales, the data from the British Survey of Fertiliser Practice for 2019 are limited since the data are generally reported for England and Wales. In Wales, fertiliser is spread on 82% of tillage area, and 64% of grassland area. Farmyard manure is spread on 66% of tillage area and 38% of grassland. Average N application rates were 90kgN/ha on tillage areas and 53kgN/ha on grassland. But dairy farms have higher applications rates - in the region of 140-150kgN/ha. Farmers appear to reduce mineral fertiliser application rates by about 20kgN/ha where they use farm yard manure, although the N content of manures is rarely measured.

There is general consensus that reducing the N rate is a reliable way for reducing N_2O emissions (Venterea et al., 2012). Measures to synchronise N supply to crop needs are often aimed to reduce N loss via NH_3 volatilization and nitrate leaching, which in total account for up to 50% of N inputs. Limited data are available to show whether synchronizing of nitrogen supply to requirements decreases soil N_2O emission from grassland.

Based on studies in cropland, it seems that improving NUE cannot consistently reduce N_2O emissions (Phillips et al., 2009), probably because the practices that improve NUE by reducing NH_3 and/or Nitrate losses may make more N available in the soil for both N uptake in crops and soil N_2O production (Venterea et al., 2012).

For the few N_2O response experiments in which more than two levels of N were applied, N_2O flux in response to increasing N rates has been described by both linear and nonlinear functions (Li et al. 2015). For example, Cardenas et al. (2010) showed that the N_2O emissions from applying AN fertiliser varied in a non-linear way – higher application rates leading to much higher emissions. However, Cardenas et al. (2019) showed that the trend of non-linearity is not consistent. In a study of 5 sites, the effect of increasing N fertiliser rate on annual N_2O emissions showed linear responses for 3 sites, and exponential curves at the remaining 2 sites. For grassland, Cardenas et al, (2019) found that from grass N offtake amounts, not all N added was used by the plants, resulting in an average surplus of 0.32 kg N per additional kg N applied.

There is evidence that N_2O emissions change slightly when fertiliser N inputs were below a certain level, but increase rapidly above that level. For example, in a study of cropland, N_2O fluxes were moderately low (ca. 20 g N_2O /ha/day) at levels of N rates up to 101 kg N/ha, where grain yields were maximised, after which fluxes more than doubled (to >50 g N_2O /ha/day) (McSwiney and Robertson, 2005).

Similarly, in another study with multiple sites, for two N fertiliser rates above those recommended for maximum economic return (135 kg N/ha), average N_2O fluxes were 43%

(18 g N_2O /ha/day) and 115% (26 g N_2O /ha/day) higher than were fluxes at the recommended rate, but the corn grain yield only increased by 2% and 6%, respectively (Hoben et al., 2011).

Overall, the above examples indicate the potential to lower agricultural N_2O fluxes with little yield penalty by reducing N fertiliser inputs to levels that satisfy crop needs. In estimating the fertiliser requirements, it is essential to factor in the contribution of applied manures to the overall crop requirements. AHDB Fertiliser Manual RB209 presents recommendations on fertiliser requirements which give the most **economical** returns for farmers, but this may be more than is justifiable in terms of GHG emissions. Currently, AHDB is revising its fertiliser recommendations to take account of the large increase of fertiliser prices caused by increased energy costs in 2022.

AHDB guidance is that it is important to get maximum response of fertilisers by ensuring that soil pH is a minimum of 6. This is important as a mitigation measure to reduce fertiliser requirements of crops.

The extent of possible N fertiliser reductions can be estimated from the DEFRA Soil Nutrient Balances (2019) available on a UK and England basis. Estimates for Wales are not available. For the UK, excluding rough grazing:

<u>N Inputs</u>

Mineral fertilisers	88.9 kgN/ha
Manures	82.7 kgN/ha
Atmospheric deposition	10.4 kgN/ha
Biological Fixation	11.4 kgN.ha
Total	192.9 kgN/ha
<u>N Offtakes</u>	108.9 kgN/ha
<u>Surplus</u>	84.0 kgN/ha

The surplus indicates the potential reduction in N inputs that may be attained though it is important that soil N levels are not mined. There could be substantial reduction of the use of mineral fertilisers given that the total surplus is equivalent to the input from mineral fertilisers.

In summary, reduction of emissions would be delivered by more efficient use of fertilisers, optimum timing of applications and ensuring the full value of manures and clover/legumes is factored into nutrient management plans required by Water Resources (Control of Agricultural Pollution) (Wales) Regulations. Good practice requires periodic soil analyses, and analyses of manures to assess the amount of mineral fertiliser required to meet crop needs. For 2018, application of mineral fertilisers caused N_2O emissions of 472Kt CO_2e – 28.3% of total N_2O emissions. Taking account of the excess N inputs from the DEFRA Soil Nutrient Balance a 50% reduction in the surplus N could be achieved by halving mineral fertiliser use. This would deliver 236Kt CO_2e emission reduction.

5.5 Land Management

5.5.1 Improving land drainage

Soil moisture is a major driver of N_2O emissions as it regulates the oxygen available to soil microbes (Butterbach-Bahl et al, 2013). Maximum N_2O emissions occur in the range of 70–80% water-filled pore space (WFPS) depending on soil type (Davidson et al, 2000). At higher soil moisture, the major end product of denitrification is N_2. Firestone and Davidson (1989) determined that oxygen supply was the most important controlling factor in fertilised soils. At WFPS between 45 and 75 % nitrification and denitrification act as N_2O sources.

Dobbie and Smith (2006) found that rises in the water table were accompanied by exponential increases in N_2O emissions, through the associated increases in the water-filled pore space of the topsoil. Modelling predicted that if the water table could be managed such that it was kept to no less than 35 cm below the ground surface, fluxes during the growing season would be reduced by 50%, while lowering to 45 cm would reduce them by over 80%. The conclusion was that draining grasslands where the water tables are only rarely nearer to the surface than 35 cm would substantially reduce N_2O emissions.

Davidson et al. (2000) also suggested that at WFPS values above 90 % only N_2 is produced. The “optimum” WFPS for N_2O emissions varies from soil to soil (Davidson, 1991). Soil structure could be influencing this effect and it strongly interacts with soil moisture (van Groenigen et al., 2005) through changes in WFPS. Luo et al. (2013) found that as well as soil moisture and temperature, there were other controlling factors such as N and C availability or microbial community dynamics that might exert a significant control on the variation of nitrous oxide fluxes.

Cardenas et al. (2010) showed the varying moisture effect on nitrous oxide emissions in comparing Western UK sites at Pwllpeiran and Rowden vs an Eastern UK site (High Mowthorpe). The N_2O emission for a 100 kg N/ha/yr application was 3.9 kg N_2O–N/ha/yr for the Western sites and 0.5 kg N_2O –N/ha/yr for the Eastern sites. The conclusion was that higher rainfall and wetter soils in the west stimulated N_2O production ‘hot-spots’ to a greater extent, generating larger and more frequent peak N_2 O emissions as a result of intense denitrification from anaerobic soil conditions.

Soil compaction due to livestock treading and the use of heavy machinery affect soil structure and emissions as reported by studies relating bulk density to fluxes (Klefoth et al., 2014) and degrees of tillage to emissions. Compaction is known to affect the size of the larger pores (macropores) thereby reducing the soil air volume and therefore increasing the WFPS (for the same moisture content) (van der Weerden et al., 2012).

The conclusion is that land drainage to reduce the water table below 35cm together with maintaining effective soil structure is an important mitigation method. The GHG Inventory takes account of WFPS using soil data from Cranfield University. Given the high rainfall in Wales, land drainage is an important option, but with substantial costs. Determination of areas requiring drainage would require substantial survey work. Drainage could have positive effects by increasing soil water capacity and reducing surface run-off. In Wales the Pontbren scheme demonstrated benefits for flood mitigation (Woodland Trust, 2013)

5.5.2 Biological fixation with legumes

Biological N fixation (BNF) in association with forage legumes provides an alternative N source for grasslands. White clover is the main legume in pastures and meadows of temperate regions. Average biological Nitrogen fixation in grazed permanent clover/grass pastures in temperate regions of the world has been reported to be 80-100 kg N/ha/yr (range 10-270 kg N/ha/yr) (Ledgard et al, 2009). This fixed N becomes available slowly over time to the grass in pastures after it is released into soil via exudates from living legume roots, by mineralization of legume tissues and in excreta after consumption by grazing animals (Ledgard et al., 2009). Andrews et al. (2007) concluded that herbage and milk production from white clover-based pastures (perennial ryegrass with 20% white clover in herbage DM) are likely to be similar to that from a perennial ryegrass pasture receiving annual input of 200 kg/ha of fertiliser N. AHDB RB209 estimates the contribution from white clover can be up to 180kgN/ha.

A data synthesis indicated that the average soil N_2O emissions from field-grown legumes, N fertilised grass pastures and crops, and unfertilised soils are 1.29, 3.22 and 1.20 kg N /ha/yr. respectively (Jensen et al., 2012). Therefore N_2O emissions from grass legume mixes are only slightly greater than background emissions.

Although limited studies have been conducted to compare nitrate leaching or NH_3 volatilization in legume and fertiliser-based systems, the current evidence suggests that N losses via these two pathways may be lower in legume-based systems (Crews and Peoples, 2004). Consequently, indirect N_2O emissions from these N losses can be reduced. NH_3 volatilization from legume residues was found to be much lower.

Schils et al. (2005) used a farm level approach to compare GHG emissions from grass/fertiliser and grass/clover farms. Compared to a grass/fertiliser-N system, the GHG emissions (excluding carbon sequestration) from a grass/clover farm were 23% lower per ha and 11% lower per kg milk

AHDB Fertiliser Manual RB209 advises that little fertiliser nitrogen is needed on swards with an appreciable clover content. On average, a good grass and clover sward (30–40% of dry matter of clover) will give an annual dry matter yield equivalent to that produced from about 180 kg N/ha applied to a pure grass sward. Lucerne and red clover crops have no requirement for N fertiliser, apart from that needed for establishment in low nitrogen status soils (up to 50 kg N/ha). Applications of fertiliser nitrogen to grass/clover swards should be made with caution because any form of mineral nitrogen inhibits nitrogen fixation by rhizobia in the clover nodules. There is a risk of the grass responding to the nitrogen and shading out the clover, which can reduce the percentage of clover in the sward.

Grasses generally have faster growth rates than forage legumes in winter and spring and this can make them more suitable for low cost ruminant livestock production systems where housing and feed handling costs are minimised by keeping the animals out grazing for as long as possible each year. Agronomic persistence, defined as the ability of a forage species to maintain herbage production per hectare over a number of years, is an important characteristic of perennial forages and is generally reported as being poorer for most forage legumes than grasses (Phelan et al, 2015).

In summary, establishment of clover on temporary grassland and on permanent grassland amenable to no-till seeding would deliver a significant reduction of nitrous oxide emissions, and reduce the need for fertiliser/manure applications with their associated emissions. However the management of grass-legume leys requires careful management in establishment and maintenance. Legumes also provide N

inputs to arable crops in a crop rotation system. But if a legume crop substitutes for a higher value crop, it can lead to economic costs. Currently biological fixation provides 5.9 % of total N inputs in the UK (DEFRA- UK Soil Nutrient Balances for 2019).

5.5.3 Adopting systems less reliant on inputs

De Ponti (2012) completed a meta-analysis of yield data comparing organic and conventional agriculture from 362 global data sets showed that currently organic yields of individual crops are on average 70%-73% of conventional yields in Northern Europe. The yield gap between organic and conventional agriculture tended to increase as conventional yields increase.

The findings conflicted with those of Badgley et al. (2007), who estimated the average organic yield ratio for all crop types at the global level as 1.32 (i.e. organic would produce 132% of the conventional yield). This estimate did not take account of regional differences. Developing countries tend to have low fertiliser inputs where organic fertilisers are the principal fertiliser. The conclusion was based on the estimate that the total N supplied by leguminous cover crops in organic systems was much greater than the amount of synthetic N currently in use, and consequently, could support yields equivalent to high-yielding conventional agriculture. The authors assumed that 100% of arable land could accept an additional legume crop, following the main crop in the same year. However much of the world's most productive land is already required to carry multiple food crops in a single year to meet food demand.

Inclusion of leguminous crops in crop rotations can be difficult if they substitute for productive crops. An adequate supply of other nutrients through other sources is essential – P, K and S. The nutrient supply to organic crops relies more than conventional crops on better closing of nutrient cycles.

Smith et al. (2019) assessed the production impacts of 100% conversion of land in England and Wales to organic farming, using a linear programming model with inputs on farm structure, soil and rainfall, nitrogen (N) supply/offtake and livestock feed demand. There would be major reductions in wheat and barley production, whilst the production of minor cereals such as oats and rye increases. Monogastric livestock and milk production also decreased considerably by 40-90% because of the reduction of cereal feeds and less intensive grassland, whilst beef and sheep numbers increased. Vegetable production levels were generally comparable to those under conventional farming.

The results showed a major drop in food production, with total food output expressed as metabolisable energy falling to 64% of non-organic baseline levels. This is lower than the conclusion of De Ponti et al. (2012). The study indicated that there would be more reliance on food imports, with the need for diet and reduction of food waste.

Smith et al. (2019) compared the Life Cycle Analyses of organic vs conventional systems for England and Wales based on the above modelling study. Lower GHG emissions under organic cropping were predicted, largely due to replacement of N fertiliser with biological N fixation in leys, resulting in less CO_2 and N_2O from fertiliser manufacture and less N_2O per unit of production. However the production penalty would only be compensated financially if the organic product has a price premium. Poultry meat and egg production generated greater emissions under organic management due to poorer feed conversion ratios, longer rearing times, higher mortality rates and greater leaching losses compared to conventional free range and fully housed systems. Organic dairy, beef and sheep production resulted in lower total GHG emissions per unit area, as a result of the increased efficiency of forage production under organic management, although greater forage intake increases the total

CH_4 contribution. Also lower animal growth rates and lower milk yields would increase emissions per unit of product.

At a national level, the direct emissions associated with organic crop and livestock production are smaller for organic farming compared with conventional: by 20% for crops, 4% for livestock and 6% overall. However, the reduction in production would need to be compensated by production in other countries – with the associated GHG emissions, or a reduction in food waste. Enhanced carbon sequestration rates would be expected for organic farming because of longer crop rotations with leys using forage legumes, as arable land is converted to grassland. However, carbon sequestration would reach a limit after about 20 years.

In summary, organic farming systems for livestock – the predominant farming in Wales – would lead to a small reduction in overall GHG emissions (4 to 6%) mainly through the use of legume crops to fix nitrogen – reducing emissions from fertiliser manufacture and reducing the attendant nitrous oxide emissions from fertiliser spreading on land. But studies show there could be a decrease of milk production, with beef and sheep production remaining broadly similar to current levels.

5.6 Summary Manures and Fertilisers

Li et al. (2013) emphasise that the effectiveness of the combined individual options to mitigate N_2O emissions may not be simply additive. Quantitative analysis of N flows and N_2 O losses in dairy farming systems reveal that the implementation of a package of measures to improve N management may reduce N_2O emissions from dairy farming systems by up to 70% (Velthof and Oenema, 1997). Table 5.1 summarises the mitigation options.

The main mitigation options for nitrous oxide reduction are:

Dairy

- **Optimising diet – reduction of crude protein in diets to reduce nitrogen emissions via urine and faeces,**
- **Slurry systems – manure store covers, and slit injection manure spreading ,**
- **Smart application of manures and fertilisers targeting application to plant needs based on nutrient analysis of soils and manures and timing related to weather conditions**
- **Optimising soil pH for most efficient fertiliser response**
- **Nitrification inhibitors spread with fertilisers to reduce nitrous oxide emissions,**
- **Biological nitrogen fixation with legumes within improved grassland.**

Beef

- **Manure covers for solid farmyard manure**
- **Smart application of manures and fertilisers targeting application to plant needs based on nutrient analysis of soils and manures and timing related to weather conditions**
- **Optimising soil pH for most efficient fertiliser response**
- **Nitrification inhibitors spread with fertilisers to reduce nitrous oxide emissions,**
- **Biological nitrogen fixation with legumes within improved grassland.**

- **Optimising diet – reduction of crude protein in diets to reduce nitrogen emissions via urine and faeces – restricted to winter housed herds fed controlled diets**

Sheep

- **Smart application of manures and fertilisers targeting application to plant needs, supported by soil and manure analysis and timing related to weather conditions**
- **Optimising soil pH for most efficient fertiliser response**
- **Nitrogen inhibitors spread with fertilisers to reduce emissions – restricted to improved grassland. Use of Nitrogen inhibitors for extensive grazing areas is ineffective unless spread very frequently.**
- **Biological nitrogen fixation with legumes on improved grassland.**

There are no specific sheep manure management interventions.

All sectors - Arable and grass cropland

- **The DEFRA UK Soil Nutrient Balances show the excess nitrogen applied to soils is equivalent to the total artificial fertiliser applied for 2019. Targeted smart application of artificial fertilisers combined with accurate spreading of manures, and biological fixation with legumes could reduce artificial application by 50%. Farmers should follow closely the recommendations given in AHDB RB209**

The main mitigation options for methane reduction are

Dairy and beef

- **slurry and solid manure covers,**
- **slurry acidification - used in Denmark to treat 20% of slurry. Requires good management of health and safety issues.**
- **methane capture in housing and covered slurry lagoons**
- **anaerobic digestion – with current technology, limited to large dairy units - it has significant capital costs and operational demands**

Table 5.1 Summary Analysis of Mitigation Options

Option	Comments	Effectiveness in decreasing emissions	
		CH_4	N_2O
Diet – protein reduction	Restricted to cattle fed rations – not grazing, difficult to model	None	-20%
Slurry store covers	Particularly effective if CH_4 can be captured and burnt	- 10 to -60%	None
Solid manure heaps compacted/covered	Mainly reduces nitrous oxide from aerobic conditions	Small	-30%
Solid-liquid separation	Additional procedure for farmers	-50% to -100%	Increases emissions
Anaerobic digestion	Expensive option, requires good process management. Waste regs if importing waste	-50 to -90%	None
Acidification of slurries	Effective for reducing NH_3 emissions. Acid handling difficult	-96%	Increases emissions
Manure spreading – injection and incorporation	Variable improvements – depending on soil conditions. Reduces NH_3 emissions and indirect N_2O	None	Variable
Nitrification inhibitors	DCD and NBPT effective for manure and fertiliser application. Mixed results for grazing animals, with rapid degradation of DCD in warm conditions	None	-42% to -20%
Fertiliser type	No detectable effects	None	None
N Fertiliser and manure efficiency	Substantial reductions possible	None	-50%
Improving land drainage	Substantial reductions possible in high rainfall areas. Expensive	None	-50% to -80%
Biological fixation with legumes	Substantial displacement of fertiliser application. Low emissions. Expensive for arable where legumes displace other crops.	None	-11% to -23%
Adopting systems less reliant on inputs	Emissions saving linked to biological fixation. Lower outputs.	None	4-6%

Based on the summary analysis above, Table 5.2 summarises the overall effect of the reduction options.

Table 5.2 Summary of Emission Reduction from Manure and Fertiliser Mitigation Options

Measure	Emission saving (kt CO2 e)	
N fertiliser and manure efficiency – 50% reduction of fertilisers	236 (N2O)	
N inhibitors – 40% reduction for remaining fertilisers plus all manures	138 (N2O)	
Reducing manure methane emissions from housing and manure stores by 50% by acidification, and methane capture or anaerobic digestion	205 (CH4)	
Total N2O reduction	374	23% Reduction
Total CH4 reduction	205	6% reduction

These are the maximum technical potentials requiring universal adoption in Wales, and not ones that can be delivered immediately. Chapter 6 reviews livestock management mitigation options which could lead to a reduction of the number of ruminants producing manure. The reduction of emissions from the reduction of manure from fewer ruminants has not been estimated.

Chapter 7 discusses the economic and practical aspects of these options. This includes economic assessments and scenarios for uptake rates which form the basis of UKCCC projections (Eory et al, 2020).

6 LIVESTOCK FARMING - DAIRY, BEEF AND SHEEP

6.1 Introduction

Livestock emissions dominate agricultural emissions in Wales (2018 Inventory). Cattle emit 46% of total agriculture emissions and sheep emit 19%. The chief components of the emissions are methane and nitrous oxide. The main methane sources are enteric emissions from the rumen and emissions from manure storage. Other emissions of nitrous oxides arise from deposition of dung and urine, and the application of fertilisers and manures (see Chapter 5). This chapter describes mitigation options for enteric methane emissions.

Enteric emissions of methane result from the digestive process in cattle and sheep. Anaerobic bacteria, protozoa, fungi, and methanogenic archaea in their rumen break down carbohydrates into simple molecules that can be digested by the animals. Methane is a by-product of this process and is released mainly through the mouth and nostrils. Non-ruminant species, such as pigs, also produce methane but amounts are much lower by comparison. Methane production represents a loss of energy ingested as feed, instead of being assimilated in the animal for production. Up to 12% of the gross energy ingested by the animal is lost this way (Johnson and Johnson (1995).

One of the issues of interpreting mitigation options is the method of presenting methane reduction data. GHG emissions are expressed on a digestible energy intake basis or per unit of animal product (i.e., Emission Intensity, EI). EI is most valuable because this reflects most accurately the effect of a given mitigation practice on feed intake and animal productivity (Gerber et al, 2013).

Emission strategies targeted on methane reduction have been researched using the following methods

(i) Diet composition

(ii) Genetic improvement and livestock management

(iii) Dietary additives

The first two methods are based on increasing production efficiency in order to reduce GHG emissions while maintaining the level of production. Notable strategies in this group include increasing feed efficiency and improving the health status of the herd or flock, which act as win–win strategies, improving the environmental sustainability and economic return. These have to be cognisant of the needs to maintain animal welfare for animals to thrive. The third method is based on strategies to reduce methanogenesis by various additives which suppress the biochemical formation of methane.

Capper et al, (2009) commented that “no ‘magic bullet currently exists that can be applied to a single component or process within the dairy system to reduce environmental impact, without incurring potential negative trade-offs elsewhere in the system.” A similar conclusion can be drawn about the beef and sheep sectors.

Improving productive efficiency (output per unit of resource input) spreads GHG emissions over more units of production, allowing the product to be produced at a lower ratio of emissions per unit production (Hyland et al, 2016). Within dairy systems, each animal’s nutrient requirement for maintenance acts as a fixed emission whilst production (growth, pregnancy or lactation) incurs further emissions. These requirements are associated with both economic and environmental costs in terms of resource inputs (feed, water, cropland, fertiliser, fossil fuels) and GHG. Similar principles exist for beef production and fat lamb production where increasing the rates of weight gain means that the proportion of total emissions from the maintenance of the animal is reduced.

6.2 Emission Reduction Options

6.2.1 Diet composition

There is a wide range of types of feed inputs for ruminants, with a high proportion of forages making up the total diet. Both the quantity and quality of the ingested feed regulate the extent of CH_4 production and emission. Methane is sourced mainly from fibre digestion, so concentrated diets based on cereals, meals and oils reduce emissions compared with diets of the same energy content based on forage. However this benefit of concentrated animal feeds has disbenefits in its reliance on non-local feed sources with associated emissions – for example, soya bean meal.

Increasing the level of concentrates in the diet reduces the proportion of dietary energy converted to CH_4 and reduces rumen pH (Blaxter and Clapperton 1965), mainly due to the associated change in fermented substrate from fibre to starch. In addition to reducing enteric CH_4 relative to dry matter intake (DMI), concentrate feeding also improves animal performance. (Beauchemin et al. 2008).

The use of cereal-based diets rich in starch is effective for the reduction of CH_4 production and emissions. Starch produces more propionate than acetate which ultimately reduces the activity of methanogens (Hassanat and Benchaar, 2013;). Due to the reduction of the fermentable substrate, the lower organic matter and fibre degradation reduces H_2 production and hence CH_4 emissions (Alstrup et al., 2015).

The utilisation of higher digestibility forages is an efficient way for CH_4 mitigation. Early harvesting of silage and hay is important to increase digestibility. Use of highly digestible fodder such as legume silage in the ruminant diet reduces the enteric fermentation and manure production, and thereby reduces CH_4 emissions (Hristov et al., 2013). This is because legume fodder has lower fibre content compared to grass fodder.

Van Gastellan et al. (2019) reviewed studies which showed the effectiveness of forage-related CH_4 mitigation strategies, including feeding more highly digestible grass (herbage or silage) or replacing different forage types with maize silage, These strategies are most effective for dairy cattle, are effective for beef cattle to a certain extent, but seem to have minor or no effects in sheep.

Plant breeding has a role in modifying forage and potentially reducing methane emissions whilst improving animal performance of the ruminants (Abberton et al., 2007). In studies of High Sugar (HS) grasses, Moorby et al. (2006) showed that increased Dry Matter intakes by dairy cows given HS grasses led to increased milk protein outputs. Misselbrook et al. (2012) found overall greenhouse gas emissions were reduced by up to 19% per litre of milk, through reductions in both CH_4 and N_2O.

However in contrast, for methane reduction, a review by Parsons et al. (2011) concluded that the prospect for reducing CH_4 emissions, whether per hectare or per unit energy intake or animal product, with HS grasses is uncertain. A simulation experiment suggested that HS grasses may actually increase CH_4 emissions (Ellis et al., 2012).

Whether feeding more grain reduces net farm GHG emissions is uncertain, given the possible losses of soil carbon from cultivations and the nitrous oxide emissions from fertiliser manufacture and application. The overall GHG emissions ultimately depends on the farming system (Beauchemin et al., 2010). Also the scope for increasing the amount of grain fed to ruminants is limited and feeding grain ignores the importance of ruminants in converting fibrous feeds, unsuitable for direct human consumption, to the high quality protein sources milk and meat (Garnett, 2009) – an important factor for Welsh agriculture which is based predominantly on grass.

The use of concentrates may be justified on nutritional grounds in terms of meeting animal requirement for energy, especially in late pregnancy (sheep) early lactation (dairy cows) and the final period of growth (beef cattle). Moreover, there are specific situations (e.g. the high-yielding dairy cow) in which the requirement for metabolisable protein cannot be met by microbial protein synthesis in the rumen and an additional supply of rumen undegraded protein and/or essential amino acids is required. This is especially the case for methionine and lysine where protected supplementation has been shown to increase milk yield (Nichols et al., 1998). On high-forage diets histidine is often first limiting due to the greater reliance on microbial protein (Lee et al., 2014, Wilkinson and Lee, 2017). It is important to recognise that concentrates are usually composed of various proteins and oils from tropical sources which have associated GHG emissions.

In summary, for dairy cows, Knapp et al. (2014) concluded that feed efficiency is best achieved by

- **feeding starchy carbohydrates to supplement forages**
- **feeding high quality forages with high digestibility achieved by early harvesting and good conservation methods, coupled with improved grazing by reseeding and controlled grazing regimes**
- **optimally processing the forages by chopping to increase dry matter digestion**

Feeding and nutrition have the potential to mitigate enteric CH_4/unit milk production in intensive dairy operations by 2.5-15%. For beef cattle and sheep, high quality forages are the most important factor since alternative feed supplements are mainly restricted to housed animals. Feeding higher concentrate diets however leads to conflicts with other societal priorities, such as maximising the availability of cereal crops for direct human consumption.

For the GHG inventory estimates, diet management would be very difficult to monitor because of the complexity of collecting activity data for varying diets.

6.2.2 Genetic Improvement and Livestock Management

Breeding goals, including production and non-production traits, have become the norm in many livestock species. Selection is usually based on a combination of production and ‘fitness’ (health, fertility, longevity) traits. Breeding goals can be devised in a number of ways including the popular method of weighting traits by their relative economic value (REV). These REVs tend to be calculated by estimating the marginal economic cost or benefit to the system of a unit change in the traits being examined.

The selective breeding of the animal is one of the most sustainable means of enteric CH_4 emissions reduction from ruminants (Wall et al., 2009) based on the principle of improving animal productivity i.e. production and quality of meat and milk. Genetic improvement of livestock produces permanent and cumulative changes in performance. There are economic incentives to use improved breeding stock, so reductions in GHG emissions can be achieved without major changes in current farming practices.

There are three routes through which genetic improvement can help to reduce emissions per kg product (Wall et al, 2009).

1. Improved productivity and efficiency in the animal;
2. Reducing wastage (e.g. involuntary culling, empty reproductive cycles) at the herd or flock level;
3. Selection based on emission levels at individual animal level, dependent on effective monitoring methods, and assumptions about heritability of the methane reduction traits.

Selection for productivity and efficiency helps mitigate GHG production as already discussed (Capper et al, 2009). Firstly, higher productivity generally leads to higher gross efficiency (converting feed into product) as a result of diluting the maintenance cost of the productive (and non-productive) animals. Selection for efficient production has a clear benefit in mitigating emissions. Selection solely for higher-producing animals has knock-on impacts on essential 'fitness' traits, which also affects sustainability. Therefore, it is important to consider a broader range of traits in selection indices and the fit with other system characteristics to ensure longer-term sustainability.

Secondly, reducing wastage enables a given level of production to be achieved with fewer higher yielding animals and their followers. In the case of the UK dairy herd, the same quantity of milk was produced in 2005 from a million fewer animals than 20 years earlier, because average annual milk yield per cow increased from 5000 to almost 7000 litres, a 2% increase per year (Gill et al, 2010). Garnsworthy (2004) calculated that the total methane (t/year) associated with the production of one million litres of milk from cows producing 9000 litres/cow per year was just over 50% of the methane which would be associated with cows producing 6000 litres/cow per year, taking into account the differing nutritional requirements. Livestock management measures that save emissions, include reducing first calving of cows to 24 months instead of 30 months, and reducing age at slaughter for beef cattle.

The third way of using genetics to reduce methane emissions is based on the premise that the traits are heritable. Pinares-Patine et al. (2013) emphasised the need to demonstrate that there is repeatable individual variation in this trait and that a portion of this variation is genetically inherited. They showed that both gross methane per day and methane yield (g CH_4 /kg Dry Matter Intake) are heritable and repeatable traits.

Pickering et al. (2013) postulated that potentially the most sustainable way of reducing enteric CH4 emission from ruminants is through genomic analyses to aid genetic selection. Sources of systematic variation in CH4 production by an individual animal include: total feed intake, the nutrient composition of the feed eaten, the proportion and rate of fermentation of that feed in the rumen, feeding frequency, rumen volume and rate of passage of digesta from the rumen, physiological state of the animal and variation between individual animals including that between sire families.

Recent research in beef cattle (Donoghue et al., 2015) and sheep (Pinares-Patiño et al., 2013) is increasingly supportive of CH4 traits being heritable with achievable improvement by direct selection. Wallace et al. (2019) carried out a 1000-cow European study to understand to what extent ruminant microbiomes can be controlled by the host animal and to identify characteristics of the host rumen microbiome axis that determine productivity and methane emissions. On the basis of the genetic determinants of the heritable microbes, they concluded that it should be possible to optimise their abundance through selective breeding programmes. The results are likely to be applicable to beef animals and other ruminant species.

Compared with genetics to improve performance, where results can be recorded and analysed relatively easily, it is important to note that breeding traits for methane reduction are more difficult to assess (Lovendahl et al. 2018). This is because direct measurement of all sources of methane emissions from individual animals (exhaled by the animal due to enteric fermentation, flatulence and to a lesser extent from manure) can prove difficult and has required development of various analytical techniques. The experimental challenges increase for monitoring large groups of animals over long time periods – a requirement to get robust data on which to base breeding decisions.

Consequently, proxy indicators of methane emissions have been used. Hegarty et al. (2007) showed that there is a decreased enteric methane production per day in beef animals selected for lower residual feed intake (RFI). Reduced RFI is akin to selection for high feed efficiency as an animal is eating less but maintaining a similar growth rate (high net feed efficiency) and, therefore, less feed is required to produce a unit of output. The research showed the possibilities for selection of reduced GHG emissions through the selection of animals, which use less feed than average to achieve a given level of performance. This correlates with animals that produce less methane.

Typically, selective breeding can achieve annual rates of response of between 1% and 3% of the mean in the trait (or index) under selection (Simm et al. 2004). Recent modelling studies in the UK by Genesis-Faraday (Jones et al., 2008) have indicated that past selection for production traits such as growth rate, milk production, fertility and efficiency of feed conversion has resulted in decreases in GHG production per unit of livestock product of about 1% per annum. These decreases have been greatest in those species in which the greatest genetic gains have been achieved – poultry, dairy cows and pigs. However, the reductions were much smaller in beef cattle and sheep. This was due to poorer rates of genetic improvement across the population in these sectors and poor dissemination of information from elite breeders to the commercial populations (Gill et al. 2010).

Knapp et al. (2014) concluded that for intensive dairy herds, genetic selection for feed efficiency, heat tolerance, disease resistance, and fertility can augment selection for milk yield in reducing enteric CH4 /milk product with the potential of 9 to 19% reductions. To achieve enteric reductions through genetic selection requires appropriate supporting management, including feeding and nutrition, health, reproduction, and housing facility design.

To give an estimate of the timescale for genetic improvement to reduce methane emissions, Alford et al. (2005) modelled the reduction of RFI as a result of using more feed efficient bulls for the Australian beef herd. This was to simulate the spread of improved RFI genes through the breeding herd over 25 years. For an individual adopting herd, the annual methane abatement in year 25 was predicted to be 15.9% lower than in year 1 i.e. an average 0.64% decrease/year.

In summary, genetics and livestock management could deliver emission savings of 9-19% at a rate of 0.6-3% per year, but for beef cattle the reduction may be less. For sheep, Cottle and Connington, (2012) suggested that genetics may deliver methane emissions reduction up to 30% over 20 years for sheep – depending on the economic incentives to reduce emissions.

6.2.3 Dietary additives

A range of additives has been tested to reduce methane generation in the rumen. Some studies have shown promise in laboratory studies, but have not been not proven *in vivo.*

6.2.3.1 Oils

Use of essential oils has been reviewed by Benchaar and Greathead (2011). The plant extracts and secondary metabolites both decrease CH_4 production in ruminants by improving feed efficiency, nutrient utilisation, defaunation and decreased degradation of dietary proteins. Plant extracts such as medium-chain saturated fatty acids and rumen modifiers increase feed conversion efficiency by 6% and thereby reduce enteric CH_4 production (Moate et al., 2011). Amending dietary feed with 1% crude fat may reduce CH_4 emission by up to 5%. Although changes due to diet management are short-term and target only changes in enteric emissions, the long-term sustainability of diet additives needs further research.

6.2.3.2 Lipids

Lipids are high-oil by-products from the biofuel industries, wet distillers grains and mechanically extracted oilseed meals incorporated into feeds. In the UK, one main source is rapeseed cake. Based on data reviewed by Hristov et al. (2013), it was concluded that inclusion of lipids in ruminant diets would mitigate CH_4 production, but it may also depress feed intake and, consequently, animal productivity. Therefore, at least part of the mitigation effect reported with lipids is a result of decreased intake of dietary carbohydrate, which is a consequence of decreased DMI as a result of lipids replacing carbohydrate in the diet.

In a study with beef cattle in Scotland, a 7.5% reduction in methane emissions was observed, using rapeseed cake (Cooke et al, 2014). In another meta-analysis of dairy herds in Canada (Jayasundara et al. 2016) the study found a 9% reduction in methane emissions.

6.2.3.3 Tannins

Tannins may also reduce enteric CH_4 emissions. Tannins are usually found in low concentrations in forage legumes such as birdsfoot trefoil, greater birdsfoot trefoil, and common vetch. Tannin extracts are also incorporated into formulated feeds. There has been a general view that tannins have a negative effect on animal performance – for example, milk production may be compromised (Beauchemin et al. 2007). In a review of studies, Aboagya and Beauchemin (2019) reported a methane reduction of 6-68% *in vivo.* The large variability occurs because of the varying differ types of tannin - the concentration, diet and animal type. The review concluded that research on the effects of tannins for CH_4 mitigation is at an early stage and warrants further investigation.

6.2.3.4 Ionophores

An ionophore - monensin, has been extensively used as a feed additive for ruminant livestock (Kobyaschi, 2010). Ionophores are compounds forming lipid-soluble complexes that transport ions across biological membranes. Monensin promotes propionate production in the rumen. Long-term administration of monensin in dairy cattle can reduce CH_4 emissions by 7% without effecting milk productivity (Odongo et al., 2007). Monensin can be delivered as a premix added to the diet, as a slow release capsule inserted into the rumen or, increasingly in pasture-based systems, in the water supply in the paddock using a form of monensin designed for in line water dispenser systems. There are concerns that the inhibitory effects of ionophores on methanogenesis do not persist over time, as discussed by Grainger et al. (2008).

6.2.3.5 Vaccines

Goopy (2019) concluded that there is very little actual evidence for the efficacy of methanogen vaccines on the production of enteric methane in vivo. Also regulations restrict the use of this mitigation option.

6.2.3.6 Enzyme inhibitors

Compounds that directly target methanogenesis in the rumen such as bromoethane sulphonate (BES) and bromopropane sulphonate (BPS) have been tested. These compounds are specific inhibitors of methyl-coenzyme M reductase (MCR), an enzyme that is involved in methane formation of methanogenic archaea. However, BES has been considered as a toxic substance (Grawert et al. 2014), negating its use as a feed additive.

Studies of nitro-compounds for reducing ruminal CH_4 emissions led to the study of 3-nitro-oxypropanol (3-NOP) which specifically inhibits MCR (Duval and Kindermann 2012). Hristov et al. (2015) tested the effect of 3-NOP on enteric methane emission in lactating Holstein cows. Feed intake, milk production, and fibre digestibility were not affected by the inhibitor. Milk protein and lactose yields were increased by 3-NOP. Rumen methane emissions were decreased by 3-NOP, averaging about 30% lower than the control. Methane emission per unit of feed dry matter intake or per unit milk were also about 30% less for the 3-NOP-treated cows. On average, the body weight gain of 3-NOP-treated cows was 80% greater than control cows during the 12-wk experiment.

The inhibitory effect persisted over 12 weeks of treatment. 3-NOP given to high-producing dairy cows at 60 mg/kg feed dry matter not only decreased methane emissions by 30% but also increased body weight gain significantly without negatively affecting feed intake or milk production and composition. In sheep, 3-NOP at 0.5 g/day also decreased CH_4 production by 29% without adverse effects on digestion or rumen fermentation.

Jayanegara et al. (2018) did a meta-analysis of the use of 3-NOP as feed additive based on 12 *in vivo* studies with dairy cattle , beef cattle and sheep. Diets provided for ruminants were total mixed ration (TMR) with either high forage or high concentrate diets. Concentrations of 3-NOP in diets varied from 0 to 280 mg/kg dry matter intake (DMI). The effect on methane decrease was linear within the range of 3-NOP between 0 and 280 mg/kg DMI. Use of 3-NOP at 100 mg/kg DMI, for instance, was estimated to decrease methane emissions by about 20% compared with the control diet.

Dijkstra et al. (2018) carried out a meta data analysis of 11 experiments and 38 treatment regimes *in vivo* using 3-NOP. The average 3-NOP dose in beef cattle experiments was higher (144 mg/kg of DM) than that in dairy cattle (81 mg/kg of DM). The CH_4-mitigating effect of 3-NOP was 22.2 ± 3.33% in beef cattle compared with 39.0 ± 5.40% for dairy cattle based on CH_4 production. The greater efficacy of 3-NOP in decreasing CH_4 emissions in dairy cattle compared with beef cattle may be associated with the higher feed intake level in dairy cattle.

In summary, 3-NOP appears to be the most effective additive reducing emissions in dairy cattle by about 20-40%, about 20% in beef cattle and about 30% in sheep. Addition of oils and lipids to diets could reduce emissions by 5-7.5%. Tannins have a variable effect.

The practicality of 3-NOP use is unknown, although its use may be approved in Europe soon. Caution must be exercised when feeding nitrate in combination with other high nitrogen inputs. Although proven to be effective in reducing CH_4 emissions, these feed additive strategies disrupt the natural rumen function and their misuse could lead to rumen disorders and potential health and other welfare problems (Llonch et al. 2017). There is a general requirement to understand the impacts of these strategies on animal health and greenhouse gas emissions (Williams et al. 2015).

6.3 Sector Analysis

6.3.1 Dairy

O'Brien et al. (2014) found that in ruminant production, there is a strong relationship between productivity and emission intensity – up to a relatively high level of productivity, emission intensity decreases as yield increases. Gerber et al. (2011) found that high-yielding animals producing more milk per lactation generally exhibit lower emission intensities for three main reasons.

- Emissions are spread over more units of milk, thus diluting emissions relative to the maintenance requirements of the animals.
- Productivity gains are often achieved through improved practices and technologies which also contribute to emissions reduction. For example, high quality feed (diets high in concentrate (starch) and lower in fibre) and high performance animal genetics are important.
- Productivity gains are generally achieved through good herd management, animal health and husbandry practices which increase the proportion of resources utilised for productive purposes rather than simply being used to maintain the animals.

This results in a reduced cow mass (both in lactating and in replacement herds) per unit of milk produced. The impact per unit of milk is therefore reduced for the individual cow. At a herd level of course the farmer with a high productivity herd may just increase herd size to fully utilise the area of land that is farmed. This would increase output and the overall GHG emissions - although high stocking rates would be limited by the Water Resources (Control of Agricultural Pollution) Regulations 2021.

The effect of increasing the average number of lactations per cow in the national herd is to reduce the number of cows and followers required to maintain milk output levels (Wall et al, 2009). The main reduction in CH_4 and N_2O emissions comes from the reduction in numbers of followers to keep the production herd, with approximately 19% reduction in GHG emissions in followers compared to 0.5% reduction from lactating cows.

The time required to increase the average parity (number of pregnancies) via genetic selection of the national herd depends on selection intensity and genetic variation. At current values for the heritability and relative economic value of lifespan (and correlated traits) used in that UK national dairy selection index, profitable lifetime index (PLI), the expected annual response is approximately 0.045 lactations per year (Wall et al., 2009). Therefore, it would take 11 years to increase lifespan by 0.5 lactations using currently available selection tools and index weights.

Wall et al., (2009) modelled that 21% less CH_4 emissions per kg of milk production would be possible after selecting a cow herd having 17% higher yield per lactation, and 14% higher gross efficiency.

Conversely, the focus on increasing lactations per cow and higher gross efficiency for milk production has the effect of reducing the number of dairy replacements and reducing the number of cull cows produced, which go to produce beef. Replacing the cull cow beef with beef from suckler cows would increase GHG emissions for beef consumed. This has been shown by Zehetmeier et al. (2012) who have challenged the narrow focus on milk production since a dairy cow production unit produces not only milk output from the dairy cow, but also beef output from culled cows and the fattening system for surplus calves. Two scenarios were modelled:

- Scenario 1 - constant milk production at the farm level by increasing cow productivity, with decreasing beef production; and
- Scenario 2 - both milk and beef production kept constant by compensating the decline in beef production from the dairy cows with beef from suckler cow production.

Scenario 1 resulted in lower GHG emissions with increasing milk yield per cow. However, when milk and beef outputs were kept constant (Scenario 2), GHG emissions remained approximately constant with increasing milk yield from 6000 to 8000 kg/cow per year. GHG emissions per kg milk decreased with increasing milk yield per cow per year, from 1.06 kg CO_2e) to 0.89 kg CO_2 e for the 6000 and 10 000 kg yielding dairy cow, respectively. However, emissions per kg of beef increased from 10.75 kg CO_2e to 16.24 kg CO_2e due to the inclusion of suckler cows. This study shows that the GHG emissions of increasing milk yield per cow in dairy farming differ, depending upon the system boundaries that are considered, particularly the assessment of the value of co-products and the assumed ratio of milk to beef demand.

In seeking genetic improvement for productivity, it is also essential not to breed animals which lead to poor health and less fertility, since these factors are also responsible for higher emissions per unit of product (Garnsworthy, 2004). Poor fertility requires a large number of animals in herd size to meet demand and hence more GHG emissions. While breeding has resulted in increases in milk yield per cow year-on-year, fertility has decreased. Garnsworthy (2004) estimated the impact of fertility on GHG emissions, through the construction of a model, which linked changes in fertility to herd structure, number of replacements, milk yield and nutrient requirements to GHG emissions. Replacements of followers contributed up to 27% of the methane and 15% of the ammonia attributed to dairy cows in the UK. Improving fertility would lead to decreased numbers of replacements required, with a consequent significant decrease in GHG emissions from the dairy herd.

To meet the challenge to health and fertility posed by increasing production there is increasing use of crossbred stock as opposed to pedigree animals. Scandinavian Red cattle have become one of the preferred breeds. The Scandinavian Red/Holstein crossbreds were not significantly different from the pure Holstein for milk fat plus protein production (Heins et al. 2006)

More productive animals could reduce CH_4 emissions by 10–24% in the case of dairy cattle (Grossi et al., 2019). However, pressure on increasing the productivity of animals may lead to a negative impact on animal health due to increased metabolic demand (Llonch et al., 2017). It should be noted that dairy cows must breed to lactate. A reduction in total livestock numbers can only be achieved with improved fertility in dairy cows if a greater proportion of the dairy-bred calves can replace beef-cow calves, i.e. through the use of a beef bull, or by the greater use of sexed semen such that heifer calves are the dairy breed and male calves are beef.

The study of Weiske et al. (2006) showed that optimising the lifetime efficiency of dairy cows, by reducing the replacement rate and exporting surplus heifers from the system as newborns, would reduce GHG emissions by up to 13%.

Knapp et al. (2014) concluded that significant reductions of 9 to 19% CH_4/unit milk can be achieved by combinations of genetic and management approaches, including improvements in heat abatement, disease and fertility management, performance-enhancing technologies, and facility design to increase feed efficiency and life-time productivity of individual animals and herds.

There is debate about the best way to maximize milk yield per hectare and profit. One option is by offering cows nutritionally precise diets in confinement (Capper et al., 2009). The second option used in some countries, notably Ireland and New Zealand, aims to minimize production costs through maximizing the proportion of grazed grass in the diet of lactating cows. O'Brien et al. (2010) used an integrated economic GHG farm model to evaluate 9 milk production systems. Three divergent lines of Holstein Friesian cows (i.e., high production North American, high durability North American, New Zealand) were examined in three feeding systems based on high grass allowance, high stocking rate and high concentrate supplementation. The most profitable combination was the New Zealand high stocking rate system. This combination also resulted in a 12% reduction in GHG emissions from the farm compared with the New Zealand high concentrate system, which had the highest emissions. The reduction in GHG emissions of the New Zealand high stocking rate system was due to the lower number of non-productive animals resulting from the higher herd fertility of this group.

High standards of management play a critical role in reducing methane emissions. Top-performing herds of both systems have carbon footprints 27 to 32% lower than average dairy systems (Capper et al., 2009). As already described, the lower carbon footprint of milk from high-performance dairy systems can be explained by their greater productive efficiency, which potentially reduces resource use per unit of milk, thereby reducing the carbon footprint. Furthermore the relative difference between average and high performance dairy systems was greater than the relative difference between top-performing grass and confinement dairy systems.

Another aspect of livestock management is the housing of cattle particularly in the 5-6 winter months. Where animals are housed, air is often extracted, and there is thus an opportunity for mitigation of methane emissions. In Europe some facilities housing cattle also have air extraction systems installed. In the 2010 report to Welsh Government, there was a recommendation to examine the potential for methane scrubbing from continuously housed cattle. This would be applied for cattle when winter housed. Fedrizzi et al. (2018) used biofilters to capture methane from cow barns. The biofilters based on commercial compost and straw achieved almost 100% oxidation of methane. If such a rate could be transposed to real field conditions in Canada, home to nearly 945,000 dairy cows, biofiltration could be applied to efficiently abate between 2-3Mt of CO_2 equivalent annually in the UK. For Wales this would be 0.53-0.80 Mt of CO_2 equivalent annually. A significant hurdle requiring research is the building design to capture the low concentration of methane from the buildings. In Scotland SRUC is researching this aspect.

If cattle are housed for long periods Haskell et al. (2006) indicated that this can have a detrimental effect on foot and leg health, when comparing zero-grazing systems in housed systems with grazing systems. However, good free-stall and floor design may reduce lameness and leg lesions. There are clearly animal welfare concerns that would need to be addressed.

In summary, reductions of methane emissions may be achievable through the combination of changes to livestock management practices and genetics. Improved management of dairy cattle management approaches include

1. **Feeding non-structural or starchy carbohydrates with high quality forages – (high digestibility) with higher intake levels, optimally processing the forages. These actions may reduce enteric CH_4 emissions/unit milk by 2.5 to 15%. (Knapp et al. 2014)**

2. **Maximise production by livestock management and genetic improvement reducing emissions by 9-19% for dairy cattle, for example by reducing age at first calving and culling rate frequency to reduce the number of herd replacements reared (Wall et al. 2009).**
3. **Use 3-NOP inhibitor to reduce methane emissions by 20-40%. Monensin use has already been researched and could also reduce emissions by 7%.**

It is difficult to assess how much these reductions are additive, particularly the use of 3-NOP on cattle where methane generation is reduced by genetic improvement. Conservatively taking 3-NOP reduction to be 20% and averaging the potential reduction from diet and livestock management, overall emissions reduction could be up to 43% This is similar to the assessment by Moxey and Thomson (2021) who estimate potential emission reductions from the dairy herd in Scotland to be about 45%.

6.3.2 Beef

In the case of meat-producing animals, with improved efficiency, slaughter weight can be achieved at a young age and a shorter lifetime will reduce CH_4 emissions per animal (Lovett & O'Mara, 2002). Increasing the efficiency of meat production decreases emissions per unit of yield. GHGs emission can be reduced up to 27% (CO_2e/ha) by a combination of the improved herd, animal and pasture efficiencies (Beukes et al., 2010).

A significant breed difference has been found, suggesting that bigger continental breeds of cattle produce about 7% fewer emissions per unit output than the smaller British breeds (Samsonstuen et al, 2019).

O’Brien et al. (2020) studied the Life Cycle Analysis (LCA) of beef systems in 4 countries including Ireland which may best replicate growing conditions in Wales. Enteric methane emissions are one major component of the LCA. Examples of reliable options that can be simultaneously implemented to substantially reduce beef carbon footprint include improving animal productivity, and increasing forage quality. Specific measures and the potential reduction in net carbon footprint are shown in Table 6.1. There is the caveat that these reductions are subject to uncertainty and are not necessarily additive.

Table 6.1. Effect of Management Options on LCA for beef production.

Mitigation Measure	Net Carbon Footprint reduction (%)
Animal Performance	
Increase daily weight gain	3-10
Reduce slaughter age	5-10
Improve animal health	5-15
Optimise first calving age – 24 months	5-10
Improve genetic merit	2-10
Diet	
Improve grassland management	3-10
Improve forage quality	3-8
Increase concentrate fraction	Plus 15 to minus 20
Feed additives	Plus 15 to minus 5
Replace soya meal with low emission alternatives	3-15

LCA of beef production systems have been conducted (Beauchemin et al., 2011), with the consensus that the suckler cow - calf phase contributes the largest proportion of total GHG emissions from beef production (i.e., 60–84%). In contrast, growing and finishing market cattle accounts for a relatively small fraction of the total GHG budget for beef production. Lower CH_4 emissions from the transition from store to fat cattle are due to the relatively brief duration that they are fed and, to a lesser extent, the use of grain based finishing rations (Beauchemin et al., 2010). Studies have examined effects of finishing systems for growing beef cattle and most conclude that GHG emissions from grain finished beef are less than those from forage finished beef (Pelletier et al., 2010). Diets with higher nutritional quality reduce enteric CH_4 emissions/kg DM intake, increase growth rates, reduce days to market and total GHG emissions/kg of beef produced.

Consistent with these findings, in Scotland there has been a particular emphasis is on reducing emissions from the suckler beef herd. Under the aegis of the Scottish Government's Suckler Beef Climate Group, options for reducing emissions at a farm scale have been identified and estimated (Bell et al. 2021). The main measures providing the greatest impact were modelled. The potential reductions were identified – see Table 6.2

Table 6.2 GHG Emissions of Beef Production in Scotland

Intervention	% Emission change from baseline (kg CO2e/kg deadweight)	Cumulative Change from baseline (kg CO_2 e/kg deadweight)
Increase calves reared	- 1.4%	-1.4%
Reduce age of 1st calving from 3 to 2 years	- 6.9%	- 8.2%
Reduce cow weight by 10%	- 1.1%	-9.2%
Reduce age at slaughter to 18 months	- 12.5%	-20.5%
Improved grassland management – rotational grazing, reseeding, better quality silage	- 7.9%	-26.8%
Methane inhibitor – 3 NOP	- 4.8%	-30.3%
Improved nutrient management	- 3.9%	-33.0%
Nitrification inhibitors in artificial fertilisers	- 6.2%	-37.2%
Reduce age at slaughter to 16 months	+ 1.4%	-36.3%
Finish as bull beef at 13 months	- 2.0	-37.6%

Note: Each intervention would not be additive totally because there are trade-offs between interventions.

Although there will some differences in management practices, animal breeds, land quality and climate between Scotland and Wales, the analysis provides a good estimate of the potential emission savings from the beef sector in Wales. Excluding improved nutrient management and nitrification inhibitors (see Chapter 5) the emission reduction could be a maximum of 30%. The livestock management interventions would lead to a significant reduction (up to 21%) in cattle numbers in Scotland whilst delivering the estimated emission reductions. If the interventions led to increasing production from an increase in animal numbers, that would negate the emission reductions.

In summary for beef cattle system, the livestock management, genetics and diet interventions may reduce emissions by about 30%, excluding the measures for manures and fertilisers. 3-NOP would be difficult to administer apart from its use for housed animals.

6.3.3 Sheep

The effect of high performance management on CH_4 emissions is not limited to cattle. Jones et al. (2014) estimated the cradle to farm-gate carbon footprints of 64 sheep farms across England and Wales using empirical farm data. Mean carbon footprints of 10.85, 12.85 and

17.86 kg CO_2e /kg live weight finished lamb were recorded for lowland, upland and hill farms respectively. Enteric methane emissions averaged 42-48% of total emissions for the three farm types. Multiple linear regression models indicated that four farm management variables had a significant impact on the size of the carbon footprint of finished lamb. Irrespective of farm category, these were

- the number of lambs reared per ewe (head/ewe),
- lamb growth rate (g/day),
- the percentage of ewe and replacement ewe lamb flock not mated (%),
- concentrate use (kg/livestock unit).

Of these, the number of lambs reared per ewe mated and lamb growth rate were the most influential. Productivity improvements are arguably most problematic for extensive hill farms but the top performing hill farms in this study outperformed the mean lowland and upland farms. The results suggest that at a national level, the emphasis for reducing the carbon footprint of lamb should be on closing the productivity gap between poor and top performing farms.

The productivity indicators showed that best performing farms were around two or more times as productive as the worst farms. For example, the number of lambs reared per ewe varied between 0.7 and 1.8; and lamb growth rate between 57 g/day and 356 g/day, demonstrating considerable potential for improvement on the poorest performing farms. Improved lamb output per ewe can be achieved through a range of interventions including selective breeding for ewe productivity, increased lamb survival through better hygiene and management at birth, and nutritional management.

For selection for growth rates in maternal lines, AHDB programmes help in advising farmers on genetic indices e.g. Estimated Breeding Value (EBV) for selection of breeding stock based on traits for production factors, health and management. For example, for farms in the sheep improvement programme, the increase in growth rates for lambs is claimed to be 50% over the period 2000-2014. (AHDB, 2015). In Wales, the Hill Ram Scheme is evaluating the use of performance recorded (EBV) rams against unrecorded rams, preliminary results suggest lamb finishing weight is increased and time to finish is reduced.

At an individual farm level, it is critical that the emission reductions of finishing lambs earlier is not negated by either a corresponding increase in emissions from additional feed, or an increase in fertiliser requirements to improve pasture productivity

In summary, significant emission reductions could be obtained by reducing the variability of farm performance within farm type (lowland, upland, mountain). This can be achieved by improving the number of lambs reared per ewe and lamb growth rate by 30% (Jones et al. 2014) to the performance levels of higher performing farms. Genetic improvement would also facilitate this reduction. 3-NOP will be difficult to administer to grazing sheep.

Emissions would be reduced by 30% by the reduction of the national breeding flock from 4.5 million to 3.15 million whilst maintaining output of around 4 million lambs/yr. Ewes emit around 75% of total flock enteric emissions, so a 30% reduction in ewe numbers takes emissions to 515 ktCO_2e/yr. Lamb emissions remain the same ~250 ktCO_2e/yr. The total reduction would be 215 ktCO_2e/yr.

6.4 Summary

Many of the studies reviewed here focus on dairy cattle, where there is good control of diets to maximise production, with considerable effort to improve genetics. For beef store cattle and sheep, the diet is more extensive and centres almost wholly on grass systems particularly in the uplands. Options to use diet additives, cereals and concentrates are facilitated by dairy systems, but are more difficult to implement for beef and sheep systems. Table 6.3 summarises the emission reduction potential for the three main sectors.

Table 6.3 Emission Reduction Potential

	2018 Emissions	**Potential percentage reduction**	**Potential Emissions from Enteric Methane reduction**
Sector	**(kt CO_2 Equivalent)**		**(kt CO_2 Equivalent)**
Dairy	792	Diet, management, genetics and 3-NOP (43%*)	341
Beef	1188	Diet, management, genetics (30%)	356
Sheep	981	Diet, management and genetics (30%)	215
Total	**2961**		**912 (31% reduction)**

** average value, very dependent on use of 3-NOP*

These estimates are of the maximum technical potential. They do not include manure and fertiliser management reductions covered in Chapter 5. There would be a small additional reduction in emissions from manures if livestock numbers decrease, particularly in the dairy herd. Practically the extent of implementation by Welsh farmers will be variable and sector dependent. The time frame for genetic improvements is long but continuous.

In inventory calculations, any reduction in emissions would result from a reduction in numbers of ruminants, through better management and improved genetics to achieve the same level of production. Improvements in diets may be reflected in better animal health, and better survivability of stock. Incorporation of dietary additives would not be captured by the inventory unless specific Emission Factors and the associated activity data are generated to account for these practices.

7 Economic and practical assessment

7.1 Economic Assessment

Marginal abatement cost curves (MACC's) show the relative cost of greenhouse gas mitigation by alternative mitigation methods and technologies (Macleod et al. 2010). A MACC ranks abatement measures in order of cost effectiveness. A typical MACC is shown in Figure 7.1 which shows the cost of saving an emission of carbon dioxide equivalent (CO_2 e) on the y axis in £/t CO_2 e. Measures to the left of the curve and below the x-axis indicate negative costs or savings whilst positive results to the right and above the x-axis show that there will be an economic cost to achieve the emission reduction. The width of the columns indicates the potential emission reduction.

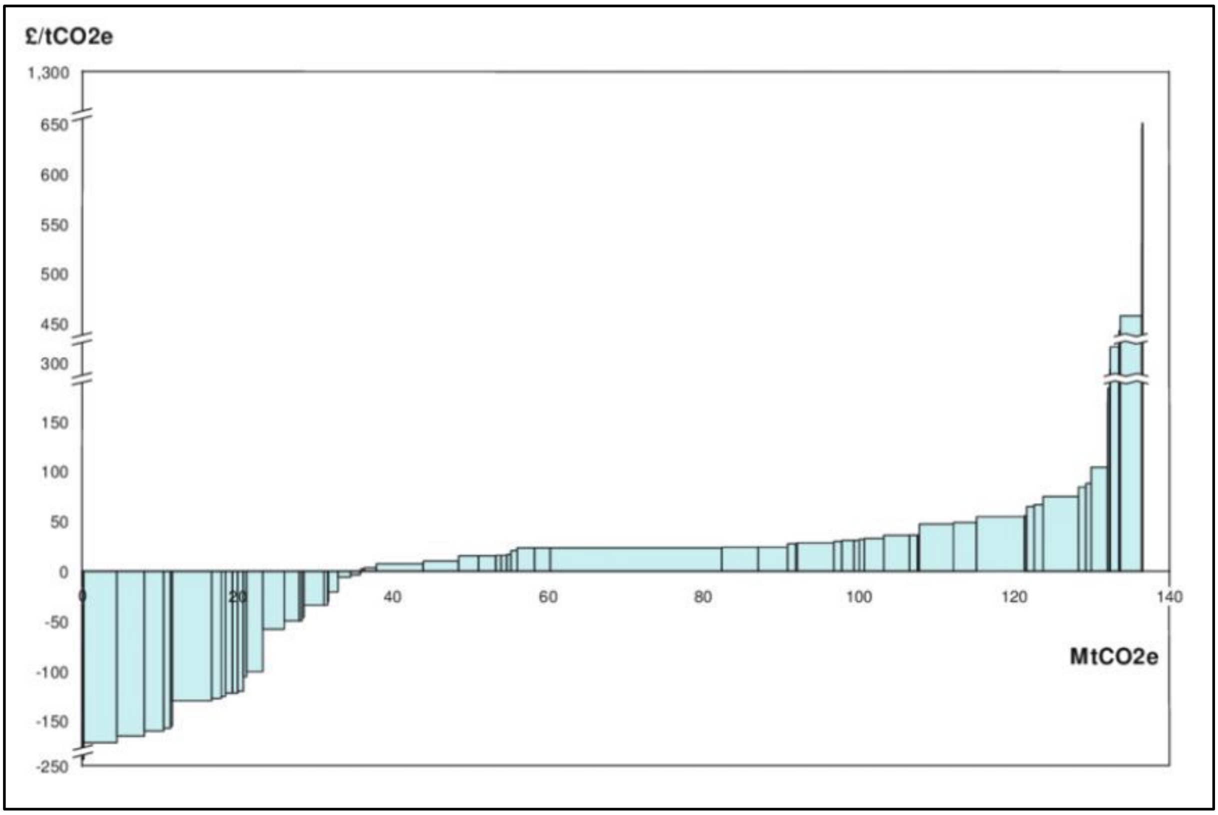

Figure 7.1 MACC for a range of mitigation options – Cost on y axis and mitigation option on x axis (Moran et al., 2008)

Table 7.1 shows the MACC table to target the options in terms of mitigation potential – mainly high and medium in England, and the cost effectiveness – the negative cost options are shown in red (Moran et al. 2020).

As previously concluded the main groups of options are

- **Improved crop management to make effective use of manures and fertilisers by precision farming, with improved crop varieties, high sugar grasses and legumes to improve biological nitrogen fixation.**
- **Improved diet, livestock management and genetics to improve productivity per animal and to reduce methane emissions.**
- **Improved manure management**

- **The use of 3-NOP for suppressing methane emissions and nitrogen inhibitors with fertilisers to suppress nitrous oxide emissions from soils.**

Table 7.1 MACC assessment for mitigation options for England.

No	Measure	Category	Mitigation Potential	Cost Effectiveness £/tCO2e
1	Improved crop varieties	Cropland	Medium	-1248
2	Catch/cover crops	Cropland	High	150
3	pH at optimum - liming	Cropland	High	-28
4	Agroforestry	Cropland	High	1793
6	Agrivoltaic solar arrays	Cropland	High	18
7	Crop Health	Cropland	Low	-2863
8	Integrating grass/herbal leys	Cropland	Medium	-28
10	Precision farming	Cropland	High	-1043
11	Avoiding N excess	Cropland	Medium	-1683
12	N inhibitors fertilisers	Cropland	High	667
15	Analyse manure prior to application	Cropland	Medium	-2127
16	Land drainage on mineral soils	Cropland	Medium	1256
17	Reduce soil compaction	Cropland	High	-21
20	Biological fixation	Grassland	High	-1770
21	High sugar grasses	Grassland	Medium	-801
26	Animal Breeding - genomics	Livestock	High	-784
28	Genetic selection to reduce methane	Livestock	Baseline	-1620
29	Current breeding practices	Livestock	High	
30	Health planning - cattle	Livestock	High	-88
31	High starch diet	Livestock	Low	0
32	Precision feeding	Livestock	Medium	58
35	3-NOP feed additive	Livestock	Medium	113
37	Increased milking frequency	Livestock	Medium	-1882
38	Dual purpose cattle breeds	Livestock	High	-10
45	Nitrate additives	Livestock	High	84
48	Sheep health planning	Livestock	Medium	16
22	Anaerobic Digestion	Livestock	High	-186
23	Methane capture	Livestock	Medium	1695
25	Permeable slurry cover	Livestock	Low	36931
47	Impermeable slurry cover	Livestock	High	59
39	Restoring organic soils	Cropland	High	0

It may be more appropriate to consider the results of MACC analysis for Scotland, although the extent of arable farming in Scotland is more significant than in Wales (Eory V et al, 2020).

Key findings were

- The mitigation measures applicable to agricultural land can save between 7 and 553 kg CO_2e every year on each hectare where they are applied.
- The single most effective measure is increased cultivation of grain legumes (i.e. peas and beans) which provides 553 kg CO_2e per hectare savings annually but at high cost because the value of a legume crop is much less than for a cereal crop.

- The second and third most effective measures (on an area basis) are variable rate nitrogen and lime application (precision farming) and soil pH management (i.e. liming when necessary), delivering 151 and 112 kg CO_2e mitigation per hectare annually, respectively.
- Intercropping can provide the highest cost savings to farmers per hectare per year (£45); variable rate nitrogen and lime application, crop varieties with higher nitrogen use efficiency and soil pH management can also provide savings. Grain legume cultivation is the most expensive option (£406 per hectare per year).
- The cattle mitigation measures assessed can save between 57 and 854 kg CO_2 e every year for each animal they are applied to 3-NOP feed additive, breeding for low methane emissions and slurry store cover with impermeable cover are the most effective.
- Cattle measures' net costs range from a saving of £359 to a cost of £31 per animal per year. The dairy breeding measure could save £359 per animal per year, and improved health of dairy animals, dairy precision feeding, beef breeding for low methane emissions and covering beef slurry stores can also save farmers money. The most expensive cattle measure is administering 3NOP feed additive to beef animals (£31 per animal per year).
- The sheep measures investigated can deliver 15 kg CO_2e mitigation per animal annually and a cost saving of £0.36 per head.

Table 7.2 presents the mitigation savings and the total cost per acre of crop or per livestock head in Scotland.

Table 7.2 Mitigation potential and costs of measures for agriculture in Scotland

Mitigation Measure	Annual total mitigation (t CO2 e /ha or t CO2 e/head)	Annual total cost £/ha or £/head)
Tillage and grassland		
Grain legumes in rotation – lower value legume crop	0.533	406.00
Variable rate nitrogen and lime application	0.151	-16.83
Soil pH management	0.122	-7.86
Intercropping e.g. legume and cereal	0.079	-45.18
Nitrification and urease inhibitors e.g. DCD	0.071	20.67
Crop varieties with higher N use efficiency	0.13	-10.17
Slurry injection	0.026	21.35
Trailing hose/shoe slurry application	0.007	8.16
Dairy Production		
3-NOP feed additive	0.855	17.78
Breeding for low methane production	0.627	-358.74

Mitigation Measure	Annual total mitigation (t CO2 e /ha or t CO2 e/head)	Annual total cost £/ha or £/head)
Covering slurry stores – impermeable covers	0.527	2.56
High starch diet	0.162	0
Precision feeding	0.104	-18.22
Improved livestock health	0.057	-26.89
Beef Production		
3-NOP feed additive	0.423	31.38
Covering slurry stores – impermeable covers	0.255	-0.25
Breeding for low methane production	0.116	-15.96
Improved livestock health	0.027	20.26
Sheep Production		
Improved livestock health	0.015	-0.36

Negative costs would deliver cost savings for farmers. Improvements to livestock and fertiliser management are the consistent ways of reducing GHG emissions. However it must be recognised that the scope for reducing emissions from arable land in Wales is less (with 90,000ha) than in Scotland (arable area of 580,000ha).

The main issue in interpreting these data is the ease of implementation and timescale. Growing more grain legumes in rotation has large costs, making it less likely to be adopted. Intercropping also requires a change of farming practice because both crops will need to be harvested together for forage – sacrificing the use of grain for other applications. Breeding for low methane emissions is cost negative for dairy cows but is a long term objective. Use of 3-NOP for dairy and beef cattle has a cost but would be more easily implemented – assuming regulatory approval.

MACC research has continued to be used by UKCCC and the latest work has modelled scenarios for the rate of implementation of mitigation measures (Eory et al. 2020). Changes in consumer and farmer behaviour can release land from agriculture. The UKCCC considered two broad groups of changes that could release land, while maintaining a strong food production sector to feed a population that increases in the UK to 73.6 million by 2050:

- Behavioural change: diet change and food waste reduction (considering population growth and constant trade).
- Improvements in agricultural practices: crop yield improvements, stocking densities and moving horticultural production indoors.

The scenarios included reductions of up to 50% in meat and dairy products with a switch to plant based products and lab-grown meat. Dairy productivity could rise up to 2.9%/year initially and grazing intensity could rise by up to 10% with a decrease of livestock on uplands. **The scale of changes would have substantial implications for Welsh agriculture. A commentary on the measures advocated by UKCCC in relation to this report is in Annex 1.**

In summary, the MACC analyses in England and Scotland, provide emission reduction savings and costs which could be used to devise support mechanisms for a new farming scheme to deliver climate change mitigation. Robust measures require MACC analysis for Welsh agriculture. It must be remembered that these are maximum technical potentials assuming 100% uptake. Practical feasibility is a critical issue to achieve progress.

7.2 Practical Feasibility

Barnes et al. (2021) have used a workshop approach to investigate areas of behaviours and drivers of uptake of measures that lead to GHG mitigation. The study focused on English farming, and may not be wholly representative of the situation in Wales. Important factors for uptake of measures were identified to be:

- The self-identity of the farmer and social norms
- Management of risk
- Acquisition of knowledge
- Long-term planning constraints due to lock-in effects of assets and practices
- The role that current practices have on determining how new ones fit
- Family life cycle factors
- The entry of newer farmers into the industry
- Institutional factors: the influencer of practice, supply chain constraints and informal institutions

Table 7.3 SRUC – NIAB-UKCEH analysis of interventions for adoption of mitigation measures

Farm Practice	Intervention medium target adoption	Intervention high adoption target	On farm technical cost	On-farm investment cost	Feasibility of doing	Ease of monitoring	Risk seen by farmers	Applicability	Adoption level in 2020	Monitored in GHG inventory
Keeping pH at optimum level	Fiscal	Regulatory	Savings	Medium	Easy	Medium	Low	Medium - grassland	Very low	Via N fertiliser use
Agroforestry	Fiscal	Fiscal	High	Medium	Medium	Easy	High	Very low	Very low	Via land use change
Agrivoltaic systems	Fiscal	Fiscal	High	High	Difficult	Easy	Medium	Very low	Very low	Via energy stats
Nitrification inhibitors	Fiscal	Regulatory	Medium	None	Medium	Medium	Medium	Medium	Very low	Via N fertiliser use
Reducing soil compaction	Fiscal	Regulatory	Savings	Medium	Easy	Medium	Medium	Medium	Low	No
Grass-legume mixes	NA	Regulatory	Savings	Medium	Easy	Easy	Low	Very high	Low	No
Restoring organic soils	Fiscal	Regulatory	High	High	Medium	Easy	High	Medium	Very low	Via land use change
AD livestock manures	Market	Market	Savings	High	Medium	Easy	High	Low	Very low	Via energy stats
Breeding with genomics	Fiscal	Fiscal	Savings	Medium	Medium	Easy	Medium	High - dairy Low - beef	Very low	Via livestock numbers
Improving health status of cattle	None	Regulatory	Savings	None	Easy	Difficult	Low	High	Very low	Via livestock numbers
Dual purpose breeds	None	None	Savings	None	Medium	Easy	High	High	Very low	Via livestock numbers
Nitrate additives to suppress enteric methane	Regulatory	NA	Medium	None	Medium	Medium	High	Medium - dairy	Very low	Need activity data and new emission factors

Aspects that affect adoption include the amount of system change needed, the scale of the operation, the opportunities for intergenerational transfer, and personal factors.

From the final scenarios, interventions were analysed – see Table 7.3. One striking aspect is that few interventions were seen to be low risk and adoption levels in 2050 were seen to be low to very low. **There is a need to assess how these options are viewed by farmers in Wales, as some assessments may differ from the workshops used in England.**

There is an important caveat in the trend for increases in productivity on farms. The natural tendency of farmers is to use all of their land for production. Unless their production levels on their farms are limited, they will increase total production which negates any benefit for reducing GHG emissions per unit production.

The land spared from production could be used for delivering public goods e.g. improving biodiversity, increasing carbon sequestration or flood mitigation. Balmford et al. (2018) confirmed that high-yield farming has the potential for land not needed for production to be used for carbon sequestration, to make a substantial contribution to mitigating climate change. Farmers will need advice and incentives to use any land released for public goods.

To understand the slow rate of change, DEFRA is monitoring its Agriculture Action Plan (DEFRA 2020). An indicator framework has been designed consisting of ten key indicators covering farmer attitudes and knowledge, intermediate outcomes relating to GHG emission intensity of production in key agricultural sectors and the uptake of mitigation methods. For some indicators (such as farmer attitudes) there are limited data currently available to assess long term trends and in the short term there is little change. Where longer term data are available, a current assessment shows the overall picture to be mixed. Over the last 10 years there has been a positive long term trend for the soil nitrogen balance (a high level indicator of environmental pressure) and for the derived manufactured nitrogen use efficiency for barley, oilseed rape and sugar beet. For intermediate outcomes relating to GHG emission intensity for the livestock sector there has been little overall change.

This situation is reflected in farmer attitudes in England in 2020, and may be similar in Wales. 18% of farmers reported that it was "very important" to consider GHG emissions when making decisions relating to their land, crops and livestock and a further 46% thought it "fairly important". However 30% of respondents placed little or no importance on considering GHGs when making decisions or thought their farm did not produce GHG emissions.

The England Action Plan has the target to progress towards achieving the industry's ambition to reduce agricultural production emissions by 3 M tCO_2e by 2020 compared to a 2007 baseline. However by February 2020, approximately 0.9 Mt CO_2e reduction in GHG emissions had been achieved from the uptake of the key mitigation methods. This compares with an estimated maximum technical potential reduction of 2.8 Mt CO_2e if all of these methods were fully implemented on relevant farms. – i.e. a 32% reduction compared with the target.

There is a substantial training programme required, with a range of regulatory and financial incentives. Access to practical advice through Farming Connect and the Levy Boards is essential to increase uptake of mitigation measures. For example AHDB and HCC programmes help in advising farmers on genetic indices e.g. Estimated Breeding Value for selection of breeding stock based on traits for production factors, health and management.

8 Conclusions

8.1 Mitigation options

This review has focused on five main mitigation options for on-farm emission reduction for ruminants and crops. The review has not considered emissions from the other main sectors – pigs, poultry and horticulture. Also it has not covered emission reduction post-farm gate, although there is an argument that the whole food chain should be considered as one coherent system. The on-farm options focused on ruminant livestock and crops accounting for eating local, unprocessed and unpackaged food for example.

- Reducing methane emissions from ruminants
- Reducing nitrous oxide emissions from the application of manures and fertilisers to soils
- Reducing methane and nitrous oxide emissions from manure storage, handling and application
- Reducing carbon dioxide emissions by maintaining carbon stocks in soils and peats,
- Expanding carbon sequestration in vegetation - woodland, hedges etc. and in agricultural soils to offset emissions from other sources.

Although combustion emissions (CO_2 from tractors and machinery are 9.8% of total agricultural emissions, the main mitigation options are still in the development phase. These include replacing diesel fuel with hydrogen (JCB), electric motors (John Deere), methane fuel (New Holland) and fuel cells. Even after introduction the reduction of emissions from the existing vehicle/machinery fleet will take some time to occur. But an emissions reduction of 56% has been estimated by UKCCC by 2050 i.e. about 310 Kt CO_2e reduction. No-till practices for arable crops would have a small effect in reducing emissions from machinery.

Since 2010, international research on ways to reduce emissions of methane and nitrous oxide has expanded greatly, but the potential reductions are mainly incremental and need to be combined to make significant inroads to achieve emissions reduction. **Even so the reductions will not get to the net-zero emission target without substantial reduction in animal numbers.**

For some, the answer today is to reduce Wales' agricultural production, but this would not solve the global challenge. It will be necessary to continue to satisfy the demand in food, and reducing the Welsh production would only shift it to countries where food production may have a higher climate footprint.

Organic farming systems have lower GHG emissions, but with the penalty of lower productivity. The need to decrease the dependence on manufactured fertilisers because of the costs and GHG emissions of manufacture, means that organic farming methods may be adopted increasingly, for example through regenerative and agri-ecology principles (Carlile and Garnett, 2021) which aim to tackle the three interconnected challenges of food production, climate change and biodiversity loss, but broadened from the technocratic aims to include changing the social, economic and political aspects of food production and use. In Wales these are promoted through organisations such as the Food, Farming and Countryside Commission.

In Denmark, the Agriculture and Food Council (2020) has set out a vision to be climate neutral by 2050 by scientific developments. The proposals for emission reduction cover all the options that have been described in this report, including the development of vaccines to reduce methane outputs from ruminants, and the capture of methane and energy crops for renewable energy.

8.2 Implementing change

Within the proposed Sustainable Farming Scheme the types of specific advice/support mechanisms that would be important are

- Woodland creation and management
- Peatland conservation particularly lowland peats
- Manure management – covers, methane extractors, slurry injection, acidification
- Improving soil management – soil monitoring of pH and nutrient requirements, and analyses to assess manure N values to meet crop requirements.
- Legume and cover crops management
- Improving livestock management – diets, animal health, genetics

Farmers are increasingly looking at ways to use carbon sequestration in woodland, hedges and soils as a potential way of offsetting emissions. Expansion of woodlands and hedges sequestrates carbon in the biomass and the underlying soil. In Wales, the high proportions of permanent grassland and soils already high in carbon severely limit the potential for carbon sequestration. The essential requirement is to maintain existing carbon stocks in soils and peats to reduce carbon dioxide emissions. **Total offsetting of agricultural emissions by soil carbon sequestration to achieve net-zero emissions is not possible under currently feasible methods.**

The recent research has shown the maximum technical potential to reduce emissions, but the practical delivery of these options by Welsh farmers depends crucially on the economic, regulatory and cultural framework to motivate uptake.

To aid uptake there is the need to engage farmers more directly in their role in emission reduction options. One example would be to look at the way that farmer led groups were established in Scotland to develop advice and proposals to the Scottish Government on how to cut emissions and tackle climate change. Social science research may help to identify ways to change farmer practices. The economic case for mitigation measures should be based on Marginal Abatement Cost Curves (MACC) for Wales.

A substantial training programme is required, with access to practical advice through Farming Connect and the levy bodies, HCC and AHDB, with a range of demonstration farms. Use of local farmer hubs would help to spread good practices.

8.3 Monitoring change

To help farmers to reduce emissions, there is the need for quality assured carbon foot-printing tools. The tools are leading to a large range of footprinting results on a farm. Work is starting on a UK standard method similar to the UK Woodland Code Carbon Calculator to aid comparison of different farm management systems. Martineau et al. (2019) have reviewed the data needs to achieve effective tracking of emission trends using a life cycle analysis approach at a farm level.

To monitor the progress of the policy to reduce GHG emissions from agriculture and land use, it is essential to improve the collection of more detailed data which can be linked to the revised Emission Factors for a range of livestock, manure and fertiliser management. There is a need to investigate the use of other data such as from the applications from the Integrated Administration and Control System (IACS) used to monitor land use in support of payments and animal movements databases. Remote sensing could also help to collect data. Another option would be for farmers to be mandated to provide activity data as a condition of participating in the Sustainable Farming Scheme. These examples need to be developed across the UK.

8.4 Benefits of emission reduction

It is important to recognise the other benefits of reducing GHG emissions. Measures to reduce emissions of nitrous oxide have multiple benefits in improving air and water quality and reducing eutrophication of habitats. Reduction of methane emissions also has a secondary benefit in reducing ozone formation in the lower atmosphere (up to 15km). Ozone itself is a greenhouse gas and causes human health problems as well as damage to vegetation. Enforcement of the Agriculture Pollution Regulations linked to the proposed new National Minimum Standards within the draft Agriculture Bill could help in the more efficient management of manures and fertilisers – critical to reducing nitrous oxide emissions.

9 Annex-1: UK Climate Change Commission 6th Report

The UKCCC report summary is presented below (italics). Comments from a Welsh perspective are made on each aspect, based on the main report.

9.1 Overview of Agriculture

Agricultural emissions were 54.6 MtCO2e in 2018, 10% of UK greenhouse gas emissions (GHGs). Completely decarbonising the agricultural sector is not possible (on current understanding) due to the inherent biological and chemical processes in crop and livestock production. However, there are options to reduce these emissions covering behaviour change, productivity improvements and the take-up of low-carbon farming practices. Our analysis starts with the assumption that land is prioritised for housing and other economic activity and food production before climate objectives. We estimate that sectoral emissions could fall to 39 MtCO2e in 2035, and to 35 MtCO2e by 2050 in the Balanced Pathway.

In Wales there are options to reduce GHG emissions through productivity improvements and take- up of low-carbon farming practices. Much of Wales' housing and economic activity is concentrated in the northern and southern coastal belts and the South Wales valleys. Food production, climate change and reversing biodiversity trends remain the priority in most of Wales.

• *Low-carbon farming practices.*

SRUC assessed the abatement potential from measures to reduce emissions from soils (e.g. grass leys and cover crops), livestock (e.g. diets and breeding) and waste and manure management (e.g. anaerobic digestion). These reduce agricultural emissions by 4 MtCO2e in 2035. This takes account of the interaction with other actions, notably diet change, which reduces the abatement potential of these measures over time.

From the assessment of the Maximum Technical Potential there are options to make significant reductions in emissions along the lines suggested by UKCCC. Human diet change is accepted as a trend which reduces the demand for livestock, but reduction of production levels should not be at the expense of requiring imports.

• *Fossil fuel use in agriculture.*

Currently 18 TWh of fossil fuels are used in agricultural vehicles, buildings and machinery, resulting in emissions of 4.6 MtCO2e. Options to decarbonise fossil fuel use are similar to those in surface transport, off-road machinery in industry and commercial buildings. These cover electrification, biofuels, hydrogen and hybrid vehicles. Our Balanced Pathway assumes biofuels and electrification options are taken-up from the mid-2020s and hydrogen from 2030, reducing emissions to 2 MtCO2e in 2035.

The timetable is very ambitious given that agricultural vehicles with alternative energy sources are still in development, and there will be a legacy effect where the existing vehicle fleet will continue to be used. Agricultural vehicles have an average life of about 20 years and longer for harvest equipment used for short periods.

• *Measures to release land.*

Changes in consumer and farmer behaviour can release land from agriculture while maintaining a strong food production sector. We considered five measures that could release *land covering societal changes and improvements in agricultural productivity. Our analysis implies that these five measures could reduce annual agricultural GHG emissions by 8 MtCO2e by 2035, rising to just over 11 MtCO2e by 2050, with diet change the most significant:*

– Diet change. Our Balanced Pathway involves a 20% shift away from meat and dairy products by 2030, with a further 15% reduction of meat products by 2050. These are substituted with plant-based options. This is within range of the Climate Assembly's recommendations for a 20- 40% reduction in meat and dairy consumption by 2050.18 Our pathway results in a reduction in livestock numbers and grassland area, delivering annual abatement of 7 MtCO2e by 2035, rising to nearly 10 MtCO2e by 2050.

The substantial reduction in meat and dairy consumption will have a substantial impact on Welsh agriculture particularly in the uplands. Much of upland Wales comprises permanent enclosed grassland and extensive grazing on moors and mountains, where reduction of grassland is only going to occur if converted to woodland and forest, or used to promote other public goods e.g. biodiversity, flood mitigation.

– Food waste. We assume food waste is halved across the supply chain by 2030 in line with the Waste and Resources Action Programme's (WRAP) UK Food Waste Reduction Roadmap. This would reduce UK emissions by almost 1 MtCO2e in 2035.

Reduction of food waste is recognised as mitigation route, but was not covered in this report.

– Productivity improvements. There is scope for further abatement from measures to increase agricultural productivity, which in our Balanced Pathway could reduce emissions by 1 MtCO2e in 2035 and 2050. These cover crops and livestock:

- *Improving crop yields without the need for additional inputs such as fertiliser and pesticides can be achieved through improved agronomic practices, technology and innovation while taking account of climate impacts. Our Balanced Pathway assumes that wheat yields increase from an average of 8 tonnes/hectare currently to 11 tonnes/hectare by 2050 (with equivalent increases for other crops).*

Improved productivity in Wales will be mainly restricted to improved and enclosed permanent grassland. Given the small area of arable land, increases in arable productivity will have little impact.

- *Stocking rates for livestock can be increased through improving productivity of grasslands and management practices such as rotational grazing. Evidence suggests there is scope to sustainably increase stocking rates in the UK.*

This is recognised as a way of reducing the number of animals for a given level of production. Currently Welsh regulations limit livestock concentrations to 1.7 livestock units per hectare.

– Moving horticulture indoors.

Shifting 10% of horticulture production indoors under a controlled environment reduces the carbon, nutrient, land and water footprint. Delivering emissions reduction should not be at the expense of increasing food imports that risk 'carbon leakage'. Therefore, both production and consumption of the highest carbon foods need to fall.

Whilst recognising the need to expand horticultural production of fruit and vegetables to reduce the 85% imported into the UK from overseas sources, the horticulture sector is very small in Wales, and will be limited mainly to lowland areas. A substantial training and investment programme will be required to change this position.

9.2 Overview of Land Use and Land Use Change

A transformation is needed in the UK's land while supporting UK farmers. By 2035 our scenarios involve planting of 440,000 hectares of mixed woodland to remove CO2 from the atmosphere as they grow, with a further 260,000 hectares of agricultural land shifting to bioenergy production (including short rotation forestry). This would see UK woodland cover growing from 13% now to 15% by 2035. Peatlands must be restored widely and managed sustainably. Low-carbon farming practices must be adopted widely, while raising farm productivity.

It is recognised that woodland expansion is required in Wales. Woodland cover is about 14% of Wales, and Welsh Government has recently described its ambition to expand woodland by 180,000 hectares, in line with UKCCC recommendations.

The largest contribution is from mass take-up of low-carbon solutions, powered by a major expansion of low-carbon electricity and hydrogen supplies. Changes in the UK's land use are also needed. Lower-carbon choices and efficiency can make a material contribution to meeting the budget. Alongside the nature-based removals, by 2035 the UK should be using bioenergy (largely grown in the UK) with CCS to deliver engineered removals of CO2 at scale.

It is recognised that changes to land use in Wales are required. Bioenergy is an option but care is required to ensure that the life cycle analysis of planting, maintenance and harvesting has a positive mitigation benefit. In addition the amount of land available for expanding arable crops in Wales is relatively small.

10 References

Chapter 2

Eory, V., Topp, K., Rees, B., Leinonen, I., Maire, J., Macleod, M., Sykes, A. and Wall, E. (2020). Marginal abatement cost curve for Scottish agriculture. www.climatexchange.org.uk

Fedrizzi, F., Cabana, H., Ndanga E. and Cobral, A. (2018) Biofiltration of methane from cow barns. Effects of climatic conditions and packing media acclimatization. Waste Management 78, 669-676.

Jones,, A. K., Jones, D.L. and Cross, P. (2014). The carbon footprint of lamb: sources of variation and opportunities for mitigation. Agricultural Systems 123: 97-107.

Moran, D., Eory, V., McVittie,A., Wall, E., Topp, K., McCracken, D. and Haskell, M. (2020) Wider implications of greenhouse gas mitigation measures in English agriculture - Defra AC0226

Welsh Government (2021). Net Zero Wales Carbon Budget 2 (2021-2025). Net Zero Wales Carbon Budget 2 (2021 to 2025) | GOV.WALES

UKCCC Advice Report: The path to a Net Zero Wales. December 2020. www.theccc.org.uk/publications

Chapter 3

AHDB RB209 Section 1 Principles of nutrient management and fertiliser use.

Allen, M. R., Shine,, K., Fuglestvedt, J., Millar, R., Cain, M., Frame, D. and Macey, A. (2018). A solution to the misrepresentations of CO2-equivalent emissions of short-lived climate pollutants under ambitious mitigation. Clim. Atmos. Sci. 1, 16.

Brown, P., Cardenas, L., Choudrie, S., Del Vento, S., Karagianni, E., MacCarthy, J., Mullen, P., Passant, N., Richmond, B., Smith, H., Thistlethwaite, G., Thomson, A., Turtle, L. and Wakeling. D. UK Greenhouse Gas Inventory, 1990 to 2019. Annual Report for Submission under the Framework Convention on Climate Change. UK Greenhouse Gas Inventory, 1990 to 2019 (defra.gov.uk)

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

Chen, D., M. Rojas, M., Samset, B.H., Cobb, K., Diongue Niang, A., Edwards, P., Emori, S., Faria, S.H., Hawkins, E., Hope, Huybrechts, P., Meinshausen, M., Mustafa, S.K., Plattner, G.K. and Tréguier A.M. (2021) Framing, Context, and Methods. In: Climate Change 2021: The Physical Science Basis. Contribution of 10 Working Group I to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change

DEFRA GHG R&D Platform, (2013). Defra, UK - Science Search

Evans, C., Artz, R., Moxley, J., Smyth, M-A., Taylor, E., Archer, N., Burden, A., Williamson, J., Donnelly, D., Thomson, A., Buys, G., Malcolm, H., Wilson, D., Renou-Wilson, F. and Potts J. (2017). Implementation of an emission inventory for UK peatlands. Report to the Department for Business, Energy and Industrial Strategy, Centre for Ecology and Hydrology, Bangor.

Matthews, R. (2020). ERAMMP Report-36 National Forest in Wales - Evidence Review Annex-4: Climate Change Mitigation

UNFCCC (2019) Report of the Conference of the Parties serving as the meeting of the Parties to the Paris Agreement on the third part of its first session, held in Katowice from 2 to 15 December 2018

Welsh Government (2021) Emissions of Greenhouse Gases by Year (gov.wales).

Chapter 4

Alison, J., Thomas, A., Evans, C.D., Keith, A.M., Robinson, D.A., Thomson, A., Dickie, I., Griffiths, R.I., Williams, J., Newell-Price, J.P., Williams, A.G., Williams, A.P., Martineau, A.H., Gunn, I.D.M. &

Emmett, B.A. (2019). Technical Annex 3: Soil Carbon Management. In Environment and Rural Affairs Monitoring & Modelling Programme (ERAMMP): Sustainable Farming Scheme Evidence Review. Report to Welsh Government (Contract C210/2016/2017). Centre for Ecology & Hydrology Project NEC06297.

Alvarez, R., (2005). A review of nitrogen fertilizer and conservation tillage effects on soil organic carbon storage. Soil Use Manag. 21, 38–52.

Ammann, C., Flechard, C.R., Leifeld, J., Neftel, A., Fuhrer, J., (2007). The carbon budget of newly established temperate grassland depends on management intensity. Agric. Ecosyst. Environ. 121, 5–20.

Axe,M., Grange, I.D. and Conway, J.S. (2017) Carbon storage in hedge biomass—A case study of actively managed hedges in England. Agriculture, Ecosystems & Environment, 250, 81-88

Bellamy, P., Loveland, P., Bradley, R. Carbon losses from all soils across England and Wales 1978–2003. Nature 437, 245–248 (2005).

Black, K., Green, S., Mullooley, G. and Poveda A. (2014). Carbon Sequestration by Hedgerows in the Irish Landscape. EPA Climate Change Research Programme (CCRP) 2007-2013 Report Series No. 32 Bioenergy crops.

Bradford, M.A, Keiser, A.D., Davies, C.A., Mersmann, C.A. and Strickland, M. S. (2013). Empirical evidence that soil carbon formation from plant inputs is positively related to microbial growth. Biogeochemistry 113, 271-281.

Bradley, R.I., Milne, R., Bell, J., Lilly, A., Jordan, C., and Higgins, A. (2005) A soil carbon and land use database for the United Kingdom. Soil Use and Management 21, 363-369.

Briske, D., Derner, J., Brown, J., Fuhlendorf, S., Teague, R., Gillen, B., Ash, A., Havstad, K. and Willms, W. (2008). Rotational Grazing on Rangelands: An Evaluation of the Experimental Evidence. Rangeland Ecology and Management, 61, 3-17.

Buckingham, S., Cloy, J., Topp, K., Rees, B., and Webb, J. (2013). Capturing Cropland and Grassland Management Impacts on Soil Carbon in the UK LULUCF Inventory. Defra project SP1113.

Cardinael, R., Chevallier, T., Cambou, A., Béral, C., Barthès, B.G., Dupraz, C., Durand, C., Kouakoua, E. and Chenu, C. (2017). Increased soil organic carbon stocks under agroforestry : A survey of six different sites in France. Agric. Ecosyst. Environ. 236, 243–255.

Chamberlain, P.M., Emmett, B.A., Scott, W.A., Black, H.I.J., Hornung, M., Frogbrook, Z.L., (2010). No change in topsoil carbon levels of Great Britain, 1978–2007. Biogeosciences Discuss. 7, 2267–2311.

Chenu, C., Angers D.A., Barré, P., Derrien, D., Arrouays, D. and Balesdent, J. (2019) Increasing organic stocks in agricultural soils: Knowledge gaps and potential innovations, Soil and Tillage Research,188, 41-52.

Conant, R.T., Paustian, K. and Elliot, E. (2001) Grassland Management and Conversion into Grassland: Effects on Soil Carbon. Ecological Applications, 11(2), 343–355.

Conant, R.T., Cerri, C.E.P., Osborne, B.B. and Paustian, K. (2017) Grassland management impacts on soil carbon stocks: a new synthesis. Ecol Appl 27, 662–668.

Daniels, S.M., Evans, M.G., Agnew, C.T. and Allott, T.E.H. (2012). Ammonium release from a balnket peatland into headwater stream systems. Environmental Pollution 163, 261-272.

Emmett, B.., Reynolds, B., Chamberlain, P.M., Rowe, E., Spurgeon, D., Brittain, S.A., Frogbrook, Z., Hughes, S., Lawlor, A.J., Poskitt, J., Potter, E., Robinson, D.A., Scott, A., Wood, C., Woods, C. (2010). CS Technical Report No. 9/07: Soils Report from 2007.

Evans, C., Rawlins, B., Grebby, S., Scholefield, P. and Jones, P. (2015). Glastir Monitoring & Evaluation Programme. Mapping the extent and condition of Welsh peat. (Contract reference C147/2010/11), Centre for Ecology and Hydrology, Bangor. https://gmep.wales/resources.

Evans, C., Artz, R., Moxley, J., Smyth, M-A., Taylor, E., Archer, N., Burden, A., Williamson, J., Donnelly, D., Thomson, A., Buys, G., Malcolm, H., Wilson, D., Renou-Wilson, F., and Potts J. (2017).

Implementation of an emission inventory for UK peatlands. Report to the Department for Business, Energy and Industrial Strategy, Centre for Ecology and Hydrology, Bangor.

Evans, C.D., Peacock, M., and Baird, A.J. (2021). Overriding water table control on managed peatland greenhouse gas emissions. Nature 593, 548–552.

Falloon, P., Powlson, D., and Smith, P. (2010). Managing field margins for biodiversity and carbon sequestration: a Great Britain case study. Soil Use Manag. 20, 240–247.

FAO (2017) Voluntary Guideleines for Sustainable Soil Management

Fornara, D A , Wasson, E-A, Christie, P. and Watson, C. (2016) Long-term nutrient fertilization and the carbon balance of permanent grassland: any evidence for sustainable intensification? Biogeosciences, 13, 4975–4984.

Fornara, D A, Olave, R. and Higgins A. (2020) Evidence of low response of soil carbon stocks to grassland intensification. Agriculture, Ecosystems and the Environment, 287, 10675

Freibauer, A., Rounsevell, M.D.A., Smith, P. and Verhagen, J., 2004. Carbon sequestration in the agricultural soils of Europe. Geoderma 122, 1–23.

Frogbrook Z.L., Bel,l J, Bradley, .RI., Evans, C., Lark, R.M., ,Reynolds B., Smith, P, and Towers, W. (2009) Quantifying terrestrial carbon stocks: examining the spatial variation in two upland areas in the UK and a comparison to mapped estimates of soil carbon. Soil Use and Management 25: 320-332.

Garnett, T., Godde, C., Muller, A., Röös, E., Smith, P., de Boer, I.J.M., zu Ermgassen, E., Herrero, M., van Middelaar, C., Schader, C. and van Zanten, H. (2017). Grazed and Confused? Ruminating on cattle, grazing systems, methane, nitrous oxide, the soil carbon sequestration question – and what it all means for greenhouse gas emissions. FCRN, University of Oxford.

Godde, C. M., de Boer, I. J. M., zu Ermgassen, E., Herrero, M., van Middelaar, C. E., Muller, A., Röös, E., Schader, C., Smith, P., van Zanten, H.H. and Garnett, T. (2020) Soil carbon sequestration in grazing systems: managing expectations. Climatic Change. 161:385–391.

Guo, L.B. and Gifford, R.M. (2002). Soil carbon stocks and land use change: a meta analysis. Global Change Biology 8, 345-360.

Henderson, B.B., Gerber, P.J., Hilinski, T.E., Falcucci, A., Ojima D.S., Salvatore, M. and Conant, R.T. (2015). Greenhouse gas mitigation potential of the world's grazing lands: Modeling soil carbon and nitrogen fluxes of mitigation practices. Agriculture, Ecosystems and Environment, 207, 91-100.

Hopkins, D.W., Waite, I.S., McNicol, J.W., Poulton, P.R., Macdonald, A.J. and O'Donnell, A.G. (2009). Soil organic carbon contents in long-term experimental grassland plots in the UK (Palace Leas and Park Grass) have not changed consistently in recent decades. Glob. Chang. Biol. 15, 1739–1754.

Eggelston, S., Buendia, L., Miwa, K., Ngara, T. and Tanabe, K. (2006). IPCC Guidelines for National Greenhouse Gas Inventories

Jeffery, S., Verheijen, F.G.A., van der Velde, M. and Bastos, A.C. (2011) A quantitative review of the effects of biochar application to soils on crop productivity using meta-analysis Agric. Ecosyst. Environ. 144 175–187.

Jeffery, S., Abalos, D., Prodana, M., Bastos, A.C., van Groenigen. J.W., Hungate, B.A., and Verheijen, F. (2017) Biochar boosts tropical but not temperate crop yields. Environ. Res. Lett. 12 053001.

Jobbágy, E.G. and Jackson, R.B. (2000). The Vertical Distribution of Soil Organic Carbon and Its Relation to Climate and Vegetation. Ecological Applications, 10, 423-436.

Jones, S.K., Rees, R.M., Kosmas, D., B. C. Ball, B.C., and Skiba, U.M. (2006) Carbon sequestration in a temperate grassland; management and climatic controls Soil Use and Management, 22 132-142

Kell, D. (2012) Large-scale sequestration of atmospheric carbon via plant roots in natural and agricultural ecosystems: why and how. Phil. Trans. R. Soc. B 367, 1589–1597.

Liu, L. and Greaver, T.L. (2010). A global perspective on below-ground carbon dynamics under nitrogen enrichment. Ecol. Lett. 13, 819–828 (2010).

Manley, J., van Kooten, G.C., and Moeltner, K. (2005). Creating Carbon Offsets in Agriculture through No-Till Cultivation: A Meta-Analysis of Costs and Carbon Benefits. Climatic Change 68, 41–65.

Matthews, R. (2020) ERAMMP Report-36 National Forest in Wales - Evidence Review Annex-4: Climate Change Mitigation. Report to Welsh Government (Contract C210/2016/2017). Centre for Ecology & Hydrology Project NEC06297

McCalmont, J., McNamara, N. P., Donnison, I., Farrar, K. and Clifton-Brown, J. (2017a). An inter-year comparison of CO2 flux and carbon budget at a commercial-scale land-use transition from semi-improved grassland to Miscanthus x giganteus. GCB Bioenergy 9, 229–245.

McCalmont, J., Hastings, A., McNamara, N., Richter, G. M., Robson, P., Donnison, I., and Clifton-Brown, J. (2017b). Environmental costs and benefits of growing Miscanthus for bioenergy in the UK. GCB Bioenergy, 9(3), 489-507.

McSherry, M.E. and Ritchie, M.E. (2013). Effects of grazing on grassland soil carbon: a global review. Global Change Biology, 19, pp. 1347–1357.

Morison, J., Matthews, R., Miller, G., Perks, M., Randle, T., Vanguelova, E., White, M. and Yamulki, S. (2012). Understanding the carbon and greenhouse gas balance of forests in Britain. Forestry Commission Research Report. Forestry Commission, Edinburgh

Neff, J.C., Townsend, A.R., Gleixner, G., Lehman, S.J., Turnbull, J. and Bowman, W.D. (2002) Variable effects of nitrogen additions on the stability and turnover of soil carbon. Nature 419:915–917

Nordberg, M. and Roos, E. (2016) Holistic management – a critical review of Allan Savory's grazing method. Swedish University of Agricultural Sciences

Paustian, K., Lehmann, J., Ogle, S., Reay, D., Robertson, G.P. and Smith, P. (2016). Climate-smart soils. Nature 532(7597), 49-57

Poeplau, C. and Don, A. (2015). Carbon sequestration in agricultural soils via cultivation of cover crops—a meta-analysis. Agric. Ecosyst. Environ. 200, 33–41.

Poeplau, C., Zopf, D., Greiner, B., Geerts, R., Korvaar, H., Thumm, U., Don, A., Heidkamp, A. and Flessa, H., (2018). Why does mineral fertilization increase soil carbon stocks in temperate grasslands? Agric. Ecosyst. Environ. 265, 144–155.

Poirier, V., Angers, D.A., Rochette, P., Chantigny, M.H., Ziadi, N., Tremblay, G. and Fortin, J. (2009). Interactive effects of tillage and mineral fertilization on soil carbon profiles, 73(1), 255-261. http://dx.doi.org/10.2136/sssaj2008.0006

Poirier, V., Roumet, C, and Munson, A.D. (2018). The root of the matter: linking root traits and soil organic matter stabilization processes. Soil Biology and Biochemistry 120: 246–259.

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

Powlson, D.S., Whitmore, A.P. and Goulding, K.W.T. (2011). Soil carbon sequestration to mitigate climate change: A critical re-examination to identify the true and the false. Eur. J. Soil Sci. 62, 42–55.

Powlson, D S, , Stirling, C. M., Jat, M. L., Gerard, B.G., Palm, C.A., Sanchez, P.A. and Cassman, K.G. (2014) Limited potential of no-till agriculture for climate change mitigation. Nature Clim Change 4, 678–683.

Quinn, R.J., Ha,H. and Volk,T.A. (2020) Life cycle assessment of forest biomass energy feedstock in the Northeast United States. GCB Bioenergy.12:728–741.

Schils, R.L.M., Verhagen, A., Aarts, H.F.M. and Šebek, L.B.J. (2005). A farm level approach to define successful mitigation strategies for GHG emissions from ruminant livestock systems. Nutr. Cycl. Agroecosystems 71, 163–175.

Schlesinger, W.H. and Amundson, R. (2019). Managing for soil carbon sequestration: Let's get realistic. Glob. Chang. Biol. 25, 386–389.

Schrumpf, M., Schulze, E.D., Kaiser, K., and Schumacher, J. (2011): How accurately can soil organic carbon stocks and stock changes be quantified by soil inventories?, Biogeosciences, 8, 1193–1212,

Schulze, E. D., Luyssaert, S., Ciais, P., Freibauer, A., Janssens, I. A., Soussana, J. F., Smith, P., Grace, J., Levin, I., Thiruchittampalam, B., Heimann, M., Dolman, A. J., Valentini, R., Bousquet, P., Peylin, P., Peters, W., Rodenbeck, C., Etiope, G., Vuichard, N., Wattenbach, M., Nabuurs, G. J., Poussi, Z., Nieschulze, J., and Gash, J. H. (2009) Importance of methane and nitrous oxide for Europe's terrestrial greenhouse-gas balance, Nat. Geosci., 2, 842– 850.

Shi, S.W., Zhang, W., Zhang, P., Yu, Y.Q. and Ding, F. (2013). A synthesis of change in deep soil organic carbon stores with afforestation of agricultural soils. Forest Ecol. Manag. 296, 53–63.

Six, J., Conant, R.T., Paul, E.A. and Paustian, K. (2002) Stabilization mechanisms of soil organic matter : Implications for C-saturation of soils. Plant and Soil, 241, 155–176.

Smith, P., Andren, O., Karlsson, T., Perala, P., Regina, K., Rounsevell, M. and Van Wesemael, B. (2005) Carbon sequestration potential in European croplands has been overestimated. Global Change Biology 11, 2153–2163

Smith, P. et al. (2007) ECOSSE - Estimating carbon in organic soils sequestration and emissions. Climate Change and Air Division, Scottish Executive, Edinburgh

Smith, P., Chapman, S.J., Scott, A.W., Black, H.I.J., Wattenbach, M., Milne, R., Campbell, C.D., Lilly, A., Ostle, N., Levy, P.E., Lumsdon, D.G., Millard, P., Towers, W., Zaehle, S., and Smith, J.U. (2007). Climate change cannot be entirely responsible for soil carbon loss observed in England and Wales, 1978-2003. Glob. Chang. Biol. 13, 2605–2609.

Smith, P., Martino, D., Cai, Z., Gwary, D., Janzen, H., Kumar, P., McCarl, B., Ogle, S., O'Mara, F., Rice, C., Scholes, B., Sirotenko, O., Howden, M., McAllister, T., Pan, G., Romanenkov, V., Schneider, U., Towprayoon, S., Wattenbach, M. and Smith, J. (2008). Greenhouse gas mitigation in agriculture. Philos Trans R Soc, 363(1492), 789–813.

Smith, P. (2008). Land use change and soil organic carbon dynamics. Nutr. Cycl. Agroecosystems 81, 169–178.

Smith P. (2014) Do grasslands act as a perpetual sink for carbon? Global Change Biology 20, 2708–2711

Smith P. et al. (2016) Biophysical and economic limits to negative CO2 emissions. Nature Climate Change 6, 42-50.

Sokol N., Kuebbing S.E., Karlsen-Ayala E., and Bradford M. (2019). Evidence for the primacy of living root inputs, not root or shoot litter in forming soil organic carbon. New Phytologist, 221: 233-246.

Soussana J.F., Allard V, Pilegaard K, Ambus C, Campbell C, Ceschia E, Clifton-Brown J, Czobel S, Domingues R, Flechard C, Fuhrer J, Hensen A, Horvath L, Jones M, Kasper G, Martin C, Nagy Z, Neftel A, Raschi A, Baronti S, Rees RM, Skiba U, Stefani P, Manca G, Sutton M, Tuba Z and Valentini R (2007). Full accounting of the greenhouse gas (CO2, N2O CH4) budget of nine European grassland sites. Agriculture, Ecosystems and Environment 121, 121–134.

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

Soussana, J.F. and Lemaire, G., (2014). Coupling carbon and nitrogen cycles for environmentally sustainable intensification of grasslands and crop-livestock systems. Agric. Ecosyst. Environ. 190, 9–17.

Thomson, A., Buys, G., Cliverd, H. Malcolm, H., Henshall, H. and Matthews R. (2020) Projections of Emissions and Removals from the LULUCF Sector to 2050/2100 : National Atmospheric Emissions Inventory Report ERG 1103

Upson, M.A., Burgess, P.J. and Morison J.I.L. (2016). Soil carbon changes after establishing woodland and agroforestry trees in a grazed pasture. Geoderma 283: 10-20.

Vanguelova, E.I., Nisbet, T.R., Moffat, A.J., Broadmeadow, S., Sanders, T.G.M. and Morison, J.I.L. (2013) A new evaluation of carbon stocks in British forest soils. Soil Use and Management 29, 169-181.

Welsh Government (2020) Land Use Statistics and research | GOV.WALES

Welsh Government (2021) Emissions of Greenhouse Gases by Year (gov.wales)

White, R.E, Davidson, B., Shu Kee Lam and Chen, D. (2018) A critique of the paper 'Soil carbon 4 per mille' by Minasny et al. (2017) Geoderma 309, 115-117.

Woolf, D., Amonette, J.E., Street-Perrott, F.A., Lehmann, J. and Joseph, S. (2010) Sustainable biochar to mitigate global climate change Nat. Commun. 1 56, 1-9.

Chapter 5

AHDB (2021) RB209 Principles of Nutrient Management

Akiyama, H., Xiaoyuan, Y. and Yagi, K. (2010) Evaluation of effectiveness of enhanced-efficiency fertilizers as mitigation options for N2O and NO emissions from agricultural soils: meta-analysis. Global Change Biology 16 ,1837-1846

Amon, B., V. Kryvoruchko, V., Amon, T., and Zechmeister-Boltenstern S. (2006) Methane, nitrous oxide and ammonia emissions during storage and after application of dairy cattle slurry and influence of slurry treatment, Agriculture, Ecosystems & Environment. 112, 2–3, 153-162.

Badgley, C., Mohhtader, J., Quintero, E., Zakem, E., Chappel, M., Aviles-Vazquez, K., Perfecto, I. (2007). Organic agriculture and the global food supply. Renewable Agriculture and Food Systems, 22, 86-108.

Banks C. J., Salter, A.M., Heaven, S. and Riley, K. (2011) Energetic and environmental benefits of co-digestion of food waste and cattle slurry: a preliminary assessment. Resources, Conservation and Recycling 56, 71-79.

Bernet, N., N. Delgenes, J. C. Akunna, J. P. Delgenes, and R. Moletta. 2000. Combined anaerobic-aerobic SBR for the treatment of piggery wastewater. Water Res. 34:611–619

Butterbach-Bahl, K., Baggs, E. M., Dannenmann, M., Kiese, R., and ZechmeisterBoltenstern, S. (2013). Nitrous oxide emissions from soils: how well do we understand the processes and their controls? Phil. Trans. R. Soc. B 368:20130122.

Cao, Y., Wang, X., Liu, L., Velthof, G. L., Misselbrook, T. H., Bai, Z. and Ma, L. (2020) Acidification of manure reduces gaseous emissions and nutrient losses from subsequent composting process. Journal of Environmental Management. 264, 110454.

Cardenas, L. M., Bhogal, A., Chadwick, D.R., McGeough, K., Misselbrook, T. H., Rees, R. M., Thorman, R. E., Watson, C. J., Williams, J. R., Smith, K. A. and 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., Thorman R, and Ashlee N. (2010) Quantifying annual N2O emission fluxes from grazed grassland under a range of inorganic fertilizer nitrogen inputs. Agric. Ecosyst. Environ., 136, 218–226.

Cardenas, L. M., Bol, R., Lewicka-Szczebak, D., Gregory, A. S., Matthews, G. P., Whalley, W. R., Misselbrook, T. H., Scholefield, D. and Well, R. (2017). Effect of soil saturation on denitrification in a grassland soil. Biogeosciences. 14 (20), 4691-4710.

Cardenas, L. M., Misselbrook, T. H., Hodgson, C. J., Donovan, N., Gilhespy, S. L., Smith, K. A., Dhanoa, M. S. and Chadwick, D. R. (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 & Environment. 235, 229-241.

Carswell, A. M., Shaw, R., Hunt, J., Sanchez-Rodriguez, A. R., Saunders, K. S., Cotton, J., Hill, P. W., Chadwick, D. R., Jones, D. L. and Misselbrook, T. H. (2018). Assessing the benefits and wider costs of different N fertilisers for grassland agriculture . Archives of Agronomy and Soil Science. 65 (5), pp. 625 - 639.

Cassman, K.G., Dobermann, A. and Walters, D.T. (2002). Agroecosystems, nitrogen-use efficiency, and nitrogen management. Ambio, 31, 132–140.

Chadwick, D. (2005). Emissions of ammonia, nitrous oxide and methane from cattle manure heaps: effect of compaction and covering. Atmos. Environ. 39, 787–799.

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

Chambers B. and Dampney P., (2009) Nitrogen efficiency and ammonia emissions from urea360 based and ammonium nitrate fertilizers. Proc No. 657, Internat Fert Soc, York, UK

Chambers, B.J., Smith, K.A., and Pain, B.F. (2000) Strategies to encourage better use of nitrogen in animal manures. Soil Use Management 16,157–161.

Crews, T. and Peoples, M. (2004) Legume versus Fertilizer Sources of Nitrogen: Ecological Tradeoffs and Human Needs. Agriculture, Ecosystems & Environment, 102, 279-297.

Davidson, E.A., Keller, M, Erickson, H.E, Verchot, L.V. and Veldkamp, E. (2000) Testing a conceptual model of soil emissions of nitrous and nitric oxides. Bioscience 50, 667 – 680.

De Ponti, T., Rijk, B. & van Ittersum, M. K. (2012) The crop yield gap between organic and conventional agriculture. Agric. Ecosyst. Environ. 108, 1–9.

Defra, UK - Science Search

DEFRA (2020) Soil Nutrient Balances - UK Provisional Estimates for 2019

Dijkstra, J., Oenama, O., Van Groenigan, J., Spek, J., Van Vuuren, A., and Bannink, A. (2013). Diet effects on urine composition of cattle and N2O emissions. Animal, 7, 292-302.

Dobbie, K.E. and Smith, K.A. (2006) The effect of water table depth on emissions of N2O from a grassland soil. Soil Use Manage., 22, 22–28.

Duncan, E.W., Dell, C.J., Kleinman, P.J. and Beegle, D.B. (2017). Nitrous oxide and ammonia emissions from injected and broadcast-applied dairy slurry. J. Environ Qual.46, 36–44.

Eory, V., Maire, J., MacLeod, M., Sykes, A., Barnes, A., Rees, R.M., Topp, C.F.E. and Wall, E. (2020) Non-CO2 abatement in the UK agricultural sector by 2050 Summary report submitted to UKCCC to support the 6th Carbon Budget in the UK

Ellis J.L., Dijkstra J., Bannink A., Parsons A.J., Rasmussen S., Edwards G.R., Kebreab E. and France J. (2011). The effect of high-sugar grass on predicted nitrogen excretion and milk yield simulated using a dynamic model. Journal of Dairy Science, 94, 3105-3118.

EU Nitrogen Expert Panel (2015). Nitrogen Use Efficiency (NUE) - an indicator for the utilization of nitrogen in agriculture and food systems. Wageningen University, Alterra, PO Box 47, NL-6700 Wageningen, Netherlands.

Fangueiro, D., Hjorth, M., and Gioelli, F. (2015). Acidification of animal slurry--a review. J Environ Manage. 149:46-56.

Firestone, M.K. and Davidson, E.A. (1989) Microbiological Basis of NO and N2O Production and Consumption in Soils. In: Andreae, M.O. and Schimel, D.S., Eds., Exchange of Trace Gases between Terrestrial Ecosystems and the Atmosphere, John Willey and Sons, New York, 7-21

Foskolos, A., and Moorby, J. (2017). The use of high sugar grasses as a strategy to improve nitrogen utilization efficiency: A meta-analysis. Advances in Animal Bioscience 8. 072 https://doi.org/10.1017/S2040470017001479

Fu, Q., Abadie, M., Blaud, A., Carswell, A.M., Misselbrook, T.H., Clark, I.M. and Hirsch, P.R. (2019). Effects of urease and nitrification inhibitors on soil N, nitrifier abundance and activity in a sandy loam soil . Biology And Fertility of Soils. 1-10.

Gillingham AG, Ledgard SF, Saggar S, Cameron KC, Di HJ, de Klein CAM, Aspin MD (2012) Initial evaluation of the effects of dicyandiamide (DCD) on nitrous oxide emissions, nitrate leaching and dry matter production from dairy pastures in a range of locations within New Zealand. In: Currie LD, Christensen CL (eds) Advanced nutrient management: gains from the past-goals for the future. Occasional report no. 25, FLRC. Massey University, Palmerston North,

Gilsanz, C., Baez, D., Misselbrook, T.H., Dhanoa, M.S., and Cardenas, L.M. (2016). Development of emission factors and efficiency of two nitrification inhibitors, DCD and DMPP. Agric. Ecosyst. Environ. 216, 1–8.

Hansen, M.N., Henriksen, K., Sommer, S.G., (2006). Observations of production and emission of greenhouse gases and ammonia during storage of solids separated from pig slurry: effects of covering. Atmos. Environ. 40, 4172–4181.

Hoben JP, Gehl RJ, Millar N, et al. (2011). Non-linear nitrous oxide response to nitrogen fertilizer in on-farm corn crops of the US Midwest. Glob Change Biol 17: 1140–52.

Huhtanen, P. and Hristov, A.N. (2009). A meta-analysis of the effects of dietary protein concentration and degradability on milk protein yield and milk N efficiency in dairy cows. Journal of Dairy Science 92, 3222–3222.

Jacobsen, B.H. (2017). Why is acidification of slurry a success only in Denmark? Transfer of environmental technology across borders. Paper presented at IFMA Congress , Edinburgh, United Kingdom.

Jensen, E. S., Peoples, M. B., Boddey, R. M., Gresshoff, P. M., Hauggaard-Nielsen, H., Alves, B. J. R., & Morrison, M. J. (2012). Legumes for mitigation of climate change and the provision of feedstock for biofuels and biorefineries: A review. Agronomy for Sustainable Development, 32(2), 329–364.

Kamal P. Adhikari, Grace Chibuike, Surinder Saggar, Priscila L. Simon, Jiafa Luo, Cecile A.M. de Klein (2021) Management and implications of using nitrification inhibitors to reduce nitrous oxide emissions from urine patches on grazed pasture soils – A review, Science of The Total Environment, 79, https://doi.org/10.1016/j.scitotenv.2021.148099.

Klefoth, R. R., Clough, T. J., Oenema, O., & van Groenigen, J. W. (2014). Soil bulk density and moisture content influence relative gas diffusivity and the reduction of nitrogen-15 nitrous oxide. Vadose Zone Journal, 13(11). https://doi.org/10.2136/vzj2014.07.0089

Kupper, T., Hani, C., Neftel, A., Kincaid, C., Buhler, M., Amon, B. and VanderZaag A. (2020) Ammonia and greenhouse gas emissions from slurry storage – a review. Agriculture, Ecosystems and Environment 300, 106963’

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

Lazcano, C., Tsang, A., Doane, T.A., Pettygrove, G.S., Horwath, W.R., and Burger, M. (2016). Soil nitrous oxide emissions in forage systems fertilized with liquid dairy manure and inorganic fertilizers. Agric. Ecosyst. Environ. 225, 160–172.

Li, D., Watson, C.J., Yan, M.J., Lalor, S., Rafique, R., Hyde, B., Lanigan, G., Richards, K.G., Holden, N.M. and Humphreys J. (2013). A review of nitrous oxide mitigation by farm nitrogen management in temperate grassland-based agriculture. J Environ Management 128: 893-903.

Ledgard S., Schils R., Eriksen J. and Luo J. Environmental impacts of grazed clover/grass pastures. Irish Journal of Agricultural and Food Research. 2009;48:209–226

Loyon, L., Burton, C.H., Misselbrook, T.H., Webb, J., Philippe, F.X., Aguilar, M., Doreau, M., Hassouna, M., Veldkamp, T., Dourmad, J.Y., Bonmati, A., Grimm, E. and Sommer, S.G. (2016). Best available technology for European livestock farms: Availability, effectiveness and uptake. Journal of Environmental Management. 166, 1-11.

Luo, G. J., Kiese, R., Wolf, B., and Butterbach-Bahl, K. (2013) Effects of soil temperature and moisture on methane uptake and nitrous oxide emissions across three different ecosystem types, Biogeosciences, 10, 3205–3219.

Maaz, T.M., Sapkota, T.B., Eagle, A.J., Kantar, M.B., Bruulsema, T.W. and Kaushik, M. (2021)Meta-analysis of yield and nitrous oxide outcomes for nitrogen management in agriculture . Global Change Biology 2021;27:2343–2360.

Marsden, K.A., Scowen, M., Hill, P.W., Jones, D.L., Chadwick, D.R., 2015. Plant acquisition and metabolism of the synthetic nitrification inhibitor dicyandiamide and naturally-occurring guanidine from agricultural soils. Plant Soil 395, 201–214

Masse, D. I., G. Talbot, and Y. Gilbert. 2011. On farm biogas production: A method to reduce GHG emissions and develop more sustainable livestock operations. Anim. Feed Sci. Technol. 166–167:436–445

McGeough, K. L., Watson, C. J., Müller, C., Laughlin, R.J., and Chadwick, D.R. (2016). Evidence that the efficacy of the nitrification inhibitor dicyandiamide (DCD) is affected by soil properties in UK soils. Soil Biol. Biochem. 94, 222–232

McSwiney, C. P. and G. P. Robertson. 2005. Non-linear response of N2O flux to incremental fertilizer addition in a continuous maize (Zea mays sp.) cropping system. Global Change Biology 11:1712-1719.

Miller, L. A., Moorby, J.M., Davies, D.R. Humphreys, M.O., Scollan, N.D. and MacRae, J.C. (2001). Increased concentration of water-soluble carbohydrate in perennial ryegrass (Lolium perenne L.). Milk production from late-lactation dairy cows. Grass Forage Sci. 56:383–394.

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

Misselbrook, T.H., Cardenas, L.M., Camp, V., Thorman, R.E., Williams, J.R., and Rollett, A.J.(2014). An assessment of nitrification inhibitors to reduce nitrous oxide emissions from UK agriculture. Environ. Res. Lett. 9:115006.

Misselbrook, T.H., Brookman, S.K.E., Smith, K.A., Cumby, T.R., Williams, A.G. and McCrory,D.F. (2005b). Crusting of stored dairy slurry to abate ammonia emissions: Pilot-scale studies. J. Environ. Qual. 34:411–419.

Mondelaers, K., Aertsens, J., and van Huylenbroeck, G. (2009) A meta-analysis of the differences in environmental impacts between organic and conventional farming. Br. Food J., 111, 1098–1119.

Moran, Dominic & Macleod, Michael & Wall, Eileen & Eory, Vera & Pajot, Guillaume & Mathews, R. & McVittie, Alistair & Barnes, Andrew & Rees, Bob & Moxey, Andrew & Williams, Adrian & Smith, Pete. (2008). UK Marginal Abatement Cost Curves for the Agriculture and Land Use, Land-Use Change and Forestry Sectors out to 2022, with Qualitative Analysis of Options to 2050. -.

Montes, F., Meinen, R., Dell, C., Rotz, A., Hristov, A.N., Oh, J., Waghorn, G., Gerber, P.J., Henderson, B., Makkar, H.P. and Dijkstra J. (2013).Special topics--Mitigation of methane and nitrous oxide emissions from animal operations: II. A review of manure management mitigation options. J Anim Sci. 91(11):5070-94.

Oenema, O., Wrage, N., Velthof, G.L., van Groeningen, J.W., Dolfing, J. and Kuikman, P.J., (2005). Trends in global nitrous oxide emissions from animal production systems. Nutr. Cycling Agroecosyst. 72, 51–65.

Petersen, S. O., and Sommer, S.G. (2011). Ammonia and nitrous oxide interactions: Roles of manure organic matter management. Anim. Feed Sci. Technol. 166–167:503–513

Phelan, P., Moloney, A. P., McGeough, E. J., Humphreys, J., Bertilsson, J., O'Riordan, E.G. and O'Kiely, P. (2015) Forage Legumes for Grazing and Conserving in Ruminant Production Systems, Critical Reviews in Plant Sciences, 34: 281-326.

Phillips R, Tanaka D, Archer D, and Hanson J. 2009. Fertilizer application timing influences greenhouse gas fluxes over a growing season. J Environ Qual 38: 1569–79.

Roboredo, M., Fangueiro, D., Lage, S., and Coutinho, J.(2011). Phosphorus dynamics in soils amended with acidified pig slurry and derived solid fraction Geoderma 189-190, 328-333

Rochette, P. and Janzen, H. (2005) Towards a revised coefficient for estimating N2O emissions from legumes. Nutr. Cycling Agroecosyst., 73, 171–179

Rom, H.B. and Henriksen, K. (2000). Nitrogen loss from cattle housed on deep litter. In Husdyrgødning og kompost (ed. SG Sommer and J Eriksen), pp. 5–13. Forskningscenter for Økologisk Jordbrug, Tjele, Denmark

Schils, R.L.M., van Groenigen, J.W., Velthof, G.L., and Kuikman, P.J. (2008) Nitrous oxide emissions from multiple combined applications of fertilizer and cattle slurry to grassland. Plant Soil, 310, 89–101.

Sigurdarson JJ, Svane S, Karring H (2018) The molecular processes of urea hydrolysis in relation to ammonia emissions from agriculture. Rev Environ Sci Biotechnol 17:241–258

Smith, K.A., Jackson, D.R., Misselbrook, T.H., Pain, B.F., and Johnson, R.A. (2000). Reduction of ammonia emission by slurry application techniques. J. Agric. Eng. Res. 77, 277–287.

Smith, L.G., Jones, P.J., Kirk, G.J.D., Pearce, B.D. & Williams, A.G. (2018) Modelling the impact of a widespread conversion to organic agriculture in England and Wales. Land Use Policy 76, 391–404.

Smith, K., Dobbie, K., Thorman, R. and Yamulki, S. (2006). The effect on N fertiliser forms on nitrous oxide emissions. DEFRA Report NT26

Smith, K.A., Dobbie, K.E., Thorman, R., Watson, C.J., Chadwick, D.R., Yamulki, S and Ball, B.C. (2012) The effect of N fertilizer forms on nitrous oxide emissions from UK arable land and grassland, Nutr Cycl Agroecosyst 93:127–149

Smith, L.G., Kirk, G.J.D., and Jones, P.J. (2019). The greenhouse gas impacts of converting food production in England and Wales to organic methods. Nat Commun 10, 4641.

Stehfest, E. and Bouwman, L. (2006) N2O and NO Emission from Agricultural Fields and Soils under Natural Vegetation: Summarizing Available Measurement Data and Modeling of Global Annual Emissions. Nutrient Cycling in Agroecosystems, 74, 207-228.

Subbarao, G., Ito, O., Sahrawat, K., Berry, W., Nakahara, K., Ishikawa, T., Watanabe, T.,

et al., (2006). Scope and strategies for regulation of nitrification in agricultural

systems: challenges and opportunities. Critical Reviews in Plant Sciences 25, 303-335.

Thorman, R. E., Nicholson, F. A., Topp, C. F. E., Bell, M. J., Cardenas, L. M., Chadwick, D. R., Cloy, J. M., Misselbrook, T. H., Rees, R. M., Watson, C. J. and Williams , J. R. (2020). Towards Country-Specific Nitrous Oxide Emission Factors for Manures Applied to Arable and Grassland Soils in the UK. Frontiers in Sustainable Food Systems. 4, p. 62. https://doi.org/10.3389/fsufs.2020.00062

Thorman, R.E., Chadwick, D.R., Harrison, R., Boyles, L.O., and Matthews, R. (2007). The effect on N2O emissions of storage conditions and rapid incorporation of pig and cattle FYM into arable land. Biosyst. Eng. 97, 501–511

Trenkel, M. (2010) Slow and Controlled Release Fertilizers. International Fertilizer Industry Association.

Van der Weerden, T.J., Laurenson, S., Vogeler, I., Beukes, P.C., Thomas, S.M., Rees, R.M., Topp, C.F.E., Lanigan, G. and de Klein, C.A.M. (2017) Mitigating nitrous oxide and manure-derived methane emissions by removing cows in response to wet soil conditions. Agricultural Systems DOI: 10.1016/j.agsy.2017.06.010

Van Groenigen, J.W., Zwart, K.B., Harris, D. and van Kessel, C. (2005). Vertical gradients of 15N and 18O in soil atmosphere N2O – temporal dynamics in a sandy soil. Rapid Communications in Mass Spectrometry 19: 1289-1295

Venterea, R.T., Halvorson, A.D., Kitchen, N. et al. (2012). Challenges and opportunities for mitigating nitrous oxide emissions from fertilized cropping systems Front Ecol Environ 10(10): 562–570

Velthof, G. L. and Mosquera, J. (2011). The impact of slurry application technique on nitrous oxide emission from agricultural soils. Agric. Ecosyst. Environ. 140, 298–308

Vestergaard, A.V. (2014) SEGES report. Status, economy and consideration by acidification of slurry. INTERREG Baltic Sea Region. Baltic Slurry Acidification

Webb, J., Chadwick, D., and Ellis, S., (2004). Emissions of ammonia and nitrous oxide following incorporation into the soil of farmyard manures stored at different densities. Nutr. Cycling Agroecosyst. 70, 67–76

Webb, J., Ryan, M., Anthony, S. G., Brewer, A., Laws, J. A., Aller, M. F. and Misselbrook, T. H. (2006). Cost-effective means of reducing ammonia emissions from UK agriculture using the NARSES model. Atmospheric Environment. 40 (37), pp. 7222-7233.

Webb, J., Pain, B., Bittman, S., and Morgan, J. (2010). The impacts of manure application methods on emissions of ammonia, nitrous oxide and on crop response - a review. Agric. Ecosyst. Environ. 137, 39–46

Welten, B.G., Ledgard, S.F. Balvert, M.J. et al. (2016). Effects of oral administration of dicyandiamide to lactating dairy cows on residues in milk and the efficacy of delivery via a supplementary feed source. Agriculture, Ecosystems & Environment, 217,111-118

Woodland Trust (2013). The Pontbren Project A farmer-led approach to sustainable land management in the uplands.

Zaman, M, Nguyen, M.L., Blennerhassett, J.D. and Quin, B.F. (2008) Reducing NH3, N2O and NO3--N losses from a pasture soil with urease or nitrification inhibitors and elemental S-amended nitrogenous fertilizers. Biol Fertil Soils 44:693–705.

Zerulla, W., Barth, T., Dressel, K.E, von Locquenghien, K.H. and Wissemeier, A.H. (2001) 3,4-Dimethylpyrazole phosphate (DMPPP)—a new nitrification inhibitor for agriculture and horticulture. Biol Fertil Soils 34:79–84

Chapter 6

Abberton, M.T., MacDuff, J.H., Marshall, A.H. and Humphreys, M.W. (2007). The Genetic Improvement of Forage Grasses and Legumes to Reduce Greenhouse Gas Emissions. Food and Agriculture Organization of the United Nations.

Aboagye, I. A. and Beauchemin, K.A. (2019) Potential of Molecular Weight and Structure of Tannins to Reduce Methane Emissions from Ruminants: A Review. 9, 856. doi: 10.3390/ani9110856.

AHDB (2015) A decade of genetic progress in the English sheep industry

Alford, A.C., Hegarty, R.S., Parnell, P.F., Cacho, O.J., Herd, R.M., Cacho and Griffith, G.R. (2006). The impact of breeding to reduce residual feed intake on enteric methane emissions from the Australian beef industry. Australian Journal of Experimental Agriculture 46(7) 813-820.

Alstrup L, Søegaard K, and Weisbjerg MR. (2016). Effects of maturity and harvest season of grass-clover silage and of forage-to-concentrate ratio on milk production of dairy cows. J Dairy Sci.99(1):328-40.

Alstrup, L., Frydendahl Hellwing, A.L., Lund, P., Weisbjerg, M.R., (2015). Effect of fat supplementation and stage of lactation on methane production in dairy cows. Animal Feed Sci. Tech. 207, 10–19.

Anderson R C, Huwe J K, Smith D J, Stanton T B, Krueger N A, Callaway T R, Edrington T S, Harvey R B, Nisbet DJ. (2010). Effect of nitroethane, dimethyl-2-nitroglutarate and 2- nitro-methyl-propionate on ruminal methane production and hydrogen balance in vitro. Biores Technol. 101: 5345–5349.

Beauchemin, K.A., Mcginn, S.M. and Petit, H.V.,(2007). Methane abatement strategies for cattle: lipid supplementation of diets. Can. J. Anim. Sci. 87, 431–440.

Beauchemin, K.A., Janzen, H.H., Little, S.M., McAllister, T.A. and McGinn, S.M. (2010.) Life cycle assessment of greenhouse gas emissions from beef production in western Canada: A case study. Agricultural Systems, 103, 371-379.

Bell, J., Beaton, CB., McDowell, MM., Hill, GJ., Stout, DS., Sellars, AS., Thomson, SG., Spencer, M., & Moxey, A. P. (2021). Suckler Beef Climate Change Group - Farm Carbon Case Studies. The Scottish Government.
https://www.gov.scot/binaries/content/documents/govscot/publications/independent-report/2021/01/suckler-beefclimate-scheme-final-report-2/documents/low-carbon-beef-case-study/low-carbon-beef-casestudy/govscot%3Adocument/low-carbon-beef-case-study.pdf

Benchaar, C., Calsamiglia, S., Chaves, A.V., Fraser, G.R., Colombatto, D., McAllister, T.A., Beauchemin, K.A., (2008). A review of plant-derived essential oils in ruminant nutrition and production. Anim. Feed Sci. Technol. 145, 209–228.

Beukes, P.C., Gregorini, P., Romera, A.J., Levy, G., and Waghorn, G.C., (2010). Improving production efficiency as a strategy to mitigate greenhouse gas emissions on pastoral dairy farms in New Zealand. Agric. Ecosyst. Environ. 136, 358–365.

Capper, J. L., R. A. Cady, and D. E. Bauman. (2009). The environmental impact of dairy production: 1944 compared with 2007. J. Anim. Sci. 87:2160–2167

Cooke, J., Duthie, C. A., Troy, S., Hyslop, J., Ross, D., Waterhouse, T. and Roehe, R. (2014) Effects of dietary nitrate and lipid on methane emissions from beef cattle are basal diet dependant. SRUC

Cottle D.J. and Conington J. (2012) Breeding for reduced methane emissions in extensive UK sheep systems. J Agric Science. 150, 570-583.

Dijkstra, J A. Bannink, J. France, Kebreab, E. and van Gastelen S. (2018) Antimethanogenic effects of 3-nitrooxypropanol depend on supplementation dose, dietary fibre content, and cattle type. J. Dairy Sci. 101:9041–9047

Donoghue, KA, Bird-Gardiner, TL, Arthur, PF, Herd, RM and Hegarty, RS (2015). Genetic parameters for methane production and relationships with production traits in Australian beef cattle. Association for the Advancement of Animal Breeding and Genetics 21, 114–117.

Duval S and Kindermann M. ()2012. Use of nitrooxy organic molecules in feed for reducing enteric methane emissions in ruminants, and/or to improve ruminant performance. International Patent Application WO 2012/084629 A1, World Intellectual Property Organization, Geneve, Switzerland.

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

Fedrizzi F., Cabana H., Ndanga E. and Cobral A. (2018) Biofiltration of methane from cow barns. Effects of climatic conditions and packing media acclimatization. Waste Management 78, 669-676.

Garnsworthy, P., (2004). The environmental impact of fertility in dairy cows: a modelling approach to predict methane and ammonia emissions. Anim. Feed Sci. Technol. 112, 211–223

van Gastelen, S., Dijkstra, J.,and Bannink, A., (2019). Are dietary strategies to mitigate enteric methane emission equally effective across dairy cattle, beef cattle, and sheep? J. Dairy Sci. 102, 6109–6130.

Gerber, P J., Hristov A.N. B. Henderson B., Makkar H., Oh J., Lee C., Meinen R., Montes F., Ott T., Firkins J., Rotz A., Dell C., Adesogan A T., YangW. Z., Tricarico J.M., Kebreab E., Waghorn G., Dijkstra J.and Oosting S. (2013) Technical options for the mitigation of direct methane and nitrous oxide emissions from livestock: a review. Animal 7:s2, pp 220–234

Gill, M, Smith, P. and Wilkinson J.M. (2009) Mitigating climate change: the role of domestic livestock Animal (2010), 4:3, 323–333

Girard M, Ramirez AA, Buelna G and Heitz M (2011). Biofiltration of methane at low concentrations representative of the piggery industry – influence of the methane and nitrogen concentrations. Chemical Engineering Journal 168, 151–158.

Grawert T, Hohmann H P, Kindermann M, Duval S, Bacher A, Fischer M. (2014). Inhibition of methyl-CoM reductase from Methanobrevibacter ruminantium by 2-bromoethanesulfonate. J Agric Food Chem. 62:12487–1249.

Goopy, J.P. (2019). Creating a low enteric methane emission ruminant: what is the evidence of success to the present and prospects for developing economies? Anim. Prod. Sci. 59 (10), 1769–1776.

Grossi G., Pietro Goglio P., Vitali A. and Williams A. J. (2019). Livestock and climate change: impact of livestock on climate and mitigation strategies Animal Frontiers. 9, 69-76.

Haskell, M.J., Rennie, L.J., Bowell, V.A., M. J. Bell, M.J., and A. B. Lawrence A.B. (2006) Housing System, Milk Production, and Zero-Grazing Effects on Lameness and Leg Injury in Dairy Cows. J. Dairy Sci. 89, 4259–4266

Hassanat, F., Gervais, R., Julien, C., Massé, D.I., Lettat, A., Chouinard, P.Y., Petit, H.V. and Benchaar, C. (2013) Replacing Alfalfa Silage with Corn Silage in Dairy Cow Diets: Effects on Enteric Methane Production, Ruminal Fermentation, Digestion, N Balance, and Milk Production. Journal of Dairy Science, 96, 4553-4567.

R.S. Hegarty, J.P. Goopy, R.M. Herd, B. McCorkell (2007) Cattle selected for lower residual feed intake have reduced daily methane production. J. Anim. Sci., 85, 1479-1486

B.J. Heins, L.B. Hansen, A.J. Seykora. (2006) Production of Pure Holsteins Versus Crossbreds of Holstein with Normande, Montbeliarde, and Scandinavian Red. Journal of Dairy Science, 89, 2799-2804,

Herrero, M., R. Conant, R., Havlik P., Hristov,A N., Smith, P., Gerber, P.J., Gill, M., Butterbach-Bahl, K. and Henderson B, et al. 2016. Greenhouse gas mitigation potentials in the livestock sector. Nat. Clim. Change. 6:452–461.

Hristov, A. N., Oh, J. Giallongo, F., Frederick, T.W., Harper, M. T., Weeks, H.L., Branco, A.F., Moate, P.J., Deighton, M.H., Williams, S.R., Kindermann, M.and Duva S.I. (2015). An inhibitor persistently decreased enteric methane emission from dairy cows with no negative effect on milk production. Proc. Natl. Acad. Sci. USA 112:10663–10668

Hristov, A.N., Ott, T., Tricarico, J., Rotz, C.A., Waghorn, G., Adesogan, A., Dijkstra, J., Montes, F. Oh, J., Kebreab, E., Oosting, S., Gerber, P. J. Henderson, B., Makkar, H.and Firkins, J.. (2013). SPECIAL TOPICS -- Mitigation of methane and nitrous oxide emissions from animal operations: III. A review of animal management mitigation options. Journal of Animal Science. 91. 5095-5113.

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

Jayanegara, A, Ageng Sarwono,K, Kondo,M, Matsui,M, Ridla,M, Laconi, E B, & Nahrowi (2018) Use of 3-nitrooxypropanol as feed additive for mitigating enteric methane emissions from ruminants: a meta-analysis, Italian Journal of Animal Science, 17:3, 650-656.

Jayasundara, S., Ranga Niroshan Appuhamy, J. A. D, Kebreab E. and Wagner-Riddle C. (2016) Methane and nitrous oxide emissions from Canadian dairy farms and mitigation options. An updated review. Canadian Journal of Animal Science 96, 306-331.

Johnson KA and Johnson DE (1995). Methane emissions from cattle. Journal of Animal Science 73, 2483–2492.

Jones A. K., Jones D.L. and Cross P. (2014). The carbon footprint of lamb: sources of variation and opportunities for mitigation. Agricultural Systems 123: 97-107.

Jones HE, Warkup CC, Williams A and Audsley E (2008). The effect of genetic improvement on emissions from livestock systems. Conference of the 59th Annual Meeting of the European Association of Animal Production, 24–27 August 2008, Vilnius, Lithuania, Session 05, no. 6, p. 28

Knapp, J. R., G. L. Laur, P. A. Vadas, W. P. Weiss, and J. M. Tricarico. (2014). Invited review: enteric methane in dairy cattle production: quantifying the opportunities and impact of reducing emissions. J. Dairy Sci. 97:3231– 3261.

Kobyaschi, Y., (2010). Abatement of methane production from ruminants: trends in the manipulation of rumen fermentation. Asian-Aust. J. Anim. Sci. 23, 410–416

Lee MRF, Theobald VJ, Gordon N, Leyland M, Tweed JKS, Fychan R and Scollan ND (2014). The effect of high polyphenol oxidase grass silage on metabolism of polyunsaturated fatty acids and nitrogen across the rumen of beef steers. Journal of Animal Science 92, 5076–5087.

Llonch, P., M. J. Haskell, R. J. Dewhurst, and S. P. Turner. (2017). Current available strategies to mitigate greenhouse gas emissions in livestock systems: an animal welfare perspective. Animal. 11:274–284.

Lovett, D.K., Shalloo, L., Dillon, P., O'Mara, F.P., (2006). A systems approach to quantify greenhouse gas fluxes from pastoral dairy production as affected by management regime. Agric. Syst. 88, 156–179.

Misselbrook T., del Prado, A. and Chadwick D. (2012). Opportunities for reducing environmental emissions from forage-based dairy farms. Proceedings of XVI Silage Conference 113-125

Moate, P. J., S. R. O. Williams, G. Grainger, M. C. Hannah, E. N. Ponnampalam, and R. J. Eckard. (2011). Influence of cold-pressed canola, brewer's grains and hominy meal as dietary supplements suitable for reducing enteric methane emissions from lactating dairy cows. Anim. Feed. Sci. Technol. 166–167:254–264.

Montes, F., R. Meinen, C. Dell, A. Rotz, A. N. Hristov, J. Oh, G. Waghorn, P. J. Gerber, B. Henderson, H. P. Makkar, et al. (2013). Special topics—mitigation of methane and nitrous oxide emissions from animal operations: II. A review of manure management mitigation options. J. Anim. Sci. 91:5070–5094.

Moorby, J. M., Evans, R.T., Scollan, N.D., MacRae, J.C.and Theodorou.M.K. (2006). Increased concentration of water-soluble carbohydrate in perennial ryegrass (Lolium perenne L.). Evaluation in dairy cows in early lactation. Grass Forage Sci. 61:52–59.

Moxey A and Thomson S. (2021). Estimated dairy emissions and their mitigation in the Smart Inventory. Scottish Government. Greenhouse gas inventory: estimated dairy emissions and their mitigation - gov.scot (www.gov.scot)

Nichols JR, Schingoethe DJ, Maiga HA, Briuk MJ and Piepenbrink MS (1998). Evaluation of corn distillers grains and ruminally protected lysine and methionine for lactating dairy cows. Journal of Dairy Science 81, 482–491

O'Brien D. , Herron J., Andurand J., Caré S., Martinez P., Migliorati L., Moro M., Pirlo G. Dollé J.B. (2020) LIFE BEEF CARBON: a common framework for quantifying grass and corn based beef farms' carbon footprints. Animal 14:4, 834–845

Odongo, N.E., Bagg, R., Vessie, G., Dick, P., Or-Rashid, M.M., Hook, S.E., Gray, J.T., Kebreab, E., France, J., McBride, B.W.,(2007). Long-term effects of feeding monensin on methane production in lactating dairy cows. J. Dairy Sci. 90 (4), 1781–1788.

O'Mara, F.P., (2011). The significance of livestock as a contributor to global greenhouse gas emissions today and in the near future. Anim. Feed Sci. Technol. 166–167, 7–15.

Ouatahar, L., Bannink, G., Lanigan, G., and Amon, B. (2021) Modelling the effect of feed management on greenhouse and nitrogen emissions in cattle farming systems. Science of the Total Environment. 776.

Parsons A.J, Rasmussen S and Rowarth J. (2011). High Sugar Grasses CAB Reviews: Perspectives in Agriculture, Veterinary Science, Nutrition and Natural Resources

Pelletier, N., Pirog, R. and Rasmussen, R. (2010) Comparative life cycle environmental impacts of three beef production strategies in the Upper Midwestern United States. Agricultural Systems, 103, 380-389.

Pickering et al. (2013) “Consensus methods for breeding low methane emitting animals” a White Paper prepared by the Animal Selection, Genetics and Genomics Network of the Livestock Research Group of Global Research Alliance for reducing greenhouse gases from agriculture. http://www.asggn.org/publications,listing,95, mpwg -white-paper.htm

C. S. Pinares-Patino , S. M. Hickey, E. A. Young, K. G. Dodds, S. MacLean, G. Molano, E. Sandoval , H. Kjestrup, R. Harland, C. Hunt, N. K. Pickering and J. C. McEwan. (2013) Heritability estimates of methane emissions from sheep. Animal, 7:316–321.

Samsonstuen, S., Åby, B.A., Crosson, P., Beauchemin, K.A., Bonesmo, H. and Aass, L. (2019) Farm scale modelling of greenhouse gas emissions from semi-intensive suckler cow beef production. Agricultural Systems, 176, 1-9.

Simm G, Bu¨ nger L, Villanueva B and Hill WG 2004. Limits to yield of farm species: genetic improvement of livestock. In Yields of farmed species. Constraints and opportunities in the 21st century (ed. R Sylvester-Bradley and J Wiseman), pp. 123–141. Nottingham University Press, Nottingham, UK

Smith, P., Martino, D., Cai, Gwary, D., Janzen, H., Kumar, P., et al., (2007). Policy and technological constraints to implementation of greenhouse gas mitigation options in agriculture. Agric. Ecosyst. Environ. 118, 6–28

Sun K, Liu H, Fan H, Liu T, Zheng C. (2021). Research progress on the application of feed additives in ruminal methane emission reduction: a review. PeerJ 9:e11151

Van der Heyden, C., Demeyer, P., & Volcke, E. I. P. (2015). Mitigating emissions from pig and poultry housing facilities through air scrubbers and biofilters: State-of-the-art and perspectives. Biosystems Engineering, 134, 74–93

Wall, E., Simm,G., and Moran.D. (2010). Developing breeding schemes to assist mitigation of greenhouse gas emissions. Animal 4:366– 376.

Wallace, R.J, G. Sasson, P. C. Garnsworthy, I. Tapio, E. Gregson, P. Bani, P. Huhtanen, A. R. Bayat, F. Strozzi, F. Biscarini, T. J. Snelling, N. Saunders, S. L. Potterton, J. Craigon, A. Minuti, E. Trevisi, M. L. Callegari, F. P. Cappelli, E. H. Cabezas-Garcia, J. Vilkki, C. Pinares-Patino, K.O. Fliegerová, J. Mrázek, H. Sechovcová, J. Kopečný, A. Bonin, F. Boyer, P. Taberlet, F. Kokou, E.Halperin, J. L. Williams, K. J. Shingfield, I. Mizrahi. (2019) A heritable subset of the core rumen microbiome dictates dairy cow productivity and emissions. Sci. Adv. 5, eaav8391.

Wilkinson J.M. and Lee M.R.F. (2017) Review: Use of human edible animal feed by ruminant livestock. Animal (2018), 12, 1735–1743

Williams, A., Chatterton J., G. Hateley G., Curwen A. Elliott J. (2015). A systems-life cycle assessment approach to modelling the impact of improvements in cattle health on greenhouse gas emissions. Adv. Anim. Biosci. 6:29–31.

Zehetmeier, M., J. Baudracco, H. Hoffmann and A. Heissenhuber, (2012). Does increasing milk yield per cow reduce greenhouse gas emissions? A system approach. Animal 6: 154-166.

Chapter 7

AHDB (2015) https://ahdb.org.uk/beef-lamb

AHDB (2020) RB209 Section 1 Principles of nutrient management and fertiliser use

Balmford, A. et al. (2018) The environmental costs and benefits of high yield farming. Nature Sustainability 1, 477-485

Barnes A., Eory V., Buys G., Stockdale E., Macleod M., Sykes, A., Norton L and Carter N. (2021) Delivering clean growth through sustainable Intensification. SRUC-NIAB-UKCEH

Capper, J.L., Pullman, W.A., Cady, R.A. and Dales, D.E. (2009) The Relationship Between Cow Production and Environmental Impact. Department of Animal Science, Cornell University, Ithaca, NY

DEFRA 2020 Greenhouse gas emissions from agriculture indicators - GOV.UK (www.gov.uk)

Eory, V., Maire, J., Macleod, M., Sykes, A., Barnes, A., Rees,, R., Topp, C. and Wall, E. (2020) Non-CO2 abatement in the UK agriculture sector by 2050. Report to UKCCC.

Eory, V., Topp, K., Rees, B., Leinonen, I., Maire, J., Macleaod, M., Sykes, A. and Wall, E. (2020). Marginal abatement cost curve for Scottish agriculture. www.climatexchange.org.uk

MacLeod, M., Moran, D., McVittie, A., Rees, B., Jones, G., Harris,, D., Antony S., Wall, E.,Eory V., Barnes A., Topp, K., Ball, B., Hoad, S. and Eory, L. (2010) Review and update of UK marginal abatement cost curves for agriculture. Report to UKCCC

Moran, D., Eory, V., McVittie,A., Wall, E., Topp, K., McCracken, D. and Haskell, M. Wider implications of greenhouse gas mitigation measures in English agriculture - Defra AC0226

Chapter 8

Carlile, R., and Garnett, T. (2021). What is agroecology? TABLE Explainer Series. TABLE, University of Oxford, Swedish University of Agricultural Sciences and Wageningen University & Research

Danish Agriculture and Food Council (2020) Climate-neutral 2050 Climate neutral 2050 (agricultureandfood.dk)

Food, Farming and Countryside Commission https://ffcc.co.uk/home

Martineau, A.H., et al. (2019). Annex 7: Systems approach to GHG emissions reduction. ERAMMP Report to Welsh Government (Contract C210/2016/2017) (CEH NEC06297)

ERAMMP Programme Office
UKCEH Bangor
Environment Centre Wales
Deiniol Road
Bangor, Gwynedd
LL57 2UW
+ 44 (0)1248 374500
erammp@ceh.ac.uk

www.erammp.wales
www.erammp.cymru