# A Desktop Review of Greenhouse Gas (GHG) Emissions

# Project reference: F08/019

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**Acknowledgments:**
The authors are very grateful for the helpful suggestions and comments about this review made by Dr David Garwes.

**Abbreviations used:**
AD, anaerobic digestion
$CH_4$, methane
$CO_2$, carbon dioxide
$CO_2e$, carbon dioxide equivalent: $CO_2$ = 1, $CH_4$ = 25, $N_2O$ = 310
FACTS, Fertiliser Adviser Certification and Training Scheme
GHG, greenhouse gas
ha, hectare
IPCC, Intergovernmental Panel on Climate Change
kg, kilogramme
kt, kilotonne
lu, livestock unit
LULUCF, land use, land use change and forestry
N, nitrogen
$N_2O$, nitrous oxide
$NH_3$, ammonia
$NH_4$, ammonium
$NO_3$, nitrate
NVZ, Nitrate Vulnerable Zone
t, tonne

**January 2009**

# CONTENTS

# 1 RECOMMENDATIONS TO FARMERS AND INDUSTRY

## 1.1 METHANE

Methane is a potent greenhouse gas and almost half of the UK's emissions come from the livestock sector. The other major contributor to national emissions is landfill but this source has been declining at a much greater speed compared with emissions from livestock. Therefore, there has been increasing pressure on the industry to act in response to the growing consensus on climate change and the associated causes. Ruminant livestock produce methane as a result of fermentation of feed in the rumen. Methods known to reduce emissions include:

- Changes to diet:
    - Feeding increased levels of starch
    - Feeding supplementary dietary fat sources
    - Reducing the proportion of fibre in the diet
- Certain feed additives also show promise as methods for reduction including:
    - Organic acids
    - Yeast
    - Plant extracts and essential oils
    - However, many additives yield inconsistent results or are currently restricted in the EU (ionophores)
- Management systems geared towards high production per cow tend to reduce the emission of methane per litre of milk as long as fertility is not greatly compromised.
- Reductions in the herd replacement rate help to reduce the emissions burden related to the rearing of youngstock.
- Changes to manure handling practices including the adoption of anaerobic digesters can yield improvements in energy efficiency as well as reduction in methane output. However, emissions from manure tend to be secondary to those associated with fermentation of feed in the rumen.

Many mathematical models have been developed to predict emissions of methane from dairy cows. However, the current national standards by which inventories are calculated are insensitive to many of the factors affecting emissions. Change is required to reflect our increased level of knowledge based on recent research within the national emissions inventory.

## 1.2 NITROUS OXIDE

Nitrous oxide is a potent greenhouse gas and is mainly a product of the soil microbial processes, nitrification and denitrification. Factors which favour nitrous oxide generation include a supply of readily available nitrogen and carbon, localised anaerobic conditions, and high soil temperature and moisture conditions. Hence, the major direct sources of nitrous oxide emissions on dairy farms include; applications of inorganic fertiliser nitrogen, manure spreading and deposition of urine/faeces by grazing stock. There are also nitrous oxide emissions (indirect) associated with nitrate leaching and nitrogen deposition. Emissions of nitrous oxide from the dairy sector can be reduced by:

- Do not exceed crop N requirements (RB209/PLANET).

- Make full allowance of manure N supply (MANNER).
- Spread manure at appropriate times/conditions.
- Increase livestock nutrient use efficiency.

## 1.3 AMMONIA

Agriculture is the major source of ammonia emissions to the atmosphere, and the dairy sector accounts for approximately one third of total agricultural emission. Emissions of ammonia from the dairy sector can be reduced by:

- The use of band spreading or injection application techniques for slurry application to land.
- Rapid (ideally within 4 h, certainly within 24h) incorporation of slurry and FYM applied to arable land.
- Allowing slurry stores to develop a manageable crust.
- Ensuring that cattle dietary N intake does not greatly exceed requirement.

There are other potential mitigation methods, such as washing of dairy cow collecting yards, switching from a slurry-based to straw-based manure management system or increasing the amount of bedding used in a straw-bedded housing system, but implementation of these will depend on the specific circumstances of the farm. It is important that all potential pollutant and production outputs are considered when planning any ammonia mitigation strategy.

## 1.4 CARBON DIOXIDE

Emissions of carbon dioxide from dairying may be reduced by:

- Careful use and monitoring of electricity use in water heating, refrigeration of milk and lighting. Separate metering of the dairy enterprise from the rest of the farm will assist this.
- Reducing fuel use in transport and cultivations where possible. This may be through planning for longer use of temporary grass through changes in seed mixtures to reduce the frequency of cultivations or a different approach to cultivations or the use of undersown grass and forage crops.
- Although maize is grown widely, there are still opportunities to extend its use and reduce emissions from forage production.
- Cafeful planning of agricultural liming where there may be an opportunity to reduce applications using soil analysis and careful targeting of agricultural lime applications.
- Using renewable forms of energy. This will usually be at a financial cost, but energy bills will be reduced over the long term.

## 1.5 CARBON STORAGE

Opportunities to *increase* C storage within dairy farming systems are through the conversion of tillage land, and to a lesser extent grassland, to farm woodland which will be recognised within the IPCC inventory, and through the conversion of tillage land into *permanent* grassland.

In contrast, situations that are likely to *decrease* C storage within dairy farming systems include the clearance of farm woodland and conversion to tillage land or grassland and the cultivation of permanent grassland and conversion to tillage land.

## 1.6 DAIRY SYSTEMS

Emissions of GHG by dairy cows could be reduced by:

- Improving the fertility of the national herd. This will reduce the number of replacements needed and reduce the methane and ammonia emissions from dairy farming.
- Reducing the incidences of lameness and mastitis, which would also contribute to a reduction in the number of replacements required.
- Improved accuracy of ration formulation to more closely meet the cow's needs.
- Increasing use of mixed diets with maize and or whole crop cereals to improve N utilisation and reduce methane.
- Increasing productivity of dairy cows. Although this would not affect output per cow, it would reduce output per unit of product.

Large changes in the industry over the last 20 years are likely to have reduced GHGs and ammonia emissions (reduced numbers of cattle, reduced fertiliser use, increased use of maize and more accurate ration formulation), but the rate of change in the future may not be as fast.

# 2 EXECUTIVE SUMMARY

## 2.1 METHANE

Agriculture is the second largest contributor to UK methane emissions after landfill sites. 90% of agricultural methane arises from enteric fermentation by ruminant livestock of which dairy animals comprise a large proportion. A small percentage of emissions from livestock production arise from manure storage and handling procedures whilst unmanaged areas of pasture (gateways, ditches etc.) may also contribute towards total methane production.

Overall, there has been a decline in national methane emissions according to the UK inventory but this reduction reflects the decline in the size of the UK dairy herd rather than targeted mitigation measures on behalf of the dairy industry. Various mitigation strategies have been studied and invariably there is an incentive to reduce emissions from an economic perspective with methane representing a loss of valuable feed energy, as well as to meet environmental objectives. Dietary nutrient composition can be altered to encourage a more glucogenic fermentation with a corresponding decline in methane yield. Dietary fat sources can be added to deliver both a rumen protected energy source and, in the case of unsaturated fatty acids, an alternative hydrogen sink at the expense of carbon dioxide reduction to methane. The efficacy of such strategies is affected by many factors, such as the composition of the basal diet, production level and the form of any supplemental nutrients such as fat sources. In particular, detrimental effects on dry matter intake and milk yield have been observed in certain situations. However, careful use of nutritional intervention strategies can deliver reasonable reductions in methane emission without serious compromise to animal production. Increases in dietary starch, reductions in fibre and increases in dietary fat all promise to deliver a less methanogenic fermentation.

Genetic selection of animals that display tendencies for reduced emissions may be a worthwhile avenue of further research, as will the selection of forages with nutrient profiles suited to more glucogenic rumen fermentation. Various feed additives have been proposed as having a direct and beneficial effect on rumen fermentation. These include plant extracts, inoculation with microbes, ionophores, anti-methanogenic vaccines and organic acids.

Changes to manure management may offer another route to reduced emissions. In particular, anaerobic digestion offers the ability to harvest the methane energy and convert it into a source of power.

Herd management policies resulting in a reduced replacement rate will lead to a reduction in the number of replacement animals required to support the milking herd. Inevitably this will help to lower herd emissions of methane when expressed as methane per litre of milk produced. In a similar manner, management for increasing milk yield will tend to lead to higher methane emissions per animal, but a reduced overall emission when expressed per litre of milk output.

Various models have been presented in the literature with the objective of predicting methane output. They emphasise the strong link between dry matter intake and methane emission although earlier models present the relationship as linear whilst more recent work has demonstrated the diminishing return in methane output per unit of intake as intake increases. Selection of a model appropriate to the type of diet or farming system is crucial to avoid misapplication of existing models and therefore misleading results. IPCC emission factors are easy to apply but are generally not powerful enough to consider the range of nutritional factors that influence emissions. The more advanced statistical models promise to be more useful in this respect whilst further work will ultimately produce dynamic models suitable for on-farm application that will be capable of considering a far wider range of factors.

The review highlights the key areas where further research is needed in search of more effective methods to increase efficiency and reduce the emission of methane from the dairy sector.

## 2.2 NITROUS OXIDE

Agriculture is responsible for *ca*. 66% of the UK's nitrous oxide ($N_2O$) emission with 61% of this $N_2O$ emitted from agricultural sources arising from direct soil emissions and 32% from indirect sources (N deposition and nitrate ($NO_3$) leached).

Annual emissions of $N_2O$ have been decreasing due to reduced use of inorganic N fertilisers and declining numbers of livestock. Emissions have reduced by *ca*. 21% between 1990 and 2006 to an emission of 81.8 kt.

The dairy sector was responsible for *ca*. 17% of the UKs agricultural emission of $N_2O$ in 2005.

The main sources of direct $N_2O$ emissions on dairy farms are inorganic N fertiliser use, manure applications and urine deposited during grazing. Emissions from these sources are highly variable and depend on inorganic N supply, availability of microbially available carbon, soil type and degree of aeration.

The main indirect source of $N_2O$ emissions on dairy farms is that associated with nitrate leaching.

Current best practices to reduce $N_2O$ emissions include; not exceeding crop N requirements, making full allowance of manure N supply, spreading manure at appropriate times/conditions, increasing livestock nutrient use efficiency and making use of improved genetic resources. Implementation of all these management practices could reduce $N_2O$ emissions from the dairy sector by *ca*. 8%.

A range of greenhouse gas calculator tools and models exist. Most are based on IPCC default values.

## 2.3 AMMONIA

Agriculture is the major source of ammonia emissions to the atmosphere, accounting for >85% of total UK emissions. Ammonia is important with respect to greenhouse gas emissions as indirect emissions of nitrous oxide can occur from deposition of atmospheric ammonia. Additionally, mitigation of ammonia emissions may influence subsequent emissions of nitrous oxide. It is important therefore that consideration of ammonia emission is included in any review or modelling of greenhouse gas emissions from dairying. The UK is required under international legislation to reduce emissions below a target ceiling of 297 kt $NH_3$ by 2010, and a new, lower, target is being negotiated for 2020. It is very likely that some form of mitigation will be required to meet the 2020 target. The dairy sector accounts for approximately one third of the ammonia emissions from agriculture, with housing (including outdoor yards) and land application of manures as the major sources.

Options for mitigation include dietary manipulation (particularly reducing excess intake of crude protein), improved cleaning of cubicle house floors and outdoor yards, increased bedding in straw-bedded systems, covering of manure storage facilities, application of slurry using band spreading or injection techniques and rapid incorporation of manure (slurry and FYM) applied to arable land. Of these, the options for mitigation at land spreading represent those which could most readily be implemented and have an immediate effect. Dietary manipulation offers great potential of reducing ammonia emissions and other N losses throughout the entire management chain, but there are challenges to implementing strict dietary control in largely forage-based feeding systems.

A number of models exist for predicting ammonia emissions from agricultural sources at a range of scales and complexities. Specific to the dairy sector, the $SIMS_{DAIRY}$ model can be used to estimate emissions for given scenarios (including mitigation options) and also includes other pollutant and production outputs. The national inventory model (NARSES) can also be used at a farm-scale.

While the knowledge base regarding ammonia emissions from agriculture has improved greatly in recent years, a number of gaps still exist, including an understanding of the nitrogen losses and transformations within solid manure (FYM) systems, quantification of slurry infiltration into soils (which has a large impact on subsequent ammonia emissions), derivation of a robust emission factor for slurry lagoons and improved management and activity data, particularly relating to dietary N intake.

## 2.4 CARBON DIOXIDE

Emissions of carbon dioxide from dairying are created by electricity use in water heating, refrigeration of milk and lighting and fuel use including transport and cultivations. In overall terms, they are not significant when compared with overall UK greenhouse gas emissions.

Estimates vary, but a figure of 3.638 t $CO_2e$ per cow would appear to be reasonable for a cow producing 6,500 litres milk plus followers.

A further source of $CO_2$ emissions is that of agricultural liming, and there may be an opportunity to reduce these by soil analysis and carefully targeting the use of agricultural lime.

## 2.5 CARBON STORAGE

There is growing emphasis being placed on soil carbon (C) storage in the mitigation of climate change and various measures are being explored to determine how best soil organic C storage (SOC) levels can be increased.

Data from the National Soils Inventory (NSI) indicates that grassland topsoil's typically contain 4.2% SOC (1995/96 data), compared with 2.8% in arable/ley soils. The conversion of tillage land to grassland can therefore result in increased SOC storage, with estimates in the range 1.1 to 7.0 $tCO_2e$/ha/year.

By contrast, the conversion of grassland or permanent cropping to tillage cropping has been estimated to result in C losses in the range 2.2 to 6.2 $tCO_2e$/ha/year, largely due to vegetation clearance, increased soil organic matter decomposition rates upon cultivation and losses of C through erosion.

Soil carbon accumulation is reversible, maintaining SOC is dependent on continuing the new management practice/land use indefinitely. One of the main mechanisms of increasing C storage within any farming system is therefore to take land *permanently* out of food production by the creation of farm woodland. Such a land use change has been estimated to increase soil C storage in the range 1.1 to 2.3 t$CO_2$e/ha/year following the conversion of tillage land to forestry. There will also be considerable gains in above ground biomass C (with estimates up to 9.2 t $CO_2$e/ha/yr).

The biggest opportunities to *increase* C storage within dairy farming systems (i.e. to change from the present day baseline) are therefore through the:

- Conversion of tillage land (and to a lesser extent, grassland) to farm woodland (this action will be recognised within the IPCC inventory).
- Conversion of tillage land into *permanent* grassland.

## 2.6 DAIRY SYSTEMS

With regard to the current UK dairy industry, there are a number of aspects of dairy herd management, particularly in relation to feed and fertility management, that could influence GHG and ammonia production.

There has been a 21% reduction in dairy cow numbers in the UK in the last 10 years (1997 to 2007). Over the same period herd size has increased to just over 100 cows and milk yield per cow has risen to 6908 litres/cow. The increased yield has been achieved through improved breeding, nutrition and management. It has been predicted that milk yield per cow will rise to 9000 litres/cow by 2030.

Structural and management changes within the dairy industry will affect GHG and ammonia emissions from dairy farming. A decline in stocking rates will have increased the proportion of forage in dairy cow diets which will have lead to increased methane production. Seasonality of production will also influence GHG emissions. Extended periods of cattle housing are associated with higher ammonia emissions but lower methane, since cows are more dependent on concentrates than in extended grazing systems. Fertiliser use has also declined over recent years with consequent reductions in forage-N and reduced N-excretion by dairy cows. Diet manipulation can influence both methane and N excretion and there is scope to decrease atmospheric emissions by changes in diet formulation.

Improvements in plant breeding (high sugar grasses) have potential for increasing the utilisation of forage N, but further research is needed to confirm overall effects on the environment. Increased use of legumes will result in a reduction in the amount of artificial fertiliser used, but nitrate leaching under legumes can be as high as soils fertilised with artificial N. However improved milk yields on mixed legume/grass silages can reduce the cost of milk production. The increasing area of maize silage fed (now producing in the order of 2.5 t per cow) has led to improved N utilisation by dairy cows and lower N excretion. The area of maize is likely to increase as it becomes more possible to grow maize consistently in the west and north of the country.

The way in which feed is offered to cows also allows scope for improved efficiency of feed use. Use of TMR machinery allows more accurate group feeding of dairy cows with consequent reductions in N excretion etc.

Declining herd fertility has resulted in high replacement rates and high numbers of heifers in order to maintain herd size.  Improving herd health (i.e. reducing the incidence if lameness and mastitis) and fertility would result in lower numbers of replacements reared.  This will result in reductions in methane and ammonia emissions from dairy herds by simply reducing stock numbers.

## 2.7 ON-FARM SOFTWARE

The recent public awareness of climate change and the industry's desire for increased efficiency has led to the development of several commercial tools aimed at benchmarking the performance of individual businesses against agreed standards or other producers.  The main software tools that are currently available to industry have been summarised in the report although a thorough analysis has been prevented due to some concerns over commercially sensitive information relating to the calculations involved.  However, these models are now being developed against the background of the publication of the PAS2050 standard which aims to add rigour and consistency to calculation methods.  This standard is not exclusive to the dairy sector and applies to UK businesses generally.  The Carbon Trust provides certification against this standard for emissions estimates from software tools that are submitted into this process.  Of those relevant to the dairy industry, currently one commercial software program has gained Carbon Trust certification for its carbon footprint estimates and two others are in the process of gaining the standard for their predictions.  The review team believe that this standard is an important step towards agreeing the best practice for estimating emissions from farms.  However, there remain some concerns over the reliance on crude Tier 1 and Tier 2 IPCC methodology.  Steps to update the PAS2050 standard would be beneficial for those interested in reflecting current mitigation technology within the available software tools.

# 3 BODY OF THE REPORT

## 3.1 OBJECTIVES

The aim of the project is to review appropriate worldwide knowledge on the contribution of the range of GB dairy farming systems to emissions of GHGs and ammonia. Through a detailed understanding of the sources of emissions within these dairy systems, the work will highlight where mitigation methods would be practical and cost-effective. The project will also consider how climate change could influence both emissions of these gases *per se*, but also via adaptation of the industry and adoption of different systems, e.g. a move to more zero grazed systems or an increased production of forage maize. The project will aim to provide the robust knowledge required for further research on modelling GHG and ammonia emissions from dairy systems.

1. Quantifying the sources of GHGs and ammonia emissions from a cross section of dairy systems in GB representing the broad range of management practices and production objectives.
2. Understanding the impact of key management factors influencing the emissions of GHGs and ammonia, e.g. factors such as nutrition, milk yield, calving pattern, inorganic and organic fertiliser inputs.
3. Highlighting the potential impacts of climate change on GHG and ammonia emissions.
4. Summarising how carbon storage might ‘balance’ GHG emissions on dairy farms.
5. Quantifying the potential to reduce GHG and ammonia emissions from production operations across the range of dairy systems, providing an indication of the costs to businesses.
6. Assessing the potential for on-farm benchmarking of GHG and ammonia emissions including a comparison of existing models and software packages.
7. Listing and prioritising any research required to address key gaps in knowledge.
8. Providing information that will feed directly into subsequent research on modelling GHG and ammonia emissions on farm.

## 3.2 MILESTONES

| No. | Milestone | Taking place / completed week ending |
|---|---|---|
| 1 | Initial meetings of project team, DairyCo and the GHG Modelling team to flesh out the boundaries of the review and identify the needs of the modelling team (week 1) | 18th July 2008 |
| 2 | Physical structure of the technical report circulated to project team, DairyCo and modelling team (week 2) | 25th July 2008 |
| 3 | Co-ordinators have identified sources of gases (week 10) | 19th September 2008 |
| 4 | Co-ordinators have completed written reviews of information on gases (week 15) | 24th October 2008 |
| 5 | Second meeting of project and modelling teams to discuss progress (week 15) | 24th October 2008 |
| 6 | Co-ordinators have identified information on mitigation (week 17) | 7th November 2008 |
| 7 | Co-ordinators have completed written reviews of information on mitigation (week 20) | 28th November 2008 |
| 8 | Draft report completed and circulated to project team (week 21) | 5th December 2008 |
| 9 | Final report and presentation of findings to DairyCo (week 23) | 19th December 2008 |

## 3.3 REVIEW

### 3.3.1 Methane

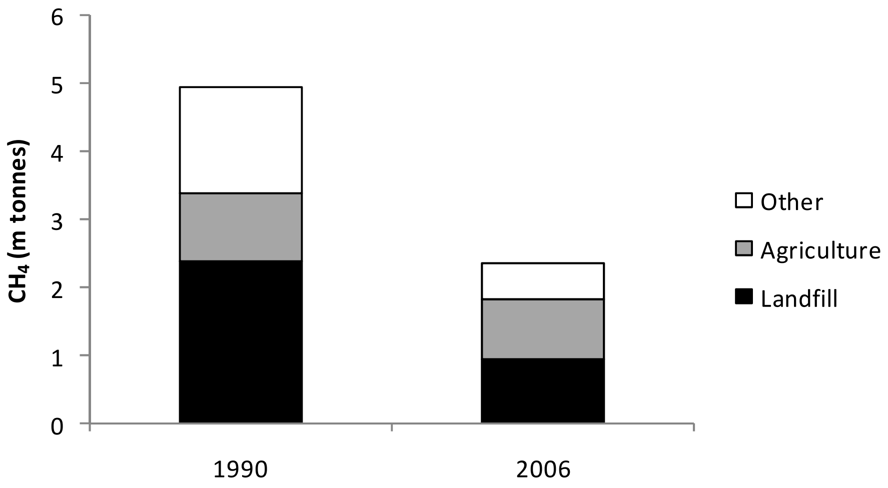
#### 3.3.1.1 Introduction

Methane-rich gasses are produced by the anaerobic degradation of organic material and sources include swamps, marshes, landfills and agriculture. The principal effect of methane on the environment is as a greenhouse gas (GHG) and it is estimated to contribute around 18% of the overall warming potential globally. Methane has a radiative forcing coefficient (RF) of 0.48 W/m$^2$, which is second only to $CO_2$ with a global warming potential (GWP) of 72 if averaged over 20 years, or 25 if averaged over 100 years. Despite its high GWP, methane has a relatively short atmospheric half life which means that action to reduce emissions can lead to significant reductions in atmospheric concentrations over a relatively short time scale. It has been estimated that methane concentrations in the atmosphere have more than doubled over the last two decades, largely due to human-related activities that include rice paddie cultivation, biomass burning, ruminant production systems, landfills, coal mining, natural gas systems and storage of livestock wastes (Hogan et al*.,* 1991). Any reduction in methane emission would be 20–60-fold more effective in reducing global warming than a similar reduction of carbon dioxide emission (Shine et al., 1990).

Agriculture is the second largest contributor to methane emissions in the UK after landfills. Following the signing of the Kyoto treaty in 1997, in which industrialised nations agreed to reduce their emissions of GHG by approximately 5.2% of their 1990 levels, there have been significant reductions in UK total emissions of methane; total methane emissions within the UK in 2006 were 2.43 m tonnes compared with 4.93 m tonnes in 1990, a reduction of approximately 51% (DEFRA, 2008).

In 1990 landfill sites within the UK emitted around 2.4 million tonnes of methane, which accounted for 48% of UK total methane emissions. By 2006 landfill emissions had fallen markedly to 0.93 m tonnes (40% of UK total emissions). In 1990 agriculture emissions stood at 1.02 m tonnes. However, unlike the significant reductions seen in landfill emissions between the years of 1990 and 2006, agricultural emissions only fell by 0.14 m tonnes. Consequently, in 2006 agriculture made a more significant contribution of 38% to total UK methane emissions compared to 21% in 1990 (Figure 1).

Figure 1. United Kingdom land fill and agricultural methane emissions (m tonnes) between 1990 and 2006 (source DEFRA, 2008).




By far the largest source of methane from the agricultural sector is from the enteric fermentation processes attributable to ruminant livestock, which accounts for approximately 90% of UK agricultural annual emissions (Figure 2) and 97% in Australia (Hegarty, 2001). On a global scale, 70-100 Tg/year of the 540 Tg/year of methane produced comes from enteric fermentation and 35 Tg/year is derived from animal wastes (http://www.ecifm.rdg.ac.uk/airpollution.htm). The majority of methane production from livestock is associated with ruminants, which accounts for around 85% of the 1143 kt produced by livestock each year in the UK (Table 1). Conversely small amounts of methane are produced by microbial digestion in the digestive tract of pigs, these and other simple-stomached livestock contribute very little to the enteric emissions from UK livestock.

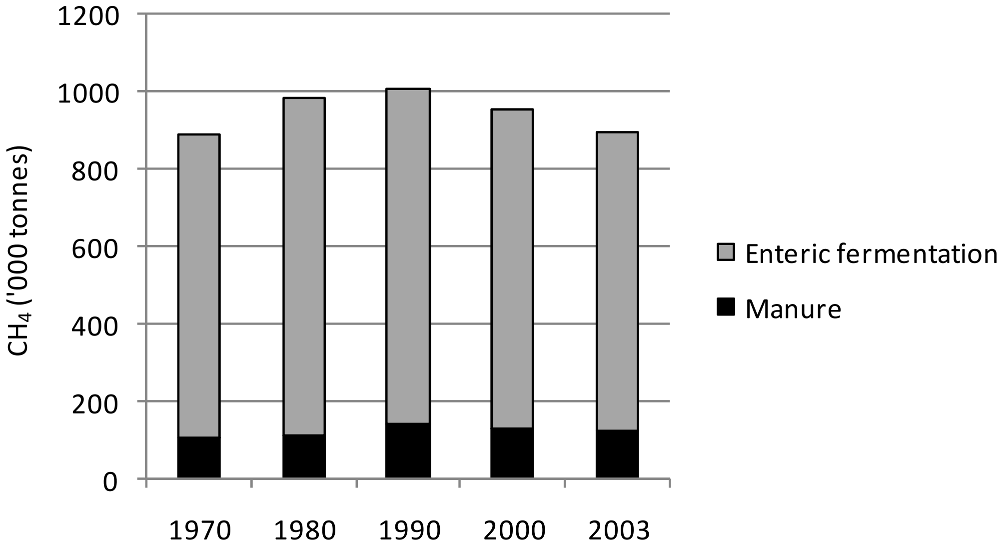
Table 1. Regional breakdown of methane and total GHG generation in the UK, 2006.

| | Methane (%) | *All greenhouse gases (%)* |
|---|---|---|
| England | 69.1 | 77.1 |
| Scotland | 12.7 | 9.1 |
| Wales | 9.9 | 7.8 |
| Northern Ireland | 6.7 | 3.4 |
| Unallocated | 1.6 | 2.6 |

Source: AEA, National Greenhouse Gas Inventory 2008.

The small decline in methane emissions from UK agriculture has been attributed to a reduction in animal numbers rather than changes in animal management practices or diets that have resulted in decreased enteric emission (DEFRA, 2006). Whilst it is true that declining animal numbers will have made a major contribution to reduced national emissions, other factors such as the effect of changing nutrition, management strategy and genetics have received only very limited consideration within the inventory process. Therefore, it may be misleading to assume that the published figures are an accurate representation of reality. Changes, to nutrition arising from management trends or altered feedstuff availability (e.g. increasing maize acreage, availability of byproduct feeds etc.) will have occurred in recent years and these will have introduced both positive and negative influences on national emissions. As our understanding of the magnitude of these effects improves, it is likely that more reliable estimates of annual emissions will be forthcoming (see modelling section). Combating GHG emissions produced by agriculture in the UK is one of DEFRA's main policies, and it is anticipated that by 2015 Britain will have one-third of milk producers' trialling new technology to cut GHG emissions.

Figure 2. Estimated annual emissions of methane from UK agriculture (National Atmospheric Emissions Inventory, 2005).




### 3.3.1.2 Main sources

Methane is liberated from the anaerobic microbial fermentation of feedstuffs in the gut (around 97%) and faeces (3%) of livestock by methanogenic organisms (methanogens) (Hegarty, 2001). About 87% of enteric methane arises from the reticulo-rumen, while the remainder is produced in the hindgut. A significant portion of the methane, about 89-98%, is absorbed and expired through the lungs, with a small amount being excreted through the anus (Murray et al. 1976; Ominski and Wittenberg 2006). On average the amount of methane produced by a sheep is about 30 litres each day and a dairy cow up to 600 litres per day, however, dietary intake and composition will greatly affect methane production.

#### 3.3.1.2.1 Enteric fermentation

Eructation of methane by cattle begins approximately four weeks after birth when solid feeds are retained in the reticulorumen. At this stage, both fermentation and methane production rates rise rapidly as the reticulorumen develops. Methanogens use the hydrogen and carbon dioxide produced as end products of microbial digestion to generate energy for growth. In this process, organic wastes are degraded in the absence of oxygen to $CO_2$, methane and small quantities of $H_2$, $N_2$ and $H_2S$ (Stafford et al., 1980), In the rumen, methane is produced when the methanogens use excess hydrogen to reduce carbon dioxide. There is day-to-day variation in methane production (Blaxter and Clapperton, 1965), which is thought to arise from changes in the metabolic activity of these microbes.

The availability of hydrogen ions in the rumen is dependent on the proportion of end products generated from fermentation of the ingested feed. Processes that produce propionate and cell dry matter act as net hydrogen proton-using reactions, whereas a reaction that yields acetate results in a net proton increase (Hegarty, 1999).

Therefore, the ratio of glucogenic (propionate, valerate) to lipogenic (acetate and butyrate) volatile fatty acids (VFA) produced in the rumen can be used to estimate hydrogen availability and therefore methane production. Higher glucogenic to lipogenic ratios will lead to lower methane emission and vice versa. Other substrates available to methanogens include formate, acetate, methanol, methylamines, dimethyl sulfide and some alcohols, however, only formate has been documented as an alternative methane precursor in the rumen (Jones, 1991).

It is known that symbiotic relationships exist between methanogens and rumen microflora. Some methanogens are ingested by, and live within protozoa as metabolically active endosymbiots, and these may generate up to 37 % of rumen methane emissions (Findlay et al., 1994) although this figure is likely to be highly variable. Stumm et al. (1982) has estimated that 10 - 20 % of rumen methanogens may be attached to the outer surface of protozoa, with attachment increasing 10 to 100 fold after feeding as compared to before feeding. Hegarty et al. (2008) measured methane emissions from lambs born to defaunated ewes and those born to faunated animals. There was no clear effect of defaunation on methane emission across dietary treatments. On the low protein diet, lambs from defaunated mothers or those defaunated following birth tended to produce more methane than faunated animals. However, this pattern tended to be reversed for lambs fed higher levels of protein.

##### 3.3.1.2.2 Manure storage and handling

Given typical production methods in the UK, indirect methane emissions are only significant for cattle and pigs but not for extensively managed sheep. Depending on the storage arrangements an estimate of 1.5 kg methane/tonne of slurry/year is typical (Costigan, 1993; Safley and Westerman, 1988). The rate of methane production from stored manure depends on a variety of factors, including animal species, ration, age of animal, collection method, storage period, temperature (daily and seasonal), amount of foreign material (i.e. bedding) incorporated into the waste, manure characteristics (e.g. the amount of volatile solids and effluent concentrations from liquids systems), and the amount of manure left in the storage facility (methaogenic inoculum), (Chen et al., 1988; IPCC, 2006).

When livestock manure is applied to land to improve soil quality, as the manure breaks down it causes methane to be released into the atmosphere. Overall, manure contributes between 11 and 14% to total methane emission from livestock (Külling et al., 2001).

Methane emissions from livestock and the slurry associated with their intensive management are relatively well quantified (Table 2) and the options for reducing emissions (either by management regime changes or by reducing herd size) are clear (see later). In contrast, emissions from other land use related sources are in many cases less well known.

Table 2. Methane emissions per head from UK livestock.

| | Direct | | Indirect | | | | Total |
|---|---|---|---|---|---|---|---|
| | Methane (kg/year) | Weighted | Slurry output (tones/head/year) | % Housed | % Slurry stored | Methane (kg/year) | Methane (kg/year) |
| Cattle beef | 65 | 55.12 | 12.4 | 50 | 20 | 1.42 | 56.54 |
| Cattle < 2 years | 51.0 | | 5.5 | 75 | 20 | | |
| Dairy in milk | 95.0 | 87.56 | 20.8 | 50 | 80 | 9.85 | 97.41 |
| Dairy not in milk | 65.0 | | 12.4 | 50 | 20 | | |
| Pigs | 1.5 | | 1.3 | 100 | 75 | 1.42 | 2.94 |
| Sheep | 8.0 | | 0.8-1.5 | 0 | 0 | 0 | 8.00 |

Source: Based on Costigan (1993).
Notes: Emission rate of $CH_4$ from slurry under UK conditions and typical management regimes is 1.5 kg.
$CH_4$ per tonne slurry per year (see text). Single estimates for beef and dairy are weighted for the UK herd.

#### 3.3.1.2.3 Soils and land use

Soils can act as an effective sink for both atmospheric methane and for methane produced in deeper soil layers. Consumption of methane by methanotrophic soil organisms depends on the volume of air-filled pores, and this is influenced by soil type, soil wetness and management history, and as these factors are variable it means that methane emissions from soils are also variable. Generally the consumption of methane by methanotrophic organisms is a much more extensive process than emissions by methanogens (involved in enteric fermentation) in the UK. However, within a grassland production system, the contribution of these effects is minimal compared with the potential generation of methane by grazing animals. For example, the amounts of methane emitted or taken up by grassland swards were estimated at 1.53 and 2.73 g methane carbon/ha/day$^{-1}$ compared with the average emissions of 17.9 and 74.5 g methane carbon/day for lambs and calves, respectively (Defra project report CC0206).

Unmanaged pasture areas (e.g. seepage areas of effluent from manure, poached areas under grazing, feeding/watering areas, gateways, tracks and ditches, streams, wetland areas and silage bales) made significant contribution to methane emissions from an intensive dairy farm (Matthews et al., 2006). However, further work is required to quantify more accurately the extent of emissions from these sources and the relationship with their physico-chemical characteristics in order to improve modelling and prediction, and also to consider how changes in farm practice and management could reduce methane emissions.

#### 3.3.1.2.4 Measurement of emissions

Kebreab et al. (2006) reviewed the various methods that have been used to measure methane emissions from livestock. They include respiration chambers, polytunnels, portable analysers, gas tracers and isotope dilution in the rumen. The strengths and weaknesses of each technique determine which is appropriate for any given circumstance. For example, whilst indirect respiration calorimeter chambers are considered to have good accuracy, they are inappropriate for the analysis of freely grazing dairy cows. It has been suggested that animals housed in respiration calorimeters may behave differently with consequences to methane emissions to those housed in polytunnels or other less restricted environments. Therefore, in any comparative analysis of mitigation strategies, potential differences due to measurement technique should be accounted for where possible.

### 3.3.1.3 Mitigation strategies

According to Hegarty (2001), there are two distinct strategies that can be expected to deliver significant abatement and there are multiple mechanisms within each strategy by which abatement can be achieved. These strategies are (1) changing attributes of the animal population and (2) the development of self-sustaining changes in the mixed microbial population of the rumen. Mechanisms for changing the attributes of the animal population to generate low methane emitting animals include selection for smaller mature size, selection for higher net-feed efficiency and selection for faster digesta kinetics. Mitigation strategies at the level of the individual animal can be broadly characterized between two types, namely pharmacological intervention and nutritional intervention.

#### 3.3.1.3.1 Nutrition

Apart from the undesirable effects that methane emissions have on the environment, they also represent a loss of productive energy for the animal. Consequently, research has tended to focus on dietary manipulation as a means of altering rumen fermentation, not only to mitigate the effect of methane emissions on the environment but also to improve productivity. The diet composition effects discussed here are those that are linked to changes in nutrient composition only. Dietary additives (e.g. ionophores), are considered in isolation later in the review.

Increasing dry matter intake is usually associated with higher methane emissions per animal, but reduced emissions per unit feed consumed. Due to the predominance of this effect above all other factors known to influence emissions, it is examined in more detail during the discussion on modelling. However, it is worth noting that this effect is due to an increasing proportion of feed energy being used for productive purposes as intakes increase, thereby spreading the emissions associated with maintenance over a higher level of production. This relationship tends to favour intensive production systems aimed at maximizing production per cow, at least as far as direct emissions per unit of product are concerned.

It has been estimated that 5 % of the variation in the proportion of gross energy lost as methane (2-12%) can be explained by the digestibility of dietary energy (Johnson and Johnson, 1995). As the organic matter digestibility of the forage decreases there is a trend for total methane loss to increase when animals were fed ad-libitum, although no difference was observed by Boadi and Wittenberg, (2002a) when intake was restricted. If the residence time of feed in the rumen is reduced, then generally there is less methane production because microbial fermentation is reduced, and it has been suggested that residence time in the rumen may account for as much as 28% of the variation in enteric methane emissions (Okine et al., 1989). Feeding regimes to encourage post-ruminal digestion at the expense of rumen fermentation will in theory lead to improved recovery of nutrients in animal product and a reduced methane loss. Such regimes have been implemented with regard to improved nitrogen retention (rumen bypass protein sources) for many years, and the increase in the proportion of maize silage in UK diets has undoubtedly led to a similar effect for carbohydrates. Both the type of dietary carbohydrates and the rate of fermentation influence the relative proportions of, and total volatile fatty acids produced during feed fermentation. For example, diets that are rich in starch (e.g.

maize silage, cereals) tend to favour propionate production leading to lower amounts of methane compared to diets comprising mainly grass silage or concentrates rich in digestible fibre (e.g. sugar beet pulp, citrus pulp). Indeed, the overall balance between starch and acid detergent fibre levels in the diet has been exploited as an indicator of methane output in recent statistical models (see later).

The ameliorating effect of dietary fat on methane emissions from ruminants has been known for many years (Blaxter and Czerkawski, 1966). However, the global concern to reduce anthropogenic sources of greenhouse gas emissions to the atmosphere has recently led many nutritionists to investigate this issue further. Several studies, both in vitro and in vivo have demonstrated marked reductions in methanogenesis as levels of supplemental fat are increased. The effects vary according to basal diet, species, type of fat and level of inclusion in the diet.

For a given level of fat inclusion in the diet, the form of this fat will determine the nature and degree of influence on rumen fermentation. The effects of fat source can be broadly categorised as either effects due to the degree of rumen protection, or those effects due to fatty acid composition (i.e. fatty acid chain length and saturation). Protection of dietary fat from rumen metabolism can be achieved by combination with calcium salts to form insoluble soaps (Jenkins and Palmquist, 1982; Palmquist, 1984). The greater the degree of protection, the less the effect on methanogenesis. However, such protection is unlikely to be wholly efficient in most circumstances and as Sutton et al. (1983) demonstrated through the feeding of protected and native coconut or linseed oils to sheep, a proportion of the fatty acids will still be subjected to hydrogenation within the rumen. Sutton et al. (1983) summarised the effects of protection within their experiment as '*intermediate between those of the basal diet and the two free lipids*'. However, Sutton et al. (1983) also highlight the variable degree of protection achieved depending on the source of the oil with coconut oil being less effectively protected from the formaldehyde treatment when compared with linseed oil.

Polyunsaturated long chain fatty acids (PUFA) have been shown to have a potent influence on rumen fermentation with an associated reduction in methane emissions (Blaxter and Czerkawski, 1966). The mode of action appears to be twofold. Firstly, a direct reduction in cellulolytic activity reduces the level of fibre degradation in the rumen and the associated shift in VFA stoichiometry towards increased propionate and reduced acetate reduces the quantity of free hydrogen that would otherwise end up reducing carbon dioxide to form methane. Secondly, the methanogens and the protozoa with which they associate are inhibited directly by the PUFA (Nagaraja et al., 1997; Doreau and Ferlay, 1995; Maia et al., 2006; Prins et al., 1972). Unfortunately, the application of PUFA in ruminant diets has been shown to be limited by their effects on diet digestibility and feed intake (Broudiscou et al., 1990).

However, certain saturated medium chain fatty acids (MCFA) and oils rich in combinations of such free fatty acids have been shown to achieve suppression in methanogenesis whilst exerting lesser influences on dry matter intake and animal performance. Machmuller et al. (1998) tested the ability of oils from coconut, rapeseed, sunflower and linseed on methane emission from the Rusitec system. This *in vitro* study confirmed that coconut, sunflower seed and linseed oils were effective at limiting methane production. Of the oils tested, coconut oil proved most effective at suppressing methane production with a 57% reduction at 6% inclusion. This compared with a 40% reduction for the sunflower and linseed oils. With 3% coconut oil added to diet on a dry matter basis, Machmuller et al. (2000) were able to demonstrate a consistent suppression of methane production in lambs fed over a seven week period. In a dose response trial with wethers, Machmuller et al. (1999)

were able to confirm earlier *in vitro* observations suggesting a near elimination of protozoa (97% reduction) with coconut oil fed at 7% of diet DM and an 88% reduction when fed at 3.5% of diet DM. However, the incremental effect on methane suppression as dosage increased was more marked with 28% and 73% reductions in emission at the 3.5% and 7% inclusion levels respectively. In contrast to the known effects of feeding fat at high levels, total tract nutrient digestibility was not affected significantly through addition of coconut oil at 7% inclusion. However, as noted by Machmuller et al. (1999), this should not be interpreted as a lack of effect on fibre degradation in the rumen given the ability of the hind gut to compensate in such a situation (Sutton et al., 1983). Dohme et al. (2000) compared the effects on methanogenesis of seven different fat sources using the Rusitec system. Whilst each oil had been selected based on a general requirement for high levels of MCFA, only three of the oils (palm kernel, coconut and high lauric acid canola) reduced methane release significantly. Solvia et al. (2003) examined closely the effects of individual fatty acids and mixtures of fatty acids on methane release in vitro. As expected, C12 was highly effective at reducing methanogenesis with a curvilinear decline in methane output as the concentration of C12 increased. However, there was a notable interaction with mixtures of C12 and C14. Whilst C14 alone showed no suppression of methane production, a 2:1 mix of C12 and C14 delivered the same reduction observed for the 100% C12 treatment.

Similar declines in methane emission to those observed in the in vitro studies have also been obtained in vivo in sheep fed at maintenance (Czerkawski et al., 1966) and as already noted, in growing lambs (Machmüller et al., 2000). However, many in vivo studies also describe a concurrent decline in feed intake and animal performance. The tendency for other added fat sources to have a negative influence on rumen fermentation can be seen elsewhere in the literature. For example, linseed is rarely used in ruminant feeds because diets containing in excess of 5% linseed have demonstrated a significant negative effect on ruminal digestion in sheep (Ikwuegbu and Sutton, 1982), but recent studies suggest that this may not be the case for dairy cattle (Ueda et al., 2003; Martin et al., 2006, 2008). Martin et al. (2008) showed that the physical form of the linseed based feedstuff impacts on the degree of suppression in methanogenesis in dairy cows. Whilst pure linseed oil produced the most significant depression in methane emission (-64%), large reductions in dry matter intake and milk production were observed. In this study crude linseed appeared to offer the most promising compromise between reduced methane emissions (-12%) and production. The negative effects of linseed on milk yield may be due to the way that linseed is presented as others have reported an increase in milk yield when linseed is offered in its purest form (Bu et al., 2007; Loor et al., 2005), whereas a decrease in milk yield has been observed with extruded linseed (Gonthier et al., 2004; Akraim et al., 2007). However, it should be noted that under practical feeding conditions, it is likely that crude or extruded linseed would be used as they are more readily available, easy to use and less costly. Confirming the potent effect of dietary oil on rumen fermentation, McGinn et al. (2004) showed that there was a 20% reduction in NDF digestibility for steers fed sunflower oil at 5% of dry matter intake. This reduction led to decreased methane emissions compared to control animals when intakes were adjusted for energy intake.

In summary it seems that raising dietary fat level is a promising strategy for reducing emissions. However, high oil content feedstuffs can be costly and the negative effect on intake and milk fat concentration (Zheng et al., 2005) constrain this strategy to situations where these negative effects can be minimised.

It is also worth noting at this stage that little research has been conducted on whether or not methane mitigation strategies, which are effective in the enteric fermentation of the animal, are prone to a compensatory higher methane release from the manure during storage, as the amount of residual fermentable organic matter available may be higher.

#### 3.3.1.3.2 Genetic selection

It is well established that any differences in the digestive anatomy or physiology of either individual animals or between breeds can result in differences in methane production. For example, Robertson and Waghorn (2002) compared two dairy cow genotypes: Friesians selected for high productivity on pasture and Holsteins derived from selection programmes based on high concentrate diets in the Netherlands and North America. Holsteins produced 8 to 11 % less methane as a percentage of GE compared to the New Zealand Friesian. Moreover, differences were exacerbated during periods of high productivity. Others found no differences in enteric fermentation emissions from Holstein (dairy; 238.0 ± 6.9 litres/d) and Charolais x Simmental (beef; 228.6 ± 7.8 litres/d) heifers of similar body weight and age (Boadi and Wittenberg, 2002b).

Genetic selection for production traits is often conducted using high concentrate diets as literature suggests that genetic selection using high concentrate diets is appropriate for mitigation of enteric methane emissions in growing or lactating animals. As much as 27 % of the variation in methane emission for cattle consuming forage diets is related to animal-to-animal variation (Boadi and Wittenberg 2002a). Further work is required to determine whether these differences are related to intake behaviour, or to potential anatomical and physiological differences in the gastrointestinal tract of cattle or the heritability of this trait. However, the degree of variability suggests that there is potential to select for low methane emitting animals.
Genetic variation in dry matter intake exists, independent of live weight and production level and this variation provides a basis for genetic selection for feed efficiency of cattle (Arthur et al., 2001). Cattle that eat less than their peers of equivalent live weight and performance have a low residual feed intake (RFI) and are more feed efficient, as shown by lines of cattle selected for RFI (Arthur et al., 1996). So selection for reduced RFI should lead to substantial and lasting methane abatement. In trials with Friesian Jersey crossbred herds, variation was found between cows and Goopy and Hegarty (2004) identified some steers as ‘high’ and ‘low’ emitters on identical diets and dry matter intakes. Therefore, work is required to determine whether these differences are related to intake behaviour, or to potential anatomical and physiological differences in the gastrointestinal tract of cattle or the heritability of this trait.

#### 3.3.1.3.3 Forage selection and management

Forage selection and management are important to any greenhouse gas mitigation strategy. Pasture management includes factors such as forage species selection, stocking rate and continuous vs. rotational grazing strategies. These have all been demonstrated to affect enteric methane emissions.

It has been suggested that one of the most promising pasture management strategies for mitigation of enteric emissions is the inclusion of legumes in the forage species mix. The enteric emissions of cow-calves offered either alfalfa-grass or grass-only pastures over the course of a grazing season showed that dry matter intake was greater in the alfalfa-grass pastures than for grass-only pastures (11.4 vs. 9.7 kg/d) (McCaughey et al., 1999). In contrast, methane production, adjusted for differences in body weight, was the reverse (0.53 vs 0.58 g/kg BW/d, respectively).

Energy lost as enteric methane emissions as a % of gross energy intake (GEI) were approximately 2% lower for the alfalfa-grass group (i.e. 7.1 vs. 9.5%, respectively) of GEI for grass-only pastures. An additional benefit of the alfalfa-grass mixture was an 11 % increase in calf growth rates; this would serve as further incentive to consider legume incorporation as a mitigation strategy. It has been proposed that the lowered methane loss observed with legumes is as a result of the lower proportion of structural carbohydrates and faster rate of passage of legumes, which will shift the fermentation pathway towards higher propionate production.

Apart from the observed effects in cattle, Waghorn et al. (2002) fed sheep a wide range of fresh cut, good quality forages and observed a two-fold range in methane emissions, from 11.5 g methane/kg DMI with birdsfoot trefoil to 25.7 g methane/kg DMI with a ryegrass, white clover pasture. It should, however, be noted that animals grazing on pasture have the ability to be more selective than animals in any feeding study and as such the possibility exists that differences between forage species is even greater for grazing animals.

Condensed tannins, which are a constituent of some legumes, have been linked to decreased enteric methane emissions (Waghorn et al., 2002). Apart from the beneficial impacts on methanogenisis, condensed tannins can limit the incidence of bloat and lower worm burdens (Niezen et. al. 1998). They also bind to plant protein complexes in the rumen, which subsequently act to reduce microbial degradation of soluble protein to ammonia, although the underlying mechanism by which this occurs remains to be established. However, Jones et al. (1994) demonstrated that tannins reduced the ability of some bacterial species to colonize on plant particles.

Plant breeders have manipulated the carbohydrate composition of grasses and this has led in recent years to the commercial application of high sugar grasses for cutting and grazing. These cultivars tend to exhibit increased levels of water soluble carbohydrate leading to improvements in digestibility and animal performance. Associated with this change in composition has been a tendency for reduced methane emissions (Lovett et al., 2006), presumably as a consequence of the change in fermentation stoichiometry.

Taken together, it is evident that forage species selection and pasture forage quality are vital factors in any mitigation strategy for grazing ruminants. The inclusion of legume-species in the diet of ruminants will increase forage DM digestibility, improve animal performance and reduce methane production per unit of product produced.

#### 3.3.1.3.4 Feed additives

One approach is to include feed additives that manipulate the rumen microfloral populations by inducing a stable, low emission, modified rumen fermentation; this can be achieved by biological control approaches aimed at methanogens and associated organisms, including vaccination, for the establishment of effective acetogenic and bacteriocin producing populations (Hegarty, 2001). However, such methods may not be ideal as they often require a withdrawal period and so can not be used in dairy cattle. Alternative strategies are adaptations of animal diets to provide alternate hydrogen acceptors or that shift the fermentation pathway, management for improved productivity (i.e. growth, milk yield, reproductive efficiency), animal reduction and environmental conditions.

Ionophores are polyether antibiotics produced by soil microorganisms that modulate the movement of cations such as sodium, potassium and calcium across cell membranes. It has been shown that ionophores cause a shift in the rumen bacterial population from gram positive to gram negative organisms, with a concurrent shift in fermentation from acetate to propionate, which is associated with a reduction in rumen methanogenisis. Van Nevel and Demeyer, (1995) showed that the use of monensin can depress methane emission by 25 % in the short-term but longer-term the data is much more variable suggesting that an adaptive response may occur following prolonged supplementation. Similarly, adult sheep exhibited significant reductions in methane emitted from the rumen and caecal contents (Mbanzamihigo et al., 1996). Monensin supplementation (250-270 mg/d) to grazing steers (McCaughey et al., 1997) or heifers (Johnson et al., 1997) had no impact on methane emissions but when yearling heifers were offered an alfalfa diet there was a 15% reduction in methane production. When monensin was given to a lactating dairy herd whole barn emissions declined in the initial month after inclusion of monensin in the lactation ration, but emissions returned to previous levels thereafter (Kinsman et al., 1995). Further work is required to determine the mitigation potential of ionophores under different management systems as well as the mitigation potential and adaptation to ionophores other than monensin.

Other mechanisms for methane inhibition include feed additives causing a direct inhibition of methanogenesis (Van Nevel and Demeyer 1995), feeding of dicarboxylic organic acids to enhance propionate production (Lopez et al., 1999b), use of acetogens as a daily feed additive to provide hydrogen (Lopez et al., 1999a), enhanced methane oxidizing bacterial populations in the rumen (Valdes et al., 1996), rumen defaunation (Hegarty, 1999), and immunization of ruminants against their own rumen methanogens (Baker, 1995).

The potential of dicarboxylic organic acids (OA) such as fumaric acid (FA) and malic acid (MA) as inhibitors of methanogenesis has been shown in vitro (Carro and Ranilla, 2003; Newbold et al., 2005) as well as in vivo (Lila et al., 2004; Wallace et al., 2006). These OA act as alternative H2 sinks in the rumen thereby decreasing ruminal methanogenesis (Newbold et al., 2005). Unfortunately, in vivo responses in methane production after OA supplementation have been inconsistent. For example, Wallace et al. (2006) reported C 61 H4 reductions of up to 75% in lambs offered FA and similarly Lila et al. (2004) reported 18% lower $CH_4$ emissions from steers offered β -cyclodextrin diallyl maleate. Other *in vivo* studies have failed to establish any effect of OA supplementation in beef heifers (Beauchemin and McGinn, 2006) and dairy cows (McCourt et al., 2008). Foley et al. (2008) demonstrated that steers fed malic acid would indeed reduce both dry matter intake and methane emission. Overall they observed a 9% reduction in methane emission per unit of dry matter intake. However, they note that the reduction in intake may have serious consequences for animal performance. Thus, in contrast to the well documented methane production response to OA in vitro, responses to dietary supplementation in vivo remain inconclusive. In particular it has been suggested that their efficiency is reduced when concentrates are fed, as evident from an in vitro trial in which the efficiency was only 4.8% (Carro and Ranilla 2003). They are also expensive to purchase and they are not suitable for grazing animals as they have to be fed daily.

Yeast based feed additives have been shown to have a methane suppressing effect in some situations with Possenti et al. (2008) observing a 12% reduction in emissions for male cattle fed *Saccharomyces cerevisiae* in comparison with a control diet. However, the effect is not universal across different diet types and McGinn et al. (2004) did not observe any significant reduction for two different yeast based feed additives for steers fed a concentrate diet. It is assumed yeast has the potential to reduce emissions in three direct ways: (1) by increasing butyrate or propionate production (Lila et al., 2004); (2) by reducing protozoan numbers (Newbold et al., 1998); and (3) by promoting acetogenesis (Chaucheyras et al., 1995).

Another option to reduce methane output is to immunise animals against their own methanogens and protozoa and an immune response to rumen protozoa has been demonstrated by administering an immunogenic preparation (Baker et al., 1997). This will indirectly affect the activity of those rumen methanogens that have a commensal relationship with rumen protozoa. However, given the variable results obtained through defaunation, it is to be expected that a direct anti-methanogenic vaccine would deliver a much greater efficacy. However, Wright et al. (2004) found a mere 7.7% reduction in methane production per kg dry matter intake after vaccination in sheep. Much more work is therefore needed to make this technique effective. Ultimately, vaccination would be a valuable tool for methane reduction as it could be applied to grazing dairy cows unlike many other mitigation options.

Recently various plant extracts have received considerable interest with regard to their potential to limit methane production from ruminants. These extracts include essential oils, saponins and related compounds. Their main effects in the rumen involve reduction of protein and starch degradation and an inhibition of amino acid degradation, due to direct influence on certain rumen microorganisms. One mode of action suggested for essential oils is an effect on the pattern of bacterial colonisation of, in particular starch rich, substrates as they enter the rumen. A second possible mode of action is their inhibition of hyper-ammonia producing bacteria involved in amino acid deamination. However, the effect of essential oils depends on the chemical composition of the essential oils used, which is not always sufficiently described in the literature. Saponins are secondary compounds produced mainly by plants. A wide range of biological effects of saponins have been described, although the majority may be ascribed to their action on membranes. There seems to be inconsistency in the effects of saponins in the rumen environment. It has been suggested that this is due in part to the development of other microbial populations capable of degrading saponins in the rumen (Hart et al., 2008), and therefore this might limit their use in practical conditions. Organosulphurous supplements prepared from garlic have experienced increasing popularity in the last decade. However, the final concentration of the active compound, allicin, in different garlic preparations varies significantly (Hart et al., 2008). Effects have been reported on volatile fatty acid production and decreased methane production, with a subsequent decrease in rumen methanogens. Hart et al. (2006) observed a decrease (94% at 20 mg/l allicin addition) in methane production in vitro and results obtained using real time PCR suggested that allicin had a direct effect on reducing the numbers of methanogens in the fermentor with no effect on the total bacterial population.

#### 3.3.1.3.5 Manure Management

Values on methane emission from differently stored manure are still scarce. Külling et al. (2001) showed that the levels and the differences between manure storage systems in methane emissions were distinctly determined by the duration of storage. Initially farmyard manure was a more important source of methane than the two types of slurry, with higher values in the urine-rich slurry/farmyard manure system than with complete slurry but after around 7 weeks this situation had reversed, when methane emission had almost ceased in all manures, except in complete slurry. Over a three month period the methane emitted from urine-rich slurry/farmyard manure storage was lower than for the complete slurry system, although two thirds of the methane emissions were within the first three months (Amon et al., 1998, 2001; Külling et al., 2001; 2002). It is likely that the formation of undisturbed crust and surface covers substantially reduced methane release from slurry and retarded the onset of major methane release (Sommer et al., 2000).

One method of potentially reducing the amount of methane emitted to the environment is to store manure in anaerobic slurry tanks. Methane losses from anaerobic digesters and methane emissions from digested and undigested slurries are currently under investigation. Berg and Pazsiczki (2006) investigated different materials for covering liquid manure storage facilities to reduce gaseous emissions and concluded that covering the slurry and lowering the pH of the slurry to below 6 can significantly reduce methane production.

Anaerobic digestion (AD) is a treatment that composts waste in the absence of oxygen, producing a biogas that can be used to generate electricity and heat. As well as biogas, AD produces a solid and liquid residue called digestate which can be used as a soil conditioner to fertilise land. The amount of biogas and the quality of digestates obtained will vary according to the feedstock used. More gas will be produced if the feedstock is putrescible, which means it is more liable to decompose. Sewage and manure yield less biogas as the animal which produced it has already taken out some of the energy content. With respect to livestock production AD systems are an attractive method of renewable energy production, reducing odours, efficient organic waste recycling, improved utilization of the manure as fertiliser and reduced GHG emissions, and many modern, large-scale, manure-based biogas plants have been constructed recently (Umetsu and Kimura 1999).

Manure must be stored until it can be applied to the land; during storage it can be covered to prevent emission of methane, carbon dioxide, ammonia and other hazardous gases (Hornig et al., 1999). The effect of the duration of storage on methane emissions from dairy cow and swine manure slurries stored at 10$^{o}$C and 15$^{o}$C in closed tanks has been reported by Masse et al. (2002). Umetsu et al., (2005) examined methane emissions from tank storage of raw dairy manure slurry (RS) and after it had been digested in a methane digester (DS). They showed that the amount of methane emitted per unit of volatile solids of the RS and DS was 0.19 litres/g and 0.10 litres/g, respectively, and at temperatures greater than 15$^{o}$C, methane concentration in the emitted gas remained more than 40% of the total gas. Moreover, methane emission from manure storage tanks at temperatures substantially less than 10$^{o}$C are negligible but at higher temperatures it is recommended that underground storage is used in order to maintain lower manure temperatures and thus reduce methane emissions.

Whilst the technical and environmental benefits of using AD systems are well established, there are several reasons why AD uptake has historically been relatively low in most countries. Barriers to uptake include financial, policy, environmental, technical, awareness and knowledge and infrastructure factors.

#### 3.3.1.3.6 Herd management

In addition to those already mentioned there are other factors that can contribute significantly to methane output from a dairy herd. As a direct consequence of the diminishing returns relationship between dry matter intake and methane emission (see modelling section), it follows that management systems geared to deliver high milk production levels per cow will tend to result in lower emissions of methane per unit of milk produced. In addition, more extensive systems that rely on lower input strategies (e.g. more milk from forage, extended grazing or organic systems), will require more animals to produce a given quantity of milk. As each animal has a basic nutritional maintenance requirement (without producing any milk), there will be a significant amount of rumen fermentation (and methane output) without any corresponding milk production. Therefore, extensive, low yield per cow systems tend to produce more methane for a given level of herd milk production.

In a similar manner, the turnover of animals within a dairy herd as defined by herd fertility, herd health and culling policy will have an impact on emissions via the overall replacement rate. Herds with high replacement rates (>25%) will require more youngstock to sustain total cow numbers, thereby increasing the requirement for unproductive animals and hence the methane emission per unit of herd production. This will be discussed in more detail during the discussion of herd level models.

#### 3.3.1.4 Mathematical modelling

Mathematical models offer the potential to evaluate intervention strategies for any given situation, thereby providing a low cost and quick estimate of best practice. Models of methanogenesis can be classified within three main groups. Firstly there are emission factors that relate specific classes of animals (e.g. youngstock, dry cows, milking cows etc.) to a yield of methane per unit of time. Secondly, there are the statistical models that relate directly the nutrient intake and other factors with methane output. Finally, there are the dynamic mechanistic models that attempt to simulate methane emissions based on a mathematical description of ruminal fermentation biochemistry.

##### 3.3.1.4.1 Predicting emissions from an individual animal

###### *3.3.1.4.1.1 Emission factors*

The construction of inventories requires a mechanism to relate the modelling of emissions at the animal level to higher levels of organisation such as the farm or region. For instance, environmental policy is generally founded on the basis of regions, be they political or geographical, which span areas containing many different farms and farming systems. Therefore, inventories are required to demonstrate estimates of current emissions and to highlight historical trends in the light of existing national and international targets. To achieve this aim the IPCC guidelines suggest the application of emission factors that describe a yield of methane from a given class of livestock per unit of time. At the most basic level the IPCC Tier 1 recommendations apply a static emission factor (e.g. 109 kg methane / head / year for western European dairy cattle). However, the Tier 2 approach is more advanced and this involves a more comprehensive analysis of the class of livestock under review with an estimate of methane emission based on the likely inefficiencies in the conversion of dietary energy to animal product. The default Tier 2 recommendation is that 6.5% of dietary gross energy is converted to methane. However, the IPCC documentation acknowledges that uncertainty in the estimation of energy intake will have a major impact on the model predictions as well as the additional uncertainty associated with the methane conversion factor. Separate emission factors are derived for enteric fermentation and manure storage. It should be noted that whilst the official IPCC guidelines involve utilisation of their recommended methodology to produce the individual emission factors using either a simplified (Tier 1) or more detailed (Tier 2) approach, it is possible to substitute other models such as some of those described below in order to arrive at the required factors, thereby giving rise to a Tier 3 approach.

*3.3.1.4.1.2 Statistical models*

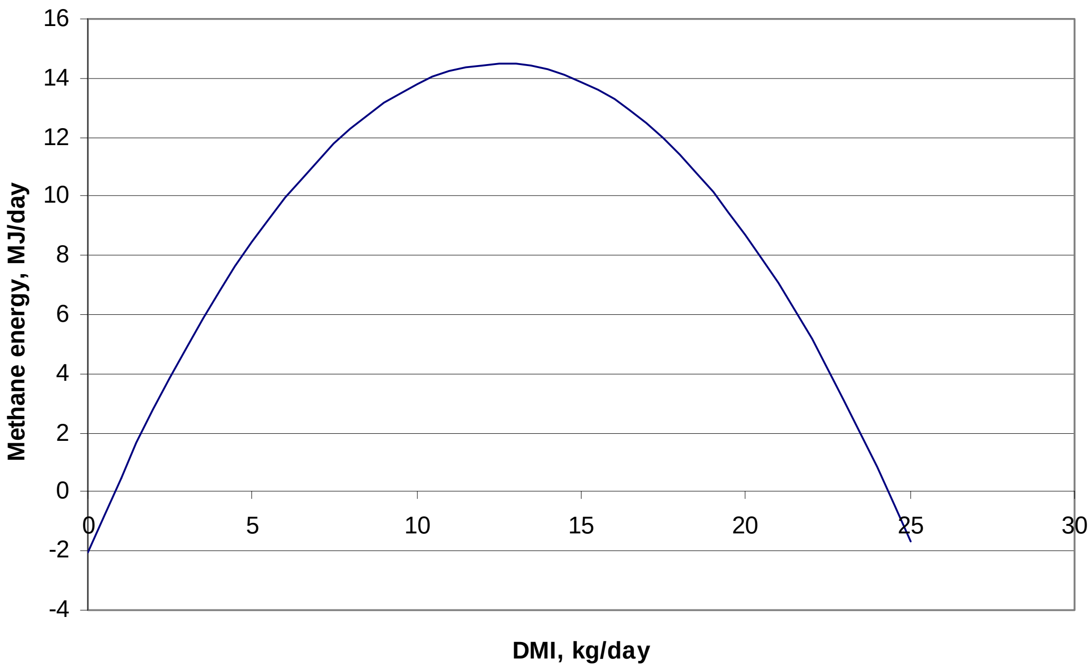
Emission factors such as those discussed above are in fact the most basic category of statistical models. More generally, statistical modelling has been used as a tool to describe empirical relationships between the animal and enteric methane production over many years (Kriss 1930; Bratzler and Forbes, 1940; Blaxter and Clapperton, 1965). The statistical models tend to be well suited to practical application for rapid diet evaluation or larger scale inventory purposes. Several statistical models constructed to predict methane emissions from cattle were summarised by Wilkerson et al. (1995). Based on a comparative evaluation against independent data, Wilkerson et al. (1995) recommended adoption of the Moe and Tyrrell (1979) linear equation for predicting emissions from dairy cows (equation 1.1). Unfortunately for those seeking to assess global emissions from dairy cows, this model, along with many others, was developed solely on the basis of North American data. Of practical importance is that it requires cellulose and hemicellulose to be known and such detailed data are unavailable on many farms. The relationship is summarised as:

$$CH_4\,(\mathrm{MJ/d}) = 3.38 + 0.51\mathrm{NFC} + 2.14\mathrm{HC} + 2.65\mathrm{C}$$

Where NFC is non-fibre carbohydrate, HC is hemicellulose and C is cellulose (all in kg/d). The limitation of these statistical models lies with their tendency to be unreliable predictors of emissions when applied outside of the production systems upon which they were developed. Factors including, species, physiological age or condition, nutrition and management all contribute to variable emissions, thereby tying statistical models to the factors and range of data that were used in their construction. An example of this situation is shown in Figure X where the relationship between dry matter intake (DMI) and methane output described by Axelsson (1949) is plotted. Given the prevalence of more recent models to rely on a linear form to describe what is clearly a curvilinear relationship, Axelsson's model showed considerable logic and foresight. However, Figure 3 shows clearly that a simple quadratic relationship cannot be used to predict emissions beyond the range of intakes upon which the model was developed (3 to 12 kg DM/day), with negative methane emissions predicted above 24kg DMI. It is worth noting that even the IPCC tier 2 based emission factors are linear models relating energy intake to methane output.

It is the restrictions of the statistical modelling approach that give rise to their ever increasing number, with each practical situation demanding a specially derived relationship. As we have seen from Figure 3, there is a problem where published models are used outside of their intended data range and, therefore, model comparisons are in danger of portraying a contradictory picture of model performance depending on the data set used for their evaluation. A better approach may be to constrain evaluations of such models to the confines of the system for which they were developed and as far as application is concerned, to think instead of selecting appropriate statistical models on a case by case basis.

Figure 3. Methane emissions as predicted from dry matter intake by Axelsson (1949).




Another danger of the empirical approach seen with many statistical models is that they imply cause and effect where none actually exists, especially when the aim of a study is to produce the strongest possible correlation for a given set of data. For example the following model was proposed as one of five by Holter and Young (1992) with the objective to predict methane production from dairy cows.

$$\begin{aligned} CH_4\,(\%GE) = {} & 2.898 - 0.0631M + 0.297MF - 1.587MP + 0.0891CP + 0.101FADF \\ & + 0.102DMI - 0.131EE + 0.116DMD - 0.0737CPD \end{aligned}$$

Where M is milk yield (kg/d), MF is milk fat (%), MP is milk protein (%), CP is crude protein, FADF is forage ADF (% DMI), EE is dietary fat (%), DMD is dry matter digestibility (%) and CPD is crude protein digestibility. This model implies a significant effect of milk yield and constituents on methane output. Milk yield and constituent concentrations are themselves the function of the animal's nutrition and other factors as pointed out by the authors. However, this relationship already accounts for the effect of major nutrients and dry matter intake and for those applying the model as a predictive tool the implications could be misleading.

In an attempt to improve on existing statistical models for UK based dairy cow diets, Mills et al. (2003) tried to harness the advantages of speed and simplicity associated with the statistical approach, whilst introducing a degree of mechanism to include known nutritional effects on methane output as follows:

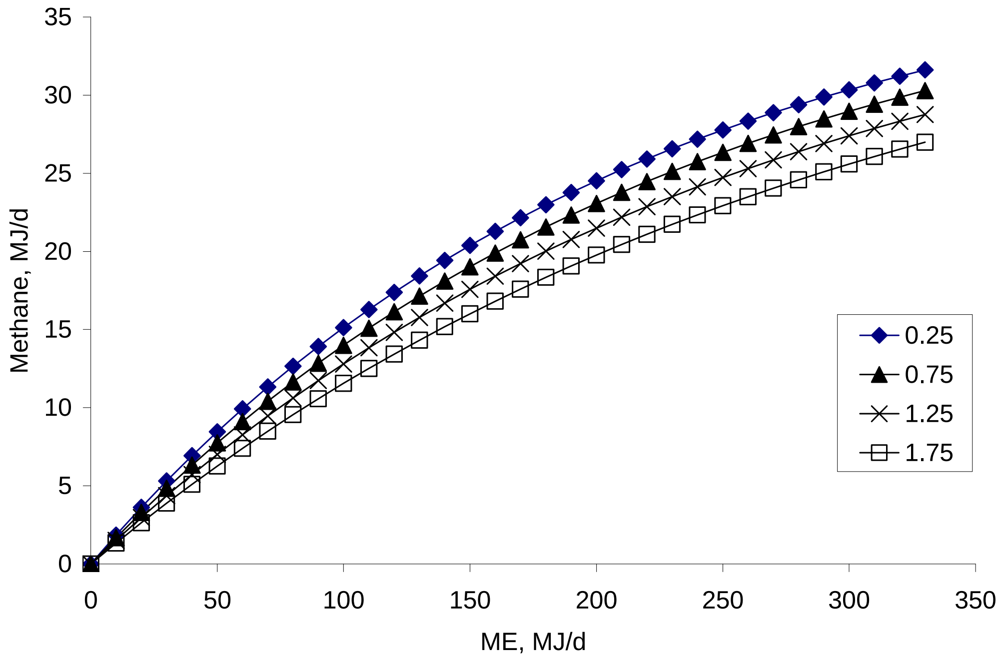
$$CH_4\,(MJ/d) = a - (a + b)e^{-cx}$$

Where $a$ and $b$ are the upper and lower bounds of methane production respectively and $c$ is a shape parameter determining the rate of change of methane production with increasing metabolisable energy (ME) intake as defined by $x$. $c$ is calculated as follows:

$$c = -0.0011\left[\frac{\text{St}}{\text{ADF}}\right] + 0.0045$$

Where St is the starch concentration in the diet (g/kg DM) and ADF is the Acid Detergent Fibre concentration (g/kg DM). Through application of a non-linear approach, this model displays the typical diminishing returns response observed with increasing intake as described by Axelsson (1949), but absent from many later models. However, as shown in Figure 4, the relationship based on the Mitscherlich equation form, is refined by altering the slope of the curve according to the form of dietary carbohydrate. This model tries to strike a balance between additional complexity and therefore, wider application and ease of use. However, it is known that many other nutritional factors absent from this model can impact significantly on methane output and as such this model is still restricted in its application. When used to estimate the likely influence of the trend towards increased levels of starch in the diet at the expense of fibrous carbohydrate, as seen in many European farming systems, this model provides a quick solution.

Figure 4. A statistical model of methane emissions from dairy cows defining the response to energy intake as affected by starch to ADF ratio (g/kg DM) (Mills et al., 2003).




The statistical modelling of Mills et al. (2003) was progressed as part of a DairyCo funded study examining carbon and nitrogen retention in dairy cows. Measurements of energy and/or nitrogen balance obtained using respiration calorimetry and digestion trials were accumulated into a database for meta-analysis of effects of key parameters on both methane and nitrogen excretion in growing and lactating beef cattle and lactating and non-lactating dairy cows. An existing database of individual measurements of energy and nitrogen balance from the University of Reading, which included measurements of methane and nitrogen excretion, was updated and expanded using more recent data from Reading and existing data from other laboratories as appropriate. Additional data were obtained from the USA, Wales, and the Netherlands, giving a total of 1819 individual measurements (1335 records of methane excretion). A multivariate analysis was conducted, with appropriate adjustments for variance associated with location and trial effects, to determine the most important dietary factors that influence methane and nitrogen excretion, based on both linear and nonlinear models.

A summary of the most effective models is shown in Table 3 and Table 4 below. As with the earlier modelling studies, the dominant effect of intake (dry matter or energy) was clear. In an extension to previous analyses, a model was constructed to describe the relationship between DMI and methane emission as a percentage of GE intake. Whilst this model did not explain as much variation as other models that were concerned only with predicting total daily methane output, it did highlight the large variability in the proportion of feed energy lost as methane. This serves to emphasise the limitations of the linear Tier 2 approach recommended by the IPCC whereby a constant parameter of 6.5% of GE is applied. A model relating digestible energy intake to methane production was constructed and this demonstrated particularly correlation during the evaluation. Where information on digestible energy intake is available, this relationship could be used effectively to predict emissions. However, estimates of dry matter intake alone are more commonly available and therefore it is likely that the non linear relationships between DMI and methane output will be better suited to application within the industry. Where information on diet composition is available, the model including the ratio of dietary starch to acid detergent fibre will account for much of the variation in emissions due to the balance between structural and non structural carbohydrate. One of the principal advantages to adopting a non linear model for relating feed intake to methane output is that changes to management systems (e.g. extensive vs. intensive), genetics or diets that result in a shift in the level of intake over time are accounted for within the model to a greater degree than with their linear alternatives.

Table 3. DairyCo meta-analysis. Model Description.

| Model | Description | Parameters |
|---|---|---|
| DMI vs $CH_4$ | $CH_4$(MJ/d) =a-(a+b)$e^{-cx}$ | a = 74.43, b = 0, c= 0.0163, x = DMI |
| $DMI_{Starch:ADF}$ vs $CH_4$ | $CH_4$(MJ/d) =a-(a+b)$e^{-cx}$ | a = 74.43, b = 0, x = DMI<br>c = 0.0187 + 0.0059 / (1 + exp(Starch:ADF - 3.1003) / 0.6127 |
| DMI vs $CH_4$ % GEI | $CH_4$(%GEI) = m DMI + c | m = -0.101, c = 7.16 |
| DEI vs $CH_4$ | $CH_4$(MJ/d) = m DEI + c | m = 0.0779, c = 2.6861 |

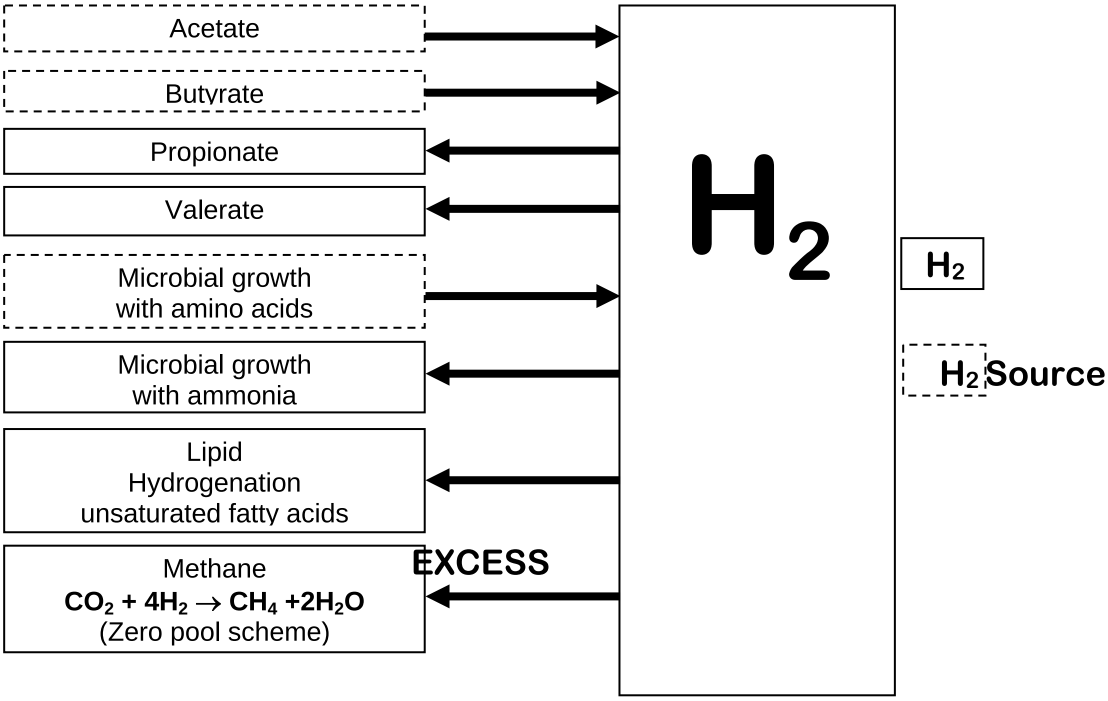
Table 4. DairyCo meta-analysis. Model Evaluation.

| Model | Observed mean (MJ/d) | Predicted mean (MJ/d) | $r^2$ | Root MSPE (% of observed mean) |
|---|---|---|---|---|
| DMI vs $CH_4$ | 15.65 | 15.60 | 0.91 | 11.02% |
| $DMI_{Starch:ADF}$ vs $CH_4$ | 16.73 | 16.83 | 0.94 | 12.48% |
| DMI vs $CH_4$ % GEI | 5.70% | 6.55% | 0.56 | 23.5% |
| DEI vs $CH_4$ | 17.22 | 17.28 | 0.96 | 9.65% |

*3.3.1.4.1.3 Dynamic mechanistic models*

Unlike their statistical counterparts, dynamic models include time as a variable and they tend to be more mechanistic in their construction. This type of model has been applied successfully on several occasions to predict methane emissions from ruminants (Mills et al., 2001). However, they too are not without their limitations and they may not deliver quick solutions based on very limited dietary information. By definition mechanistic models describe in more detail the fermentation processes occurring in the gut that result ultimately in the formation of methane as a sink for excess hydrogen. Within a model of rumen digestion, Baldwin et al. (1987) described a scheme for calculating the sources and sinks of this reducing power during fermentation. Ulyatt et al. (1991) evaluated this model using independent data for New Zealand livestock and compared predictions with those from the models of Blaxter and Clapperton (1965) and Moe and Tyrrell (1979). Ulyatt et al. (1991) highlighted the improved prediction seen when using the mechanistic model although they note that the predictions were not without bias. The same scheme was incorporated subsequently into the whole rumen model of Dijkstra et al. (1992), firstly by Benchaar et al. (1998) and then with revisions by Mills et al. (2001) and used to evaluate the potential impact of various mitigation options.

Figure 5. A mechanistic scheme for methane production in the rumen (Baldwin 1987; Mills et al., 2001).




When integrated with a larger model of rumen digestion and metabolism, schemes such as that displayed in Figure 5 have the advantage that any model input can be assessed as to its effect on methane production, assuming that the underlying biology is represented sufficiently. Indeed, this model has been applied to suggest the most effective nutritional strategies for limiting emissions whilst maintaining an adequate nutrient supply to the host animal (Mills et al., 2003) and also in the broader context of limiting emissions of both methane and nitrogen pollutants from dairy production (Kebreab et al., 2006). Whilst practical experimentation can demonstrate the effect of individual treatments for a given animal type and environment, mechanistic models are able to extend the value of these results by providing indications of response outside of the bounds of the experimental treatments. The potential to reduce environmental damage and to improve productive efficiency in dairy cows was demonstrated for increasing intake, increased supplementation and changing non-structural carbohydrate source by Mills et al. (2001). Such a tool can therefore, form part of an integrated system to help producers and policy makers manage the threat of climate change with respect to livestock production.

#### 3.3.1.4.2 Predicting emissions from the herd and farm

Attempts to relate mechanistic model predictions of emissions to their consequences on a larger scale have been very limited and it has remained an area where static emission factors or simple statistical models have predominated. A herd of cows or flock of sheep comprise animals at different physiological states and on different nutrition. These factors, that are essentially functions of management, need to be considered when collating individual estimates of methane production in order to examine the effect of mitigation strategies on a larger scale. Of particular importance with dairy cows is the need to account for the change of herd structure over time. Modern dairy farming systems in the developed world do not conform to the standard model with mature animals calving once per year. Calving intervals for many animals are extended well beyond 400 days and age at first calving varies greatly depending on the management system and effects of breed. Garnsworthy (2004) describes a dairy herd model that integrated and updated work from previous studies (Stott et al., 1999; Grossman et al., 1999; Yates et al., 2000), in an attempt to consider the effects of changes in herd management on methane and ammonia emissions. A particular focus of this work was to evaluate the effect of herd replacements on overall emissions. In principle, higher individual milk yields facilitate a smaller overall herd size and hence fewer replacements and a reduced yield of methane per litre of milk produced. However, high producing herds are also associated with declining fertility levels and associated extended calving intervals and increased culling that may negate at least part of this effect. Garnsworthy (2004) showed that the individual effects of milk yield and fertility respectively were of greater significance than changes to forage quality when it came to influencing total methane emissions. As stated previously, the effects of milk yield and fertility are certainly not independent of each other, especially when one considers the case for shorter calving intervals that increase the proportion of time an animal spends producing milk during early lactation. Garnsworthy (2004) demonstrated that improvements to herd fertility (conception rate) and the associated management (oestrus detection) produced significant benefits regardless of level of intensity or the presence of production quotas. However, the effect of improving fertility from current UK levels (50% oestrus detection and 37% conception rate) to 'ideal' levels (70% oestrus detection and 60% conception rate) depended on the presence of quota restrictions and average milk yield per cow. With a 6000 litre herd average and no quota, improving fertility by the stated amount indicated a 10% reduction in total herd emissions. For a 9000 litre herd with a milk quota of 1m litres per year, the associated reduction in methane was almost 25%. Garnsworthy (2004) thereby indicated the dangers of focusing too heavily on modelling single mitigation issues such as nutrition in an attempt to limit overall emissions.

Whilst Garnsworthy (2004) helped to broaden the scope and application of models to predict methane emissions, there remain elements of the production system that have tended to be considered in isolation. At least for the sake of completeness a farm model of emissions should include an estimate of production from manure storage and distribution. However, just as animal type, nutrition and management affect methane production from individual animals; there are a variety of factors that influence emissions during collection, storage and distribution of manure. This diversity introduces complexity to the task of simulating emissions and together with their relative insignificance in relation to enteric emissions, these factors have contrived to limit modelling efforts to relatively crude inventories, the most well known of which is that used by the IPCC.

### 3.3.1.5 Knowledge gaps

#### 3.3.1.5.1 Experimentation

Whilst the available knowledge base concerning the factors controlling methane emissions from dairy farms has been expanding greatly over recent years, there still remain some important areas where information is scarce. For example, an analysis is urgently required of the effects of nutritional mitigation strategies to reduce enteric emissions (e.g. added fat, increased starch), on the composition of manure and subsequent emissions from storage or spreading. In addition a study of the extent of emissions from unmanaged areas on dairy farms is required to help determine how significant these emissions are over a broad range of farm types. At present the absence of information has led this element to be omitted from farm level GHG inventories.

The scope for breeding animals which produce less methane per day without any associated losses in production has not been adequately assessed especially given its large potential impact. It is speculated that animals with reduced methane emissions but uncompromised performance occur within a population due to a more efficient feed conversion ratio, smaller frame size and faster digesta kinetics or altered digestive function. Work is therefore required to determine whether these differences are related to intake behaviour, or to potential anatomical and physiological differences.

Further research is also necessary on potential biological control organisms targeting rumen methanogens or the protozoa which they associate with. In addition, the adaptation of methanogens to chemical inhibitors is not understood and monitoring and assessment of adaptation is required for novel control strategies such as vaccines or biological controls. The role of natural bacteriocins in inhibiting rumen methanogenesis looks promising but needs further evaluation. There is a limited amount of information on the range, activity and hydrogen threshold of naturally occurring gut-dwelling reductive acetogens.

#### 3.3.1.5.2 Modelling

Despite the recent efforts to model the nutritional influence on ruminant methane emissions, there exist several nutritional and non-nutritional factors that have yet to be given adequate consideration within models of rumen function. The tendency for existing mechanistic schemes to underestimate higher level emissions from dairy cows (Mills et al., 2001) may demonstrate the need to expand the models to account for some of these effects. For example, the description of microbial metabolism within extant mechanistic models is restricted in part by their limited characterisation of different microbial groups, with even the more complex models relying on one (Baldwin et al., 1987) or two (Dijkstra et al., 1992) distinct subdivisions of the microbial population. Further developments in this area have been constrained by the availability of suitable quantitative data and the problems associated with modelling the interrelationships observed between multiple microbial groups competing for the same substrates (Baldwin, 1995). Recent indications of the dynamic metabolic behaviour observed with different levels of substrate availability for individual microbial species (Soto Cruz et al., 2002) suggest that additional complexity will need to be integrated within mechanistic models if we are to refine further our estimates of methane output.

As work is published regarding the quantitative effect of supplemental fat in dairy cow diets on resulting methane emissions, our biological understanding of the mechanism involved has advanced. However, models of ruminant metabolism have failed to keep pace with such developments and as discussed earlier, it seems unlikely that the more basic statistical models will provide a quick solution to quantifying the effects of supplemental fatty acids. A major research initiative in this area is therefore required.

A significant barrier to applying any model that attempts to relate available nutrition to methane emissions has been the poor description of intake. Many statistical models of intake in ruminants are available, but by their nature they are tied to the specific environment in which they were created. Therefore, both mechanistic and statistical models of methane output have tended to require intake to be defined as an input to the model. This is not a problem where intake has been measured directly for a given diet, as is often the case in studies directed at model evaluation, but it does create substantial problems where models are required to predict future emissions with no estimates of likely intake, such as is the case for inventory purposes.

It has been shown that as the rate of fermentation increases due to the feeding of increased readily fermentable carbohydrate, the rate of methane emission declines per unit of feed degraded (Pelchen and Peters, 1988). This fits neatly with observations of reduced methanogenic activity as rumen pH declines. However, existing models do not account directly for this effect, with even the most advanced mechanistic schemes including only a crude effect of pH on cellulolytic activity (Dijkstra et al., 1992). Simulating the effects of rumen pH on the diverse microbial groups present in the rumen remains a challenge, but perhaps an even greater task is to model adequately the diurnal fluctuations in pH itself.

Figure 5 shows the concept of sources and sinks for excess reducing power produced during the anaerobic fermentation process with conversion to methane acting as the final sink following all other transactions. Our knowledge of fermentation biology is more advanced than this scheme would suggest, with several other sources and sinks likely to play a role in the overall process. Compounds such as nitrates and sulphates act as sinks and are unaccounted for in the present model. Oxygen transfer at the rumen epithelium may be significant but experimental estimates are lacking. Apparently minor microbial species may also play a role in controlling emissions. Czerkawski (1986) suggests that in some animals acetotrophic methanogens could convert acetate directly to methane whilst conversely acetogens could lead to lower methane emissions in other animals (Joblin 1999). Dynamic models have yet to include these effects and progress is likely to remain limited in these areas without additional quantitative observations.

Non-nutritional factors including the effects of direct pharmacological interventions such as rumen modifiers (e.g. ionophores) and selective vaccination have yet to be incorporated into mechanistic models. Yet again, the lack of distinct subdivisions between distinct microbial groups in existing models has thwarted what could have been a relatively straightforward application of these simulation models whereby one or another group is selectively limited or destroyed. Based on genetic sequencing observations Whitford et al. (1997) suggest that only a small fraction of rumen microbes have in fact been cultured, with quantitative metabolic data on known species appearing infrequently. At the same time we can assume that methane control agents such as vaccines are likely to display high host specificity. Models that fail to account for these targeted responses will be of limited use in assessing their likely impact on animal production systems. The task of incorporating additional microbial groups is not a simple one given the numerous metabolic relationships present with other groups or species. For example, Dijkstra (1994) describes the integration of an explicit protozoal pool within the model of rumen function. At first glance this should create the potential to account for the variable effects of defaunation on methane production (Hegarty, 1999). However, to be properly representative for a broad range of diets such an exercise would require additional elements describing specific microbial species and their ability to live in or on specific species of protozoa Finlay et al. (1994).

Another, problem occurs when one tries to model the long term consequences of such interventions. Extant models of rumen function have been developed to describe the steady state biology as characterised by an animal well adapted to its diet and environment. In practice, intervention studies rarely consider treatments in these terms leading to questions about the longer term efficacy of these methods.

Indeed Johnson et al. (1994) indicated that the effect of monensin and lasalocid was not sustained beyond 16 days of treatment. This is most likely the result of adaptation by the rumen microorganisms given their short generation time and genetic diversity. If enough data were available to describe the long term effects of pharmacological treatments, there would still exist an opportunity to model the adaptation to these treatments. This applies equally to the potential to model the transitional response following a conventional nutritional change.

Existing modelling research has focused upon animals adapted to their diet for good reason, namely because this is quantitatively more important as far as emission inventories and animal production are concerned. However, the process of adaptation from one diet to the next is of significant interest. A quantitative description of the transitional phase would further our understanding of gastric fermentation, possibly identifying key control mechanisms in the shift from one type of fermentation to another. Any attempts to model this phase would almost certainly require a more reductionist approach than has been published previously for models of whole rumen function. Microbial competition for nutrients will impact on fermentation end products with the relative proportions of different microbial groups dependent on their abilities to metabolise dietary nutrients, intermediate compounds from other groups and even other microbes themselves such as in the case of protozoa and their predatory behaviour. A model capable of accurately simulating transient changes in emissions through dietary manipulation will have to account for these inter-microbial relationships.

A longer term view is also important when considering how to account for changes in methane emissions with the inevitable progression from one physiological state to another. As time advances, animals grow and develop and they move from juvenile to adulthood. Subsequently they will also experience physiological changes as they progress through pregnancy and lactation. Homeorhetic control throughout this development is brought about through the endocrine system, itself a function of an individual's genes. Extant models assume these factors will ultimately manifest themselves as effects of intake and nutrition. However, it is conceivable that other modes of action to affect methane emissions are possible. For instance, it is known that intestinal morphology can change substantially following parturition with Gibb et al. (1992) demonstrating a 15% increase in small intestinal length between calving and mid-lactation in Holstein dairy cows. The same study showed that a similar increase in mass was observed for rumen tissues and rumen contents. These changes will affect the extent of degradation in the rumen and also the quantity of nutrients available for fermentation in the hind gut. Unfortunately, the invasive and expensive nature of the experimentation required to provide further quantitative estimates of such physiological changes will limit the available data for model construction an evaluation.

### 3.3.1.6 References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### 3.3.2 Nitrous oxide

#### 3.3.2.1 Introduction

The UK is committed to the United Nations Framework Convention on Climate Change, which was agreed in 1990 and came into force in 1994. European Union countries adopted the Kyoto Protocol in 1997 and agreed a reduction of GHG emissions of 8% of 1990 levels by 2012, with the UK agreeing to a reduction of 12.5% as part of the overall EU 'bubble'. The UK government's draft Climate Change Bill, introduced in March 2007, aims to unilaterally implement statutory target reductions of $CO_2$ emissions of 60% (not including international aviation and shipping emissions) by 2050, and 26-32% by 2020, against 1990 baseline levels.

Although agriculture is only responsible for *ca*. 7% of the UK's total greenhouse gas emissions, agriculture was the source of *ca*. 66% of UK emissions of nitrous oxide ($N_2O$) in 2006, with 61% of this $N_2O$ emitted from agricultural sources arising from direct soil emissions and 32% from indirect sources (N deposition and nitrate ($NO_3$) leached). Based on the current Intergovernmental Panel on Climate Change (IPCC) inventory methodology, UK agricultural emissions of $N_2O$ were 81.8 kt in 2006 (where one kt = kiloton = 1000 metric tons). Nitrous oxide is a potent greenhouse gas, with an approximately 300-fold greater global warming potential than $CO_2$. Nitrous oxide is also involved in stratospheric ozone depletion (Cicerone, 1987).

Nitrous oxide is produced by soil microbial processes (mainly nitrification and denitrification). Nitrification is the aerobic oxidation of ammonium ($NH_4^+$) (supplied from soil mineralization processes, additions of excreta, fertiliser inputs and atmospheric deposition) to nitrate ($NO_3^-$) with the evolution of $N_2O$ (as well as $NO_x$ gases) as the result of some of the intermediary biochemical processes conducted by the microbial populations involved. Denitrification is the anaerobic reduction of $NO_3^-$, ultimately to $N_2$ gas, but with $N_2O$ as an intermediate product. Both processes are highly variable under field conditions due to the temporal and spatial nature of the major controlling factors. Under high soil moisture conditions, the main product of denitrification is $N_2$, decreasing relative loss of $N_2O$ compared with $N_2$ emission.

The major factors influencing rate of production of $N_2O$ are; percentage water filled pore space (Dobbie & Smith, 2003b), inorganic N supply (ammonium and nitrate) e.g. as inorganic fertiliser (Eichner, 1990), and supply of a readily available form of carbon, e.g. via slurry applications. As with all microbially mediated processes, temperature and pH are also controlling factors for $N_2O$ production.

#### 3.3.2.2 Main sources

Before describing the specific $N_2O$ emission sources from UK dairy production, it is worth setting such emissions in perspective. In 2006, the UK total agricultural $N_2O$ emission was 81.8 kt (IPCC revised 1996 Guidelines – see Table 5). The trend in total annual $N_2O$ emissions from agriculture in the UK has been a reduction of 21% from 1990 to 2006. This has been principally as a result of reduced livestock numbers and inorganic fertiliser use. A recent Defra project (SFF0601 – *Baseline projections for UK agriculture*) estimated that the change in agricultural land use, decrease in nitrogen fertiliser use and decrease in livestock numbers would result in a reduction of $N_2O$ emissions by *ca*. 3% between 2004 and 2025. However, this project (SFF0601) was completed prior to the large increase in fertiliser prices, which may result in less inorganic N fertiliser being applied to land and thus reduced $N_2O$ emission from this source

Table 5. Total nitrous oxide emissions from UK agriculture (source: IPCC methodology for the UK).

| kt N2O/yr | 1990 | 1995 | 1998 | 1999 | 2000 | 2001 | 2002 | 2003 | 2004 | 2005 | 2006 |
|---|---|---|---|---|---|---|---|---|---|---|---|
| **Total** | 103.9 | 98.9 | 99.3 | 97.2 | 93.4 | 87.8 | 89.5 | 87.4 | 87.0 | 85.8 | 81.8 |
| **Animals** | 5.5 | 5.4 | 5.5 | 5.4 | 5.1 | 4.8 | 4.7 | 4.6 | 4.6 | 4.5 | 4.5 |
| **Direct** | 62.7 | 60.3 | 60.9 | 59.5 | 57.6 | 53.8 | 55.4 | 54.1 | 53.8 | 53.0 | 50.0 |
| **Indirect** | 34.8 | 32.6 | 32.5 | 31.8 | 30.2 | 28.6 | 28.8 | 28.1 | 28.1 | 27.8 | 26.7 |
| **Bio fix by improv grass** | 0.5 | 0.5 | 0.5 | 0.5 | 0.5 | 0.5 | 0.5 | 0.5 | 0.5 | 0.5 | 0.6 |
| **Burning** | 0.3 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 |

There are both direct and indirect sources of $N_2O$ from agriculture. The sources of direct $N_2O$ emissions include; inorganic nitrogen fertiliser applications to soil, manure applications to soil, deposition of urine and dung at grazing, ploughing in crop residues, cultivation of histosols and biological fixation of N. Indirect sources of $N_2O$ include emissions associated with deposition of nitrogen from the atmosphere (following emissions of $NO_x$ and ammonia) and nitrate leaching. Emissions during storage of manure are also considered indirect but are reported under manure management (animals).

Emissions of $N_2O$ in Table 5 are categorised by source according to IPCC definitions, where 'Animals' = $N_2O$ emissions from manure stores and livestock buildings, 'Direct' = $N_2O$ emissions from soil, viz. N fertiliser applications, manure applications, grazing deposits (dung, but particularly urine) and crop residues, 'Indirect' = $N_2O$ emissions associated with nitrate leaching and N deposition to agricultural land.

The relative annual losses of $N_2O$ for the UK dairy sector from each of these sources can be seen in Table 6. These data are for the year 2005 and were used to inform the Defra report - *Market Mechanisms for Reducing Greenhouse Gas Emissions from Agriculture, Forestry and Land Management*. It is worth noting that extracting information from the IPCC spreadsheet and attributing emissions to different livestock categories is not a straight forward exercise and relies on various assumptions, e.g. the quantity of fertiliser N applied to the different land uses. Hence, values when added together do not necessarily exactly match the IPCC total value for the same year. In this case, the IPCC agreed UK total $N_2O$ emission for 2005 was 85.8 kt (Table 5), compared to the disaggregated emission (Table 6) of 83.1 kt, for the same year. Nevertheless, the relative differences are useful to note. It is clear that $N_2O$ loss from manure and inorganic N fertiliser spreading, as well as deposition of dung and urine during grazing are the major direct sources from the dairy sector, whilst N leaching is the major indirect source of $N_2O$ emission.

The dairy sector was responsible for *ca*. 17% of the total agricultural emissions of $N_2O$ in 2005 (Table 6), with the sheep sector being responsible for a similar percentage. The greatest source of $N_2O$ appears to be the beef sector due to a greater number of animals (*ca*. 24% of the total agricultural emission). The pig and poultry sectors are not necessarily the small sources that they first appear to be in Table 6, since some of the $N_2O$ emitted from the fertiliser and crop residue sources under 'arable' would be associated with crops grown to provide grain for these livestock groups.

Table 6. Sources of nitrous oxide emissions from different agricultural sectors (2005).

| | **Direct emissions** | | | | | **Indirect emissions** | | |
|---|---|---|---|---|---|---|---|---|
| kt/yr | **Manure spreading** | **Storage and housing** | **Grazing** | **N fertiliser** | **Crop residues** | **N leaching** | **N deposition** | **TOTAL** |
| **Dairy** | 2.26 | 0.75 | 3.02 | 2.91 | - | 3.94 | 1.02 | 13.90 |
| **Beef** | 2.94 | 2.02 | 4.92 | 2.91 | - | 5.38 | 1.52 | 19.68 |
| **Sheep** | 0.09 | 0.11 | 5.76 | 1.00 | - | 4.01 | 1.03 | 12.00 |
| **Pigs** | 0.69 | 0.68 | 0.09 | 3.05 | - | 1.13 | 0.27 | 5.91 |
| **Poultry** | 1.33 | 0.53 | 0.14 | 1.18 | - | 1.43 | 0.36 | 4.97 |
| **Arable** | - | - | - | 11.17 | 7.99 | 6.55 | 0.97 | 26.68 |
| **TOTAL** | 7.30 | 4.09 | 13.92 | 22.22 | 7.99 | 22.44 | 5.17 | 83.13 |

#### 3.3.2.3 Direct sources

Below, we review the IPCC emission factors and some of the literature evidence for each of the direct $N_2O$ sources within the dairy sector. Table 11 summarises the range of emission factors derived from a selection of the literature, as well as the IPCC guidelines currently used for estimating the UK's agricultural $N_2O$ emissions (revised 1996 Guidelines) as well as the updated 2006 IPCC Guidelines. (The UK still uses the revised 1996 IPCC Guidelines and 2000 Good Practice Guidance for compiling its agricultural $N_2O$ emissions inventory there is no requirement to use the 2006 Guidelines until the next commitment period).

##### 3.3.2.3.1 Manure applications to soil

Livestock manure supplies a supply of inorganic N, readily available carbon and moisture to soil, enhancing conditions for denitrification and $N_2O$ emission. Different manure types will vary in the inorganic N and carbon supply, depending on diet, addition of bedding material, manure storage conditions and/or treatment prior to application. The rate of application will affect the magnitude of $N_2O$ fluxes as will timing of application. For example, Thorman et al. (2007) demonstrated that emissions of $N_2O$ were reduced by 50% following slurry application in Spring compared to Autumn.

As part of the development of the MANNER-NPK decision support system, ADAS and North Wyke Research collated $N_2O$ emission factors for manure spreading to grassland and tillage land from a range of experiments. Average emission factors derived from 51 slurry applications, 27 FYM applications and 14 poultry manure applications were 0.57, 0.28 and 0.75 of the total N applied, respectively (Thorman et al., 2006) These average emission factors are all below the IPCC revised 1996 guidelines default emission factor for manure applications to land, 1.25% of N applied (allowing for $NH_3$ and $NO_x$ leaching), although, these values may be an underestimate of the annual emission because of the short term nature (< 3 months) of many of the measurements. The 2006 guidelines standard emission factor has been reduced to 1%, as a result of more recent data becoming available. However, in the 2006 guidelines, for the Tier 1 approach, the amounts of applied organic nitrogen fertilisers are no longer adjusted for the amounts of $NH_3$ and $NO_x$ volatilisation after application to soil.

##### 3.3.2.3.2 Manure in buildings and stores

Solid manure as bedding and in stores are sources of $N_2O$ emissions, whilst slurry in buildings and slurry stores are not thought to be $N_2O$ sources due to their anaerobic nature (i.e. the mineral form of N remains as $NH_4$ in the slurry as there is little opportunity for nitrification to $NO_3$, unless there is active aeration of the slurry or the development of a crust, which then becomes the source, Table 11). Chadwick (2005) demonstrated that the emission factor can range between 0.1 and 3.0% of the initial heap N during the first 3 months of cattle FYM storage. Other researchers have measured lower values (see Table 7). Much will depend on the mineral N content of the FYM as this is the source of the $N_2O$ emission. The C:N ratio, moisture content and density of the FYM will also affect $N_2O$ emissions, since these factors affect the composting process and heat generation. Parkinson et al. (2004) have shown that active composting of cattle FYM increases $N_2O$ emissions, presumably through stimulation of microbial activity and heat generation following increased aeration of the heap.

There are a range of emission factors used by IPCC for manures in buildings and manure stores, ranging from 0% of manure N, e.g. for a slurry store without a natural crust to a value of 10% (IPCC 2006 guidelines) for a solid manure that is actively mixed in windrows (Table 11).

3.3.2.3.3 Deposition of urine and dung at grazing

Emissions of $N_2O$ from dairy-grazed pastures are temporally and spatially variable due to uneven distribution of grazing returns, soil heterogeneity, animal treading and water and temperature variations (Saggar, 2004, 2007; Bhandral et al., 2007). Urine deposition is the major source of $N_2O$ emissions from grazing livestock, due to the high available N content. A urination event can deliver the equivalent of up to 1000 kg N/ha to a small area of grassland, which is far in excess of plant uptake. Hence, the excess N in the soil is a risk of loss to the environment as $N_2O$ and nitrate leaching. There is little $N_2O$ emission from dung, since the main form of N is in the organic form. Compaction by livestock on wet soil will enhance conditions for $N_2O$ emissions.

The time/season of urine deposition also influences emissions of $N_2O$ (Allen et al., 1996), see Table 11). Emission rates will also be influenced by diet, soil type and conditions. The IPCC default emission factor is 2% of applied N (in grazing returns) under the revised 1996 guidelines for all stock types (dairy cows, beef cattle, sheep, pigs and poultry). The IPCC 2006 guidelines have separated sheep from the other grazing stock types, with an emission factor of 1% for sheep (and 2% for the remaining stock types). A recent review (de Klein, 2004) indicated that the lower emission factor for sheep is due to more even urine distribution (smaller and more frequent urinations) and smaller effects on soil compaction during grazing.

3.3.2.3.4 Inorganic nitrogen fertiliser applications to soil

Emission rates of $N_2O$ after inorganic N fertiliser applications are controlled by rate and timing of application, soil type and conditions and potentially by the form of N applied. Eichner (1990) reviewed published literature and showed greater $N_2O$ emission factors from ammonium nitrate compared to urea (see Table 11). Dobbie and Smith (2003a) measured $N_2O$ fluxes after application of different forms of N fertiliser, demonstrating reduced emissions from urea compared with ammonium nitrate in the first month after application to grassland plots in June. However, a Defra funded study (NT2605) showed no clear differences in $N_2O$ emission factors between ammonium nitrate, urea and urea ammonium sulphate (see Table 11), across a range of grassland sites.

There are few data on the effect of application rate on $N_2O$ emissions, so a Defra project (AC0101) is currently assessing the relative $N_2O$ emission factors following the application of ammonium nitrate at a range of application rates to tillage land and grazed grassland. Preliminary results suggest that emission factors are not independent of application rate (as assumed in the IPCC methodology) and that there is a non-linear relationship between application rate and $N_2O$ emission factors. The information from these studies will feed into the new Defra Inventories project (AC0112) to help develop a smarter UK agricultural $N_2O$ inventory.

The IPCC revised 1996 guidelines default emission factor for inorganic N fertiliser application to land is 1.25% of N applied (allowing for $NH_3$ emissions and $NO_3$ leaching). The 2006 guidelines standard emission factor has been reduced to 1%, as a result of more recent data becoming available. However, in the 2006 guidelines, for the Tier 1 approach, the amounts of applied mineral nitrogen fertilisers are no longer adjusted for the amounts of $NH_3$ and $NO_x$ volatilisation after application to soil.

3.3.2.3.5 Crop residues

The IPCC revised 1996 guidelines default emission factor for crop residues is 1.25% of N incorporated into the soil. The 2006 guidelines standard emission factor has been reduced to 1%, as a result of more recent data becoming available.

#### *3.3.2.4* Indirect sources

Emission factors for indirect $N_2O$ emissions are 1% of N volatilised and re-deposited and 2.5% of N leached/runoff (IPCC revised 1996 guidelines), and 1% of N volatilised and re-deposited and 0.75% of N leached/runoff (IPCC 2006 guidelines). In the revised 1996 IPCC guidelines it is assumed that 10% of applied N in inorganic fertiliser and 20% of applied N in organic manures or in grazing returns (dung and urine) is volatilised, and that 30% of applied N in inorganic fertilisers, organic manures or in grazing returns is lost as nitrate leaching/runoff.

#### 3.3.2.5 Other sources

One further category of $N_2O$ and $CH_4$ sources is the 'unmanaged' non-productive areas of fields, e.g. gateways, tracks, ditches, seepage/wet areas, and areas around drinking and feeding troughs. These areas are either frequented by grazing livestock and hence receive direct dung and urine deposition, or they receive seepage from manure stores and field heaps, or uncontained runoff from hard standings. They also tend to become compacted and waterlogged, thus enhancing conditions for $N_2O$ and $CH_4$ emissions. The increased potential for large emissions from these areas are not accounted for in the IPCC inventory or any farm-scale carbon/GHG calculators. A Defra project (AC0101) is currently assessing the relative contribution of these areas to the total $N_2O$ and $CH_4$ emissions from dairy and beef farms.

#### 3.3.2.6 Mitigation options

Mitigation methods should be aimed at reducing emissions from the major sources of $N_2O$ from the dairy farm. Mitigation can be targeted at both direct and indirect $N_2O$ emissions. Methods to reduce direct $N_2O$ emissions from agriculture have been reviewed. For example, Monteney et al. (2006) discussed management practices such as: selection of fertiliser type/form, use of nitrification inhibitors, increased land drainage, compaction of solid manure stores, and choice of management system (slurry or solid manure).

A recent Defra project (AC0206) – *A review of best practices to reduce greenhouse gases from agriculture* – categorised a range of mitigation methods into those that could be implemented immediately, those that were more speculative and needed more UK based evidence before taking forward, and those which required a concerted R&D effort before assessing their cost-effectiveness. Some of these mitigation methods are applicable to dairy farmers, and they are listed below:

##### 3.3.2.6.1 Current methods to reduce nitrous oxide and methane emissions

Based on current scientific evidence and understanding of GHG emissions derived from previous (largely) Defra funded research, the review of research *A review of best practices to reduce greenhouse gases from agriculture* (Defra project AC0206) identified mitigation methods *currently* available for farmers and land users to follow/use as best practice to reduce GHG emissions. Four of the mitigation methods apply solely to reducing $N_2O$ emissions whilst two methods apply to reducing $CH_4$ emissions (although one may also result in reduced N excretion and hence contribute to reduced emissions of $N_2O$). The methods are:

Nitrous oxide methods

- Do not exceed crop N requirements (RB209/PLANET).
- Make full allowance of manure N supply (MANNER).

- Spread manure at appropriate times/conditions.
- Increase livestock nutrient use efficiency.

Methane methods

- Make use of improved genetic resources.
- Make use of anaerobic digestion technology for farm manures and slurries.

Further mitigation methods for reducing $CO_2$ emissions and enhancing C storage are also summarised in the AC0206 report. The most relevant for farmers in terms of current best practice include establishing permanent grasslands/woodlands and growing biofuel/biomass crops. The report did not include direct $CO_2$ emissions or C storage within its scope, so no further attention was paid to it.

The AC0206 report also identified a number of methods that offer the potential to reduce emissions of $N_2O$ and $CH_4$ but which are currently at a stage where further evidence of their efficacy and/or long-term consequences is required. These include reduced/zero tillage methods, movement of stock off 'wet' ground to reduce poaching, change from a solid manure to slurry system (to reduce $N_2O$ emissions), use of methods to increase dairy cow efficiency (nutrition, breeding), use of nitrification inhibitors, mineral N fertiliser timing strategies, and the use of plants with improved N use efficiency. These methods do not have sufficient evidence from which to estimate reduction efficiencies and recommend as best practice for UK farmers at this time.

Finally, the report also identified a number of speculative methods that are still at the concept stage, where some evidence for the potential to reduce $N_2O$ and $CH_4$ emissions is available, but the methods by which they could be implemented and/or what their true potential is, are unknown at this time. These include methods for improved animal feed characterisation, vaccination of ruminants against methanogenic rumen bacteria, modification of the rumen population by antibiotics and natural products, and natural nitrification inhibitors produced by crops.

Methods that are considered to be practical now (not listed in any order of priority), with a robust scientific evidence base are summarised in Table 7. Estimates of potential effects (in % of target gases: $N_2O$, $CH_4$ and $CO_2$) are given where possible. The percentage changes for $N_2O$, $CH_4$, $CO_2$, $NH_3$ and $NO_3$ relate to the *specific source*. Where estimates are not possible (i.e. there is no evidence base), a direction of change only is given: ↓ = reduction (positive effect); ↑ = increase (negative effect); ? = unknown (knowledge gap); ~ = neutral effect.

Table 7. The effect of mitigation methods on the direction and percentage changes for direct $N_2O$, $CH_4$ and $CO_2$ emissions (as well as the secondary effects on $NH_3$ emissions and $NO_3$ leaching).

| | | Source of target gas | Direct effects | | | | |
|---|---|---|---|---|---|---|---|
| | | | $N_2O$ | $CH_4$ | $CO_2$ | $NH_3$ | $NO_3$ |
| Nitrous oxide: | | | | | | | |
| 1 | Do not exceed crop N requirements | $N_2O$ - Fertiliser, Manure spreading Grazing | ↓5% | ~ | ↓ | ↓5% | ↓5% |
| 2 | Make full allowance of manure N supply | $N_2O$ - Manure spreading | ↓5% | ~ | ↓ | ~↓ 5% | ↓5% |
| 3 | Spread manure at | $N_2O$ - Manure | ↓2-10% | ~↑ | ~ | ↑10- | ↓5-15% |

| | | | | | | | |
|---|---|---|---|---|---|---|---|
| | appropriate times/conditions | spreading | | | | 20% | |
| 4 | Increase livestock nutrient use efficiency | $N_2O$ – Manure storage, manure spreading. $CH_4$ – enteric and manure | ↓6% | ↓? | ↓ | ↓3-10% | ↓2-6% |
| Methane: | | | | | | | |
| 5 | Make use of improved genetic resources | $N_2O$ – Manure storage, manure spreading. $CH_4$ – enteric and manure | ↓3% | ↓3% | ~ | ↓2-5% | ↓1-3% |
| 6 | Anaerobic digestion | $CH_4$ - manure | ? | ↓90% | ~ | ? | ? |
| 7 | Establish permanent grasslands/woodlands | | ↓? | ~ | ↓? | ~↑ | ↓50-95% |
| 8 | Grow biofuel/biomass crops | | ↓? | ~ | ↓? | ~↓ | ~↓ |

We have used the percent reductions in direct $N_2O$ emissions as a result of the mitigation methods shown in Table 7 for the UK dairy sector (year 2005 as shown in Table 6). These mitigation methods target specific sources. The effects of the individual mitigation methods are shown in Table 8, where it is clear that the most effective method is to not exceed crop N requirements (*ca*. 4% reduction in direct losses, excluding emissions related to crop residues). Other mitigation methods result in a <1 to 2% reduction in direct emissions (not including crop residues). If these mitigation methods were additive, the total reduction of direct $N_2O$ emissions would be *ca*. 8% (excluding crop residues), assuming 100% uptake of all these methods by all dairy farmers and 100% efficacy of each mitigation method.

Table 8. Impact of current mitigation methods on direct $N_2O$ emissions from the dairy sector (excluding emissions from crop residues) for the year 2005.

| kt N2O/yr | No mitigation | Don't exceed crop N reqs | Make full allowance manures | Spread manure at at app time | Incr. livestock nut. Use eff | Use imp genetic res |
|---|---|---|---|---|---|---|
| **Manure storage** | 2.26 | 2.26 | 2.26 | 2.26 | 2.12 | 2.19 |
| **Manure applications** | 0.75 | 0.71 | 0.71 | 0.68 | 0.71 | 0.73 |
| **Grazing deposition** | 3.02 | 2.87 | 3.02 | 3.02 | 3.02 | 3.02 |
| **N Fertilisers** | 2.91 | 2.76 | 2.91 | 2.91 | 2.91 | 2.91 |
| **Total** | 8.94 | 8.60 | 8.90 | 8.86 | 8.75 | 8.84 |
| **% reduction** | | 3.7 | 0.4 | 0.8 | 2.0 | 1.0 |

The AC0206 report also identified a number of methods that offer the potential to reduce emissions of $N_2O$ and $CH_4$ but which are currently at a stage where further evidence of their efficacy and/or long-term consequences is required (Table 9). These include reduced/zero tillage methods, movement of stock off 'wet' ground to reduce poaching, change from a solid manure to slurry system (to reduce $N_2O$ emissions), use of methods to increase dairy cow efficiency (nutrition, breeding), use of nitrification inhibitors, mineral N fertiliser timing strategies, and the use of plants with improved N use efficiency. These methods do not have sufficient evidence from which to estimate reduction efficiencies so are not included in this study.

3.3.2.6.2 Future potential methods to reduce nitrous oxide and methane emissions

Methods that are considered to have future potential (not listed in any order of priority), but require further research or change of regulations are summarised in Table 9. Estimates of potential effects (in %: $N_2O$, $CH_4$ and $CO_2$) are given where possible. The percentage changes for $N_2O$, $CH_4$, $CO_2$, $NH_3$ and $NO_3$ relate to the ***specific source***. The direction of change is also indicated: ↓ = reduction (positive effect); ↑ = increase (negative effect); ? = unknown (knowledge gap); ~ = neutral effect.

Table 9. The effect of mitigation methods needing further research on the direction and percentage changes for direct $N_2O$, $CH_4$ and $CO_2$ emissions (as well as the secondary effects on $NH_3$ emissions and $NO_3$ leaching).

| | | Magnitude of source and target gas (kt $CO_2$e) | Direct effects | | | Secondary & indirect effects | |
|---|---|---|---|---|---|---|---|
| | | | $N_2O$ | $CH_4$ | $CO_2$ | $NH_3$ | $NO_3$ |
| 1 | Reduced/zero tillage | | ? | ~ | ↓<1% | ~ | ↓ |
| 2 | Take stock off 'wet' ground | Fertiliser, manure spreading, grazing returns | ↓ | ↑? | ~ | ↑? | ↓? |
| 3 | Change from a solid manure to slurry system | Manure storage, fertiliser applications | ↓15% ↓780 kt | ↑ | ↑ | ↑ (cattle) ↓ (pigs) | ↑ |
| 4 | Use of hormones and increased milking frequency | $CH_4$ –dairy cattle enteric fermentation and manures | ↓ | ↓ | ~ | ↓ | ↓ |
| 5 | Use of nitrification inhibitors | $N_2O$ - fertilisers, manure spreading and grazing returns | ↓ | ~ | ~ | ~ | ↓ |
| 6 | Improved mineral fertiliser N timing strategies | $N_2O$ - fertilisers, manure spreading and grazing returns | ↓ | ~ | ~ | ~ | ↓ |
| 7 | Use plants with improved nitrogen use efficiency | $N_2O$ - fertilisers, manure spreading and grazing returns | ↓ | ~ | ↓ | ↓ | ↓ |

3.3.2.6.3 Speculative methods to reduce nitrous oxide and methane emissions

Finally, the AC0206 report also identified a number of speculative methods that are still at the concept stage, where some evidence for the potential to reduce $N_2O$ and $CH_4$ emissions is available, but the methods by which they could be implemented and/or what their true potential is, are unknown at this time (Table 10). These include methods for improved animal feed characterisation, vaccination of ruminants against methanogenic rumen bacteria, modification of the rumen population by antibiotics and natural products, and natural nitrification inhibitors produced by crops.

Estimates of potential effects (in % of target gases: $N_2O$, $CH_4$ and $CO_2$) are given where possible. The percentage changes for $N_2O$, $CH_4$, $CO_2$, $NH_3$ and $NO_3$ relate to the ***specific source***. The direction of change is also indicated: ↓ = reduction (positive effect); ↑ = increase (negative effect); ? = unknown (knowledge gap); ~ = neutral effect.

Table 10. The effect of speculative mitigation methods on the direction and percentage changes for direct $N_2O$, $CH_4$ and $CO_2$ emissions (as well as the secondary effects on $NH_3$ emissions and $NO_3$ leaching).

| | | Magnitude of source and target gas (kt $CO_2$e) | Direct effects | | | Secondary & indirect effects | |
|---|---|---|---|---|---|---|---|
| | | | $N_2O$ | $CH_4$ | $CO_2$ | $NH_3$ | $NO_3$ |
| 1 | Improved feed characterisation | $N_2O$ manure storage, manure applications<br>$CH_4$ – enteric fermentations and animal manures | ↓ | ↓ | ~ | ↓ | ↓ |
| 2 | Vaccination against methanogens | $CH_4$ – from enteric fermentation and manures | ? | ↓8%<br>↓1,460 kt | ~ | ? | ~ |
| 3 | Modification of rumen microbial fermentation by ionophores and natural extracts | $CH_4$ – from enteric fermentation and manures | ? | ↓ | ~ | ? | ? |
| 4 | Production of natural nitrification inhibitors by plants | $N_2O$ – fertilisers, manure spreading and grazing returns | ↓ | ~ | ~ | ↓ | ↓ |
| 5 | Use of cloned animals | $CH_4$ – from enteric fermentation and manures | ↓ | ↓ | ~ | ↓ | ↓ |
| 6 | Genetic manipulation of livestock | $CH_4$ – from enteric fermentation and manures | ↓ | ↓ | ~ | ↓ | ↓ |

3.3.2.6.4 Potential for the current UK IPCC inventory methodology for agricultural $N_2O$ emission to reflect mitigation methods

It is clear that under the proposed IPCC methodology for constructing the UK inventory for agricultural $N_2O$ emissions, that currently the only way of reflecting mitigation methods is by reducing N fertiliser use or reducing livestock numbers. A new Defra project (AC0112) has been commissioned to further develop this emissions inventory, moving it towards a Tier 2 and Tier 3 methodology. This is a five year project and the aim is for the new structure to facilitate a more direct reflection of mitigation methods than our current inventory. The aim of this project is also to link the three inventories, $N_2O$, $CH_4$ and $NH_3$, together.

#### 3.3.2.7 Models of nitrous oxide emission

Table 11. Emission factors for direct sources of $N_2O$ from agriculture – with reference to dairy and beef based systems.

| | Treatment | EF (% N applied or % N in manure store) | Reference | Measurement period |
|---|---|---|---|---|
| Manure spreading | Cattle slurry – Apr<br>Cattle slurry – July<br>Dilute Cattle slurry – Oct<br>Cattle FYM – Oct | 0.97<br>0.12<br>0.38<br>0.20 | Chadwick et al. (2000) | 1 month |
| | Cattle FYM 1999<br>Cattle FYM 2000 | 0.30<br>0.24 | Webb et al. (2004) | 2 months |
| | Cattle slurry – 2002<br>Cattle slurry – 2003 | 0.5<br>0.2 | Jones et al. (2007) | 12 months |
| | 2001<br>Cattle slurry @ 120 kg N/ha (SI)<br>Cattle slurry @ 180 kg N/ha (SI)<br>Cattle slurry @ 240 kg N/ha (SI)<br>2002<br>Cattle slurry @ 120 kg N/ha (SI)<br>Cattle slurry @ 180 kg N/ha (SI)<br>Cattle slurry @ 240 kg N/ha (SI) | 0.34<br>0.31<br>0.40<br>0.02<br>0.04<br>0.20 | Schils et al. (2008) | 12 months |
| | All manure applications | 1.25 | IPCC r1996 | Assumed 12 months |
| | All manure applications | 1.00 | IPCC 2006 | Assumed 12 months |
| | | | | |
| | | | | |
| Manure stores | Cattle FYM | <1.0 | Amon et al. (1997) | ?? |
| | Cattle FYM | <1.0 | Sommer et al. (2001) | ?? |
| | Cattle FYM<br>Cattle FYM<br>Cattle FYM | 2.3<br>0.1<br>1.3 | Chadwick (2005) | 4 months<br>3 months<br>3 months |
| | Daily spread slurry<br>Dairy spread FYM<br>Liquid system<br>Solid storage | 0.0<br>0.0<br>0.1<br>2.0 | IPCC r1996 | Assumed 12 months |
| | Daily spread slurry<br>Dairy spread FYM<br>Liquid system + natural crust<br>Liquid system – no crust<br>Solid storage<br>Pit storage below animal building<br>Cattle deep bedding | 0.0<br>0.0<br>0.5<br>0.0<br>0.5<br>0.2<br>1.0<br>7.0 | IPCC 2006 | Assumed 12 months |

| | | | | |
|---|---|---|---|---|
| | Active mixing<br>Others | 0.0 to 10.0 | | |
| | | | | |
| Grazing | Cattle – summer<br>Cattle - winter | 0.0<br>0.8 – 2.3 | Allen et al. (1996) | ?? months |
| | Cattle | 0-5 -1.0 | Yamulki et al. (1998) | 12 months |
| | Cattle | 2.3 | Sagaar et al. (2004) | 12 months |
| | All grazing stock types | 2.0 | IPCC r1996 | Assumed 12 months |
| | Cattle, Dairy cows, pigs, poultry<br>Sheep | 2.0<br>1.0 | IPCC 2006 | Assumed 12 months |
| | | | | |
| Inorganic N fertilisers | Ammonium nitrate<br>Urea | 0.44<br>0.11 | Eichner (1990) | Various studies reviewed with different monitoring periods |
| | Ammonium nitrate | 0.30 | Perala et al (2006) | 5 months |
| | <u>2002</u><br>Ammonium nitrate<br>Urea<br><u>2003</u><br>Ammonium nitrate<br>Urea | <br>1.4<br>0.4<br><br>0.1<br>0.1 | Jones et al. (2007) | 12 months |
| | <u>2001</u><br>CAN 120 kg N/ha<br>CAN 240 kg N/ha<br>CAN 330 kg N/ha<br><u>2002</u><br>CAN 120 kg N/ha<br>CAN 240 kg N/ha<br>CAN 330 kg N/ha | <br>0.11<br>0.23<br>0.17<br><br>0.14<br>0.12<br>0.12 | Schils et al. (2008) | 12 months |
| | Ammonium nitrate<br>Urea<br>Urea ammonium sulphate | 0.78 – 2.07<br>0.12 – 1.97<br>0.54 – 1.30 | NT2605 Defra study* | 12 months |
| | For all fertiliser types | 1.25 | IPCC (r1996) | Assumed 12 months |
| | For all fertiliser types | 1.00 | IPCC (2006) | Assumed 12 months |
| | | | | |
| | | | | |
| Crop residues | For all crop types | 1.25 | IPCC (r1996) | Assumed 12 months |
| | For all crop types | 1.00 | IPCC (2006) | Assumed 12 months |
| | | | | |

SI = shallow injection.
*seasonally weighted means for fertiliser applications made at four different times of the year at three grassland sites in England and Scotland.

UK-DNDC is a highly mechanistic model of carbon and nitrogen cycling in agro-ecosystems at a sub-daily time step. It consists of four interacting submodels: soil and climate, plant growth, decomposition and denitrification. Originally developed in the United States (Li et al., 1997) it has been adapted/developed by a number of countries to better reflect their agriculture, climate and soil types. The model has been adapted for UK conditions and is used to assess emissions of $N_2O$ at regional and national scale (Development and application of a mechanistic model to estimate emission of nitrous oxide from UK agriculture. $N_2O$ emissions are sensitive to soil type, cropping type, climate and nitrogen inputs. Nitrogen inputs include; inorganic N, organic N (via livestock manure applications and grazing returns), crop residues (and biological fixation). It is being developed for the UK as part of the new Defra funded inventory project (AC0112) with the aim of using the revised model to generate country, and even site, specific emission factors. Emissions of $N_2O$ are expressed as kg/ha/yr.

Despite the growing literature on nitrification, denitrification and emissions of $N_2O$ from the range of agricultural sources, prediction of emissions remains challenging and has a high degree of uncertainty depending on time and rate of application (fertiliser, manure), time of urine deposition, soil type and soil conditions (water filled pore space, soil compaction). Hence, the UK currently uses IPCC Tier 1 methodology to estimate annual $N_2O$ emissions. Defra has recently commissioned a five year research project to develop the UK IPCC approach to estimating $N_2O$ emissions, which would better reflect UK soil and climate conditions and potential mitigation methods.

#### 3.3.2.8 Knowledge gaps

The current IPCC $N_2O$ inventory methodology for the UK needs to be further developed to include country specific emission factors which better reflect the UK's soil, climate and management systems. This means development to Tier 2 and eventually Tier 3 methodology. Further review and experimental work is required to generate country specific emission factors for the main agricultural sources of $N_2O$. The new five year Defra funded inventory project will work towards this.

Mitigation methods need to be developed, tested and validated in the field, and the IPCC inventory developed to reflect the main mitigation methods. The new five year Defra funded inventory project aims to provide a framework that acknowledges key mitigation methods.

The effect of diet on N excretion and subsequent losses of $N_2O$ and $NH_3$ is still not fully understood, and obviously linked to effects of dietary manipulation to control methane emissions.

The methane, nitrous oxide and ammonia emissions inventories need to be linked to ensure that the same activity data are used in all three inventories, enable efficient updating and importantly to reflect the impacts of mitigation methods for one gas on the other gases.

### 3.3.2.9 References


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### 3.3.3 Ammonia

#### 3.3.3.1 Introduction

Emissions of ammonia to the atmosphere are of concern because of the damaging effects of subsequent deposition to sensitive ecosystems through eutrophication and soil acidification (Fangmeier et al., 1994) and, increasingly, because of the role that atmospheric ammonia has in the formation of fine particulates which have human health implications (Hughes et al., 2002). A significant proportion of the ammonia emitted from a source can be transported in the atmosphere over long distances, crossing international boundaries, before deposition, which has led to the formulation of international legislation (UNECE Gothenburg Protocol, 1999; EC National Emissions Ceilings Directive, 2000). Under this legislation, the UK is required to comply with an agreed emission ceiling (currently set at 297 kt by 2010, but new targets are being negotiated for 2020). In addition to long range transport, much of the emission may be deposited locally (within 1 km of the source), increasing the importance of closely located significant sources to the consideration of sensitive habitat protection.

Ammonia emissions are important with respect to GHG emissions for two reasons: firstly, a proportion of the atmospheric deposition of nitrogen, which ammonia emissions contribute to, will produce nitrous oxide emissions; secondly, proper accounting of ammonia emissions in the nitrogen cycle relevant to agricultural systems is required for accurate assessment of the proportion of nitrogen from which nitrous oxide emissions can occur (and for fully accounting for the impacts of modifications to the system). Ammonia emissions from agriculture derive predominantly from the urea content of livestock urine (or uric acid in the case of poultry) and also from urea or ammonia/ammonium forms of inorganic fertiliser. Following excretion, through the action of the ubiquitous enzyme urease, urea (or uric acid) will be hydrolysed to form ammonium carbonate. Ammonia will volatilise from the ammonia in solution, the rate and extent being dependent on factors such as temperature, pH and air movement. Ammonia emissions will occur, therefore, from any areas where livestock are present (e.g. livestock housing and yards, grazing) and subsequently from manure management (storage and spreading).

In the UK, agriculture accounts for >85% of total ammonia emission (NAEI, 2008), the remainder deriving from a large number of relatively minor sources (e.g. waste water treatment, transport, composting). The most recent inventory of emissions from UK agriculture (Misselbrook et al., 2008) estimated a total emission for 2007 of 242 kt, of which 71 kt derived from livestock sources within the dairy sector (Table 12) and a further 10 kt from fertiliser applications to grassland (which are also attributable to the beef and sheep sectors). There has been a significant decline in emissions since 1990, largely due to reductions in livestock numbers and fertiliser nitrogen use (Table 13). Projections to 2020, based on forecasts of livestock numbers and fertiliser use from the Defra-funded project SSF0601 (Baseline projections for Agriculture) and a 'business as usual' scenario which anticipates introduction of certain mitigation measures on large pig and poultry farms under IPPC legislation, show little further decline in emission (Table 13). Projections show that the UK should meet the NECD target for 2010, but will be unlikely to comply with a downward revised target for 2020 without implementing further measures.

Table 12. Estimate of ammonia emission from UK agriculture for 2007.

| ***Source*** | ***kt $NH_3$*** | ***% of total*** |
|---|---|---|
| ***Livestock category*** | | |
| *Cattle* | *134.0* | *55* |
| Dairy | 71.2 | 29 |
| Beef | 62.7 | 26 |
| *Sheep*[†] | *11.0* | *5* |
| *Pigs* | *20.9* | *9* |
| *Poultry* | *31.7* | *13* |
| *Horses* | *4.9* | *2* |
| | | |
| ***Management category*** | | |
| *Grazing/outdoors* | *30.3* | *13* |
| *Housing* | *64.8* | *27* |
| *Hard standings* | *22.7* | *9* |
| *Manure storage* | *31.1* | *13* |
| *Manure application* | *53.5* | *22* |
| | | |
| *Fertiliser application* | *39.7* | *16* |
| | | |
| ***TOTAL*** | ***242.1*** | |

*[†]Including goats and deer.*

Table 13. Estimates of ammonia emission from UK agriculture 1990 – 2020.

| *Source* | *1990* | *1995* | *2000* | *2005* | *2007* | *2010* | *2015* | *2020* |
|---|---|---|---|---|---|---|---|---|
| | | | | | | *Projections* | | |
| ***Total*** | **315.6** | **290.9** | **264.2** | **248.0** | **242.1** | **241.7** | **242.1** | **240.6** |
| | | | | | | | | |
| *Cattle* | 153.7 | 150.2 | 143.5 | 137.5 | 134.0 | 133.3 | 131.3 | 130.6 |
| Dairy cattle | *81.7* | *84.9* | *80.5* | *75.9* | *71.2* | *74.6* | *73.9* | *74.5* |
| Other cattle | *72.0* | *65.3* | *63.0* | *61.5* | *62.7* | *58.6* | *57.4* | *56.1* |
| *Sheep* | 14.0 | 14.1 | 13.7 | 11.6 | 11.0 | 11.5 | 11.4 | 11.3 |
| *Pigs* | 42.0 | 39.8 | 31.3 | 22.0 | 20.9 | 19.9 | 19.2 | 18.4 |
| *Laying hens* | 13.6 | 12.0 | 10.0 | 9.5 | 8.5 | 9.1 | 9.0 | 8.6 |
| *Other poultry* | 25.9 | 24.3 | 28.2 | 24.9 | 23.2 | 24.8 | 24.7 | 24.8 |
| *Horses* | 2.6 | 3.4 | 3.7 | 4.4 | 4.9 | 5.1 | 5.8 | 6.5 |
| *Fertiliser* | 63.7 | 47.1 | 33.7 | 38.1 | 39.7 | 38.1 | 40.7 | 40.4 |

#### 3.3.3.2 Main sources

Ammonia emissions from the dairy sector derive from livestock housing, including the use of outdoor concrete yards (for collecting and/or feeding animals), manure storage, manure spreading and cattle grazing (Table 14 shows the estimated split between sources for 2007). Emissions also derive from fertiliser applications to grassland used both for grazing and conserving (as silage or hay), although it is difficult to provide a robust estimate of the proportion of fertiliser applications to grassland which are attributable to the dairy sector. Cattle housing and manure application to land are the major emission sources, although significant emissions also arise from hard standings used by cattle and manure storage areas.

Table 14. Ammonia emissions (kt $NH_3$) from sources within the dairy sector, 2007.

| Source | Emission (kt $NH_3$) |
|---|---|
| Housing | 21.4 |
| Hard standings | 14.0 |
| Manure storage | 11.4 |
| Manure application | 18.8 |
| Grazing | 5.6 |
| Fertiliser to grassland (all sectors) | 10.2 |

#### 3.3.3.2.1 Cattle housing

For estimating ammonia emissions, cattle housing is categorised into two systems: slurry-based (cubicle housing) and straw-bedded solid floor. Current estimates are that 66% of dairy cows, 18% of followers and 0% of calves are housed on a slurry-based system, with the remainder in straw-bedded housing. A direct comparison study using the experimental housing facilities at North Wyke (Gilhespy et al., in press) measured greater emissions from slurry-based systems than for straw-bedded systems (Defra project WA0632). This difference was considered to be due to both the physical effect of the straw, acting as a barrier to air movement from the emitting surface, and to the carbon addition with the straw leading to increased immobilisation of the readily available N. Lower emissions from straw-bedded systems has been confirmed through the results of a number of measurement studies, both on commercial farms and experimental facilities, from which mean emission factors of 37 (range 29-51) and 22 (range 13-35) g/lu/d $NH_3$-N have been derived for slurry and straw-based systems, respectively (Defra projects AM0102, AM0103, WA0618, WA0632, WA0653; Demmers et al., 1997; Dore et al., 2004; Hill, 2000). Measurements from housed calves have shown even lower emissions (Demmers et al., 1997, Koerkamp et al., 1998), with a mean of 12 g/lu/d, as the ratio of straw to excreta tends to be greater. Mean emission factors expressed as a proportion of the readily available nitrogen deposited in the house are estimated as 32, 23 and 8% for slurry-based, straw-bedded adult and straw-bedded calves, respectively.

Factors influencing emissions from cattle housing (in addition to housing 'type') can be categorised as those influencing the source area and/or concentration and those influencing the transfer of ammonia from the source to the atmosphere. Ammonia emissions are strongly related to the exposed surface area of the source and therefore an inverse relationship will exist between emission expressed per livestock unit and stocking density within the house (e.g. Misselbrook et al., 2006; Defra project AC0102). The quantity of readily available nitrogen deposited within the house from which emission can occur will be related to the length of time the cattle spend in the house. However, interactions between volume of excreta and exposed surface area, and amount remaining after scraping, result in a non-linear relationship between time spent in the house and emission, with no significant reduction in emission from the house from cattle housed for 12 h compared to 24 h (Gilhespy et al., 2006). Cleaning method and frequency will influence the efficiency with which excreta are removed from the floor, and hence ammonia emission. Factors influencing the transfer of ammonia from the source to the atmosphere include temperature, ventilation and the physical barrier provided by bedding. Theoretical consideration would predict increasing emissions with increasing temperature and air movement, and a trend for this was observed in measurements in Defra project AM0102. We would therefore expect seasonal variation in emissions from cattle housing, although there will be interactions with seasonal changes in the duration of housing, as discussed above, and indeed with cattle diet. In straw-bedded systems, increasing the quantity of straw bedding used has been shown to reduce emissions (Gilhespy et al., in press), due to the increased physical barrier to emissions from infiltrated urine and also the increased potential for immobilisation of readily available N.

A fairly detailed mechanistic model of ammonia emissions from slatted floor dairy cow cubicle housing is given by Monteny et al. (1998), although this would require some modifications and re-parameterisation to be applied to UK cubicle house systems.

#### 3.3.3.2.2 Outdoor yards

A significant number of dairy farms have unroofed concrete yards used for collecting cows prior to milking (dairy cow collecting yards) or for feeding cattle. A survey of hard standings (Webb et al., 2001) estimated that 65% of dairy cows use collecting yards (in both winter and summer) and 30% of dairy cows and associated followers use feeding yards (winter only). Ammonia emissions from excretal returns on yards are much greater than returns to pasture because of the lack of infiltration on a concrete surface. Measurements indicate that an average of 75% of applied urine N will volatilise (Misselbrook et al., 1998, 2006).

Emissions from concrete yards will be influenced by the efficiency with which excreta (urine in particular) are removed from the yard, and by the environmental conditions (temperature, wind speed, rainfall). Survey data suggest that dairy cow collecting yards are cleaned (mostly by scraping, but a proportion by washing) on a daily basis while feeding yards may only be scraped once every few days (Webb et al., 2001). Removal efficiency by daily scraping has been estimated at 60%, based on limited measurements (Misselbrook et al., 2006). As with cattle housing, the livestock density on the yard will also be an influencing factor, with emissions per animal decreasing as stocking density increases.

#### 3.3.3.2.3 Manure storage

Cattle slurry may be stored in an above ground slurry store (*c*. 33% according to survey data), an earth-banked lagoon (*c*. 33%), a weeping-wall store (*c*. 18%) or spread directly from the house/yard with no storage (*c*. 16%). Stored farm yard manure (*c*. 70%) from straw-bedded housing is stored in a heap either on a solid concrete base or in a field.

The ammonia emission factor for cattle slurry storage has been derived from a number of measurements (Hill et al., 2000; Misselbrook et al., 2005a; Philips et al., 1997; Smith et al., 2007; Defra projects WA0625, WA0632, WA0641, WA0714, WA0717), with an overall mean emission factor per unit surface area of storage of 1.7 $g/m^2/d$ $NH_3$-N. Cattle slurry stores generally readily develop a crust, and the mean emission factor is based on measurements from crusted stores. A surface crust has been shown to reduce ammonia emissions by approximately 50% (Misselbrook et al., 2005a; Smith et al., 2007), therefore emissions from stores without a crust (e.g. if they are regularly agitated to prevent crust formation) are estimated to be 3.4 $g/m^2/d$ $NH_3$-N. Expressing the emission factors as a proportion of the readily available N within the slurry store gives mean factors when no crust is present of 10 and 50% for above ground tanks and earth-banked lagoons, respectively. Weeping wall stores are assumed to have an emission factor equivalent to crusted above ground tanks, i.e. 5% of the readily available N. The much greater emission factor for lagoons is because of the much larger surface area to volume ratio of lagoons.

The rate of ammonia emissions from a slurry store will be a function of the concentration at the emitting surface, the resistance to diffusion across the surface layer and the rate of removal from the air above the surface layer. Temperature will influence the diffusion and convection of the ammoniacal nitrogen within the slurry store and the rate of replenishment of that lost from the emitting surface. The movement across the surface layer will be influenced by the viscosity and will be greatly reduced by the presence of a physical barrier, such as a crust or floating cover. Release from the surface will be enhanced by higher temperature and pH (Sommer et al., 2006). Finally, the rate of removal of the air above the emitting surface will depend on wind speed, the height of the freeboard above the slurry surface and the presence or absence of a store cover. Mineralisation of organic N will occur during slurry storage (*c.* 10%), the rate largely dependent on temperature.

The mean emission rate from stored farm yard manure, derived from UK measurements (Defra projects WA0519, WA0618, WA0632, WA0707, WA0716) is 35% of the readily available N, although the range of measured emissions is very large (<5 - 90%). Emission rate is greatest within the first two weeks of storage, associated with the temperature increase within the heap due to composting activity, and the majority of total emission will occur within the first month of storage. Emission will be influenced by the heap temperature, moisture content and bulk density of the heap (Sommer et al., 2006), with conditions favouring thermophilic composting and high temperature increases leading to the greatest ammonia emissions. Other nitrogen losses (through leaching and denitrification) and transformations (mineralisation or immobilisation) will influence the total nitrogen content of the manure at the end of storage and the proportion that is readily available.

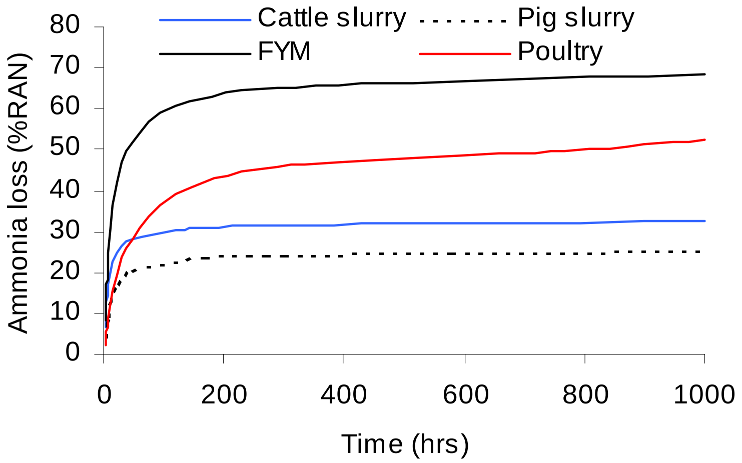
#### 3.3.3.2.4 Manure application to land

Ammonia emissions from manure applications to land can be extremely variable, depending on a number of weather, soil, manure and management factors (Misselbrook et al., 2005b; Sommer et al., 2003). A key influencing factor is the duration for which the manure remains exposed on the soil/crop surface; with emission increasing for a given surface area the longer it remains exposed. Much of the total emission from applied manure occurs within the first few hours following application. For slurries, both the dry matter content of the slurry and the ability of the soil to receive the slurry (soil texture, moisture content) will influence the rate and extent to which infiltration occurs following surface application. For slurries and solid manures, conditions which lead to large evaporative losses (high temperatures and wind speeds), will be associated with high ammonia emission rates. Rainfall will generally reduce emissions by washing available nitrogen into the soil matrix, although light rainfall may prolong emissions by keeping the emitting surface wet.

Method of slurry application and rapid soil incorporation are important management factors which influence the rate and extent of emissions. Placement of slurry in narrow bands reduces the emitting surface area compared to surface broadcast application and is generally associated with some reduction in emissions, although the increased dry matter loading rate to the soil within the slurry bands can potentially reduce slurry infiltration, therefore prolonging emission from the bands. Placement of the bands beneath a crop canopy (either by using trailing shoe to grassland swards or trailing hose on growing arable crops) will potentially further reduce emissions by reducing temperature and wind speed at the emitting surface. Rapid soil incorporation of slurries and solid manures will greatly reduce the exposed surface area and intimately mix the available nitrogen within the soil matrix.

Ammonia emissions following manure applications to land are estimated using MANNER_PSM (Defra project KT0106), where standard emission curves showing proportion of readily available nitrogen lost as ammonia against time have been developed from field measurements (Figure 6). The standard curves are then modified according to manure, weather, soil and management factors to estimate emissions for a given application.

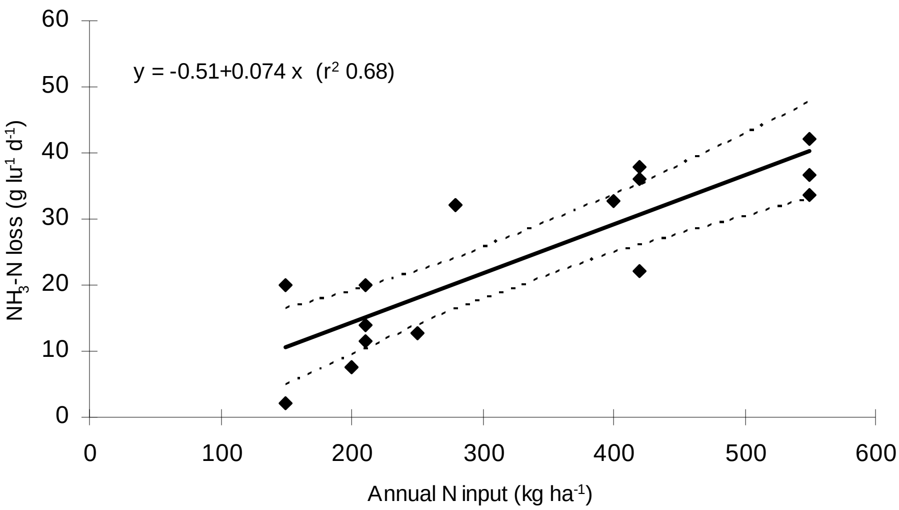
Figure 6. Standard ammonia emission curves for different manure types (from MANNER_PSM).




3.3.3.2.5 Grazing

Ammonia emissions from cattle grazing derive predominantly from the urine returns of the grazing livestock, with minimal volatilisation from dung (e.g. Petersen et al., 1998). Emissions are generally much lower than from excretal returns in livestock housing or outdoor yards because of the rapid infiltration of urine into the soil. The major factor influencing emissions is the available nitrogen concentration of the urine, which will be influenced by the nitrogen content of the herbage being grazed, which in turn will be influenced by the amount of nitrogen fertiliser applied. A good relationship has been established between fertiliser N input to grazed pastures and ammonia emission, expressed as g $NH_3$-N per livestock unit (500 kg live weight) per day (Misselbrook et al., 2000; Figure 7). Ammonia emission expressed in terms of the urine returns to the pasture is estimated to be *c.* 6% of urine N.

Figure 7. Relationship between ammonia emission (g $NH_3$-N/lu/d) and fertiliser N input to grazed pastures.



3.3.3.2.6 Fertiliser use

Ammonia emissions will occur following application of ammonia- and urea-based fertilisers. Emissions are much greater from urea-based fertilisers (urea, urea ammonium nitrate, urea ammonium sulphate) because of the high pH generated when urea hydrolyses to form ammonium carbonate. Misselbrook et al. (2004) proposed a simple model in which emissions are a function of fertiliser type, soil pH, land use, application rate, rainfall and temperature. This model was modified based on results from the Defra funded NT2605 project for use in the UK ammonia emissions inventory, deriving average emission factors for fertiliser applications to grassland of 1.8, 10, 5 and 3% of applied nitrogen for ammonium nitrate, urea, urea ammonium nitrate and ammonium sulphate/diammonium phosphate fertiliser types, respectively.

**3.3.3.3 Mitigation options**

There has been a considerable amount of research conducted aimed at developing techniques and strategies for mitigating emissions of ammonia from agricultural sources. Much of this is summarised in the Ammonia Mitigation User Manual (Misselbrook et al., 2008), and brief details of techniques relevant to the dairy sector are given below, together with a summary of potential impacts on the dairy sector of implementation of these techniques (Table 15).

3.3.3.3.1 Dietary manipulation

Avoiding excess N in the diet and/or making dietary N more available allows the concentration of N in the diet to be reduced without adversely affecting animal performance. Both reduce the amount of N excreted (and, importantly, the amount of N excreted per unit of production), thereby reducing the potential for ammonia volatilisation losses throughout the whole livestock and manure management continuum. For dairy cattle, which are predominantly on forage-based diets, and particularly while at grazing, control of dietary N intake presents a major challenge. Indeed, a good knowledge of the level of dietary N intake is likely to be lacking in most cases. However, it is also likely that in most cases, particularly at certain times of year, dietary N intake will be in excess of requirements. Ongoing research (Defra project AC0209) aims to provide potential solutions for uptake, although the difficulties in implementation within the dairy sector are not to be underestimated. Based on limited laboratory studies and controlled feeding trials, it is estimated that reductions in ammonia emission of up to 20% might be achieved through strict control of dietary N intake (e.g. Misselbrook et al., 2005c).

3.3.3.3.2 Mitigating emissions from housing

Options for mitigating emissions from dairy cattle housing include more frequent removal of slurry from cubicle housing floors, changing of housing system from slurry-based to straw-bedded, increasing bedding allowance and decreasing the time spent in the house.

For dairy cubicle houses, increasing the frequency of scraping the fouled floor areas from once to up to four times per day can reduce emissions (by up to 20%), provided this is combined with a floor design which leads to the rapid draining of urine (Braam et al., 1997). Similarly, improved cleaning of dairy cow collecting yards by pressure washing has been shown to be very effective in reducing ammonia emission (by 90% compared with daily scraping; Misselbrook et al., 2006), although the additional labour requirement and volume of additional slurry produced may inhibit uptake.

Straw-bedded housing systems are associated with lower ammonia emissions than cubicle houses (*c*. 25% lower) as discussed above, so one potential mitigation measure would be to change housing system. However, there is unlikely to be a significant move towards straw-bedded systems because of the costs of conversion, high costs and lack of availability of sufficient straw in some areas, increased manure management costs and concerns over animal hygiene and milk quality. For farms which do use straw-bedded housing, a reduction in emissions (of up to 50%) can be achieved by providing an additional 25% of bedding (above the standard 8 kg per head per day), targeted particularly to the dirty areas of the house (Defra project AM0103).

Reducing the duration of housing can be achieved by extending the grazing period from earlier in the spring to later in the autumn/winter and this will be associated with a reduction in ammonia emissions as emissions from grazing are much less than from housing (although Gilhespy et al. (2006) showed that there was only a significant reduction in housing emission if the cattle were out of the house for greater than 12 hours per day). Extended grazing is already practised to some extent in areas of suitable soil type and climate. However, there may be reluctance for further uptake because of concerns of soil compaction and sward damage associated with grazing under marginal conditions. Additionally, extending the grazing period into the late autumn/winter is contrary to advice given to avoid water pollution. The use of outwintering pads/woodchip corrals as an alternative to winter housing for part, or all, of the herd is also likely to be associated with reductions in ammonia emissions. Ongoing research seeks to provide further information on the potential use of such systems, including quantification of associated emissions of ammonia and nitrous oxide (Defra project LK0676).

#### 3.3.3.3.3 Mitigating emissions from manure storage

The main option to reduce emissions form manure storage is to cover the store (slurry or FYM). The majority of dairy slurry stores will develop a crust, due to methane and carbon dioxide emissions within the slurry bringing fibrous material to the surface, which will then dry out and form a semi-solid/solid crust. Managing the store to maintain a surface crust will reduce ammonia emissions by *c*.50% compared to non-crusted stores (Misselbrook et al., 2005a; Smith et al., 2007). A surface crust has been shown to also reduce methane emissions from slurry storage (Peterson et al., 2005), but may increase nitrous oxide emissions through nitrification and denitrification of ammonium in the crust layer. Alternatively, slurry stores may be covered with a rigid cover with a vent (above ground tanks), which will reduce emissions by c. 80%, or a floating cover (tanks and lagoons) of plastic, chopped straw, oil-based liquid, peat, bark etc., which will reduce emissions by *c*.50%. Covering of FYM heaps with heavy duty polythene sheeting can be effective at reducing ammonia emissions (by 15 - 90%, Defra project WA0716) and may also be combined with compaction prior to covering (Chadwick, 2005).

#### 3.3.3.3.4 Mitigating emissions from manure application to land

Slurry application methods such as trailing hose, trailing shoe and injection have been developed to reduce ammonia emissions following application to land (Misselbrook et al., 2002; Smith et al., 2000), as will rapid soil incorporation of slurry and FYM applied to land for cultivation.

With trailing hose, slurry is placed in narrow bands above the soil surface via trailing hoses. As ammonia volatilisation occurs from the slurry surface, applying the same volume of slurry in narrow bands rather than as an overall (broadcast) surface cover will reduce the surface area to volume ratio of the applied slurry, reducing the area from which emission can occur. Band spreading will result in a thicker layer of slurry on the soil surface, for a given volume of slurry applied, thereby increasing the hydraulic loading rate per unit area, which may impede the infiltration of slurry into the soil. Some combinations of slurry composition (usually high dry matter content) and soil and climatic conditions can result in the slurry bands remaining on the soil surface and continuing to emit ammonia for an extended period of time. When applied to taller crops, slurry will be delivered below the canopy, which will reduce air movement and temperatures at the emitting surface, thereby reducing emissions. Applications to a shorter crop will not benefit from this effect and coating of crop leaves with slurry may also occur. Reduction efficiencies can vary greatly, but mean emission reduction is estimated at *c*.30% compared with surface broadcast slurry.

With trailing shoe, slurry is placed in narrow bands onto the soil surface beneath the grass canopy. The presence of a canopy results in a greatly reduced air movement and temperature at the soil surface, so emissions from slurry placed there will be lower than for slurry placed onto the crop canopy. The taller and denser the canopy, the greater this effect will be. Mean emission reduction is estimated at *c.*30% for applications to short grass and up to 60% for taller grass, compared with surface broadcast slurry.

Injection of cattle slurry will predominantly be shallow injection to grassland, whereby slurry is placed into the soil in shallow slots (5-10 cm deep) which are cut by preceding discs and will generally remain open. Shallow injection is suitable for grassland where field slopes or stoniness are not limiting (estimated to rule out approx. 30% of agricultural land) and for arable land prior to crop establishment. Mean emission reduction is estimated to be *c.*70%, compared with surface broadcast slurry.

Reduction in emission achieved through rapid soil incorporation of manure on arable land will depend on the method and timing of incorporation (Webb et al., 2006). Incorporation of FYM by tine cultivation 24 hours after application will only give 15% reduction in emission compared with FYM left on the surface, whereas immediate incorporation of slurry or FYM by ploughing will give 90% reduction.

All of these mitigation methods have the potential to increase emissions of nitrous oxide or losses via nitrate leaching, but generally, if applied at an agronomically suitable time (e.g. in spring) and rate, the risk of pollution swapping will be minimal.

For applications of FYM to grassland, there are no realistic methods for reducing ammonia emissions. However, storing of FYM (for a period >3 months) removed from the cattle house prior to application does offer a potential alternative. This method relies on reducing the quantity of readily available nitrogen in the FYM at the time of spreading and thereby reducing the potential for ammonia loss at that stage, such that emissions from storage and spreading are less than that from the spreading of ‘fresh’ FYM. During open-air storage of the FYM, there will be losses via ammonia volatilisation, but readily available nitrogen will also be immobilised in straw and lost via denitrification (following a nitrification stage), the products of which are the gases nitrous oxide, nitric oxide and dinitrogen (the ratio in which these gases are produced will depend upon conditions in the FYM heap).

#### 3.3.3.3.5 Mitigating emissions from fertiliser application to land

Ammonia emissions from nitrogen fertiliser applications can be reduced by using less fertiliser, by substituting urea-based fertilisers with other N types or by the incorporation of urease inhibitors within urea-based fertilisers.

Robust recommendation systems (e.g. MANNER, RB209) can be used to provide a good estimate of the amount of nitrogen (and other nutrients) supplied by manure applications. This information can then be used to determine the amount and the ideal timing of additional fertiliser N required by the crop. Fertiliser use statistics suggest that, in most cases, this will result in a reduction in fertiliser N inputs (particularly on arable crops) compared with current practice and a concomitant reduction in ammonia emissions from fertiliser N use (and particularly from urea-based fertilisers).

Urea and urea-based fertilisers are associated with a much greater ammonia emission factor than other nitrogen fertiliser types. Replacement of urea-based fertilisers with another type will therefore result in significant reductions in emission. Alternatively, the incorporation of a urease inhibitor with urea-based fertilisers, which will delay the hydrolysis of urea to ammonium carbonate allowing time for the fertiliser to become incorporated into the soil matrix, will reduce emissions by approximately 50% (Defra project NT2605).

Table 15. Potential methods for mitigating ammonia emissions from the dairy sector.

| Method | Applicability (%) | Current implementation (%) | Reduction efficiency‡ (%) | Potential reduction in sector (kt $NH_3$) |
|---|---|---|---|---|
| Use slurry band spreading application | 100 | 3 | 30-60 | 13.3 |
| Convert slurry systems to straw-based | 100 | 34 | 25-30 | 11.7 |
| Use slurry injection application | 70 | 1 | 70 | 10.1 |
| Wash dairy cow collecting yards | 100 | <5 | 90 | 5.0 |
| Rapid soil incorporation of manure | 50-90 | <10 | 60-85 | 1.9 |
| Increased scraping in cubicle house | 100 | <10 | 20 | 1.5 |
| Store all FYM for >3 months | 100 | 70 | 30 | 1.0 |
| Allow slurry stores to crust | 100 | 80 | 50 | 0.8 |
| Cover FYM heaps with sheeting | 100 | 0 | 15-90 | 0.2 |
| Extend the grazing period | 50? | 40? | 1-2 | <1 |
| | | | | |
| *Less certain methods:* | | | | |
| Reduce dietary N intake | 100 | 0 | 20? | 11.0 |
| Increased, targeted bedding | 100 | 0? | 50 | 1.5 |
| Out-winter on woodchip pads | 100 | <1 | ? | ? |

‡Compared to 'standard' practice.

Table 15 lists the potential methods for mitigation of ammonia emissions. Reduced emission slurry spreading techniques give some of the maximum reductions of any single method for the dairy sector and, combined with rapid incorporation of manure applied to arable land, represent the most sensible first options in trying to achieve reductions from this sector. Almost all dairy cattle housing is naturally ventilated, with limited options for reducing emissions. The conversion of slurry-based to straw-based systems presents one option, with potentially large reductions in emission, but would require significant structural changes to management systems and a greatly increased demand for straw bedding (which may not be able to be met), making this a less viable option. Out-wintering of stock on woodchip corrals may offer an alternative housing reduction option and ongoing research will yield more quantitative information for this option in the near future. Manure storage options are also limited; with most existing slurry stores developing a crust the potential for further reduction (by crusting or covering) is small. Storing of FYM as opposed to spreading directly from the house, combined with covering of the heaps offers some potential reductions although further work is required regarding the practicalities of covering manure heaps. Dietary N reduction can potentially give large reductions in emission and ongoing research aims to provide potential solutions for uptake, although the difficulties of achieving this with largely forage-based diets are not to be underestimated.

### 3.3.3.4 Models of ammonia emissions

There are a number of models for predicting ammonia emissions either from components of the agricultural system, at the whole farm scale or at the national scale. The most relevant of these are presented here.

#### 3.3.3.4.1 Single component models

A number of models have been developed for predicting ammonia emissions following application of manure to land. A mechanistic approach was taken by Genermont and Cellier (1997), who developed a model that simulates the influence of the various factors on volatilization, accounting for the transfers and equilibria in the topsoil and between the soil and the atmosphere. The model uses readily available input data, including soil, meteorological and slurry data. It includes energy balance and advection submodels, which make it suitable for field scale applications using simple meteorological data. Sensitivity analysis showed that soil pH has a large influence on volatilization. The model is also sensitive to soil adsorption capacity and some hydraulic characteristics (saturation water conductivity, water content at field capacity). A number of simpler, statistical models have been developed from experimental data (e.g. Menzi et al., 1998; Misselbrook et al., 2005b) and a decision support tool for farmers/advisers using this approach for UK conditions has been developed under Defra project KT0106 (MANNER_PSM). Data from several European countries were pooled and a statistical approach taken to produce the ALFAM model (Sogaard et al., 2002). For emissions from fertiliser applications to land, Misselbrook et al. (2004) developed a simple process-based model which has been further modified using experimental data from Defra project NT2605 for inclusion in the national inventory model.

For cattle housing (with slatted floor), a detailed mechanistic model was developed by Monteny et al. (1998). Components in the floor module describe urine deposition, enzymatic conversion of urea, dissociation of ammonia, and ammonia volatilization. Slurry pit module components describe urine and faeces production, and ammonia dissociation, and volatilization. A sensitivity analysis on the input parameters for which measured data were unavailable showed that urease activity had little effect on the simulated emission and that the largest effect was due to pH. The reliability of the model for predicting ammonia emission for practical conditions will mainly depend on the validity of assumptions concerning the time independent character of most of the parameters and on the accuracy with which parameters can be determined.

#### 3.3.3.4.2 Farm-scale models

Farm-scale models developed specifically to predict ammonia emissions at the farm scale level include MAST (Ross et al., 2002) and for dairy farms specifically (in the United States) the model of Pinder et al. (2004). The Pinder et al. model tracks the volume of manure and mass of ammoniacal nitrogen as the manure moves through the housing, storage, application, and grazing stages of a dairy farm. Most of the processes of ammonia volatilization are modelled explicitly, but poorly understood processes are parameterized and tuned to match empirical data. The model predicts monthly ammonia emission factors based on farming practices and climate conditions, including temperature, wind speed, and precipitation. The model can also be used to predict the seasonal and geographic variations in ammonia emission factors. MAST was based entirely on UK-derived emission factors, but many of these require updating to align with current knowledge. Hutchings et al. (1996) developed a dynamic model to predict the ammonia volatilization from grazing livestock farms and to allow potential control measures to be evaluated. The relationships within the model were based on the underlying physical and chemical processes, but empirically based factors were used to reduce the demand for input data and where the understanding of the underlying processes was inadequate, on a daily basis, the model simulates the partitioning of dietary nitrogen into dung and urine and its subsequent fate within the pasture or the slurry handling system. The fate of dry matter and water added in dung, urine and from other sources is also predicted. The model illustrates the indirect interactions between ammonia sources, highlights the influence of slurry management on ammonia losses, stresses the need for integrated, whole farm measurements and demonstrates that assessments of the impact of control measures may be misleading unless considered at the scale of the whole farm.

Dairy farm models which include ammonia emissions in addition to other production and pollutant outputs include $SIMS_{DAIRY}$ (del Prado et al., 2006), developed for UK dairy conditions. $SIMS_{DAIRY}$ integrates existing models for nitrogen (ammonia volatilisation, nitrate leaching, nitrous oxide and $NO_x$ emissions) and phosphorus, equations to predict methane losses and the cows' nutrient requirements, 'score matrices' for measuring attributes of biodiversity, landscape, product quality, soil quality and animal welfare and an economic model. $SIMS_{DAIRY}$ is capable of optimising dairy management factors in order to find more sustainable systems.

#### 3.3.3.4.3 National scale models

The inventory of ammonia emissions from UK agriculture is compiled using the NARSES model (Webb and Misselbrook, 2004). NARSES uses a mass balance approach to model the flow of nitrogen through the livestock and manure management system, with ammonia losses at each stage modelled as a proportion of the total ammoniacal nitrogen present within that stage. Other nitrogen losses (via leaching or denitrification) and transformations (immobilisation, mineralisation) during livestock housing and manure storage are included. NARSES makes use of the MANNER_PSM and NT26 models to derive national emission factors for manure and fertiliser applications to land, respectively. NARSES has been validated against a number of other well developed national ammonia inventory models for both slurry-based (Reidy et al., 2008) and solid manure (Reidy et al., in press) management systems. The model structure and parameters are reviewed annually as part of the UK ammonia emission inventory compilation process.

NARSES can be used as a farm-scale model, or incorporated into more holistic farm-scale models (such as $SIMS_{DAIRY}$).

### 3.3.3.5 Knowledge gaps

Significant advances have been made in recent years in improving our understanding and ability to model ammonia emissions from agricultural sources. However, a number of knowledge gaps remain, requiring further research effort:

- Despite a large database of experimental measurements, predictions of ammonia emissions following slurry application to land are still associated with high uncertainty. Much of this is due to difficulties in predicting the rate and extent to which slurries will infiltrate into the soil matrix.
- Nitrogen losses and transformations within solid manure management systems, particularly losses of nitrogen as dinitrogen through denitrification.
- A robust emission factor for slurry lagoons, expressed as a percentage of ammoniacal nitrogen available in the lagoon. The current EF is a factor of ten greater than that for above ground tanks, but based on very few measurements.
- Farm management activity data (although this has improved with the annual Farm Practices Survey since 2004), in particular data on cattle diet and nitrogen intake.

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3.3.4 Carbon Dioxide

### 3.3.4.1 Introduction

In the UK Greenhouse Gas Emissions Inventory (UKGHGI), (NAEI 2008), the energy sector is responsible for 86.1% of $CO_2$e emissions, industrial processes 4.1%, agriculture 6.75%, land use and land use change and forestry (LULUCF) -0.3% and waste 3.37%.

Generally, activity produces emissions of the three main greenhouse gases, carbon dioxide, methane and nitrous oxide. However, land use change, that is, from one state to another can result in emissions or reductions of greenhouse gases – see carbon sequestration section. The negative figure for LULUCF is due to the amount of carbon dioxide being absorbed from the atmosphere into crop growth in forestry and grassland and the soil micro-flora being slightly greater than that being released by cultivation of soils and the subsequent oxidation of stored carbon. This puts into context the place of agriculture in terms of the national picture.

UK agricultural emissions appear in three Categories (NAEI 2008). Agricultural emissions in Category 1 comprise emissions from energy use, which are 4.28 Mt of $CO_2$e. This is further broken down into 3.89 Mt liquid fuels, 0.012 Mt solid fuels and 0.372 Mt gas fuel, with no further breakdown between sectors of agriculture.

Category 4, Agriculture, covers production for which there are no carbon dioxide emissions cited. This is because carbon dioxide absorbed into crops, livestock and livestock products is treated as being released on consumption.

Category 5, Land Use, Land Use Change and Forestry is responsible for 1.99 Mt removal of $CO_2$e. However, this is the net effect of significant sources (emissions) and sinks (removals). The losses and gains from these areas can be separated as follows:

Table 16. Losses and gains of kt $CO_2$e in AFLULUC.

| Land Use, Land-Use Change and Forestry | $CO_2$ net | $CO_2$ sinks | $CO_2$ sources |
|---|---|---|---|
| Forest Land | -15,111.54 | -15,111.54 | |
| Cropland | 15,279.27 | | 15,279.27 |
| Grassland | -7,985.26 | -7,985.26 | |
| Wetlands | | | |
| Settlements | 6,218.63 | | 6,219.63 |
| Other Land | | | |
| Other | -395.86 | -395.86 | |
| | **-1,995** | **-23,492** | **21,499** |

Source, NAEI (2008).

The figures demonstrate that even though there are net gains in terms of $CO_2$, there are significant gains and almost equal emissions. The former are from carbon sequestration principally in forests and grassland and the latter, emissions, arising from cultivation of arable land. See section 5, Carbon storage in dairy farming systems.

Table 17. $CO_2$ emissions (kt) from all sectors.

| Categories | $CO_2$ (kt) |
|---|---|
| 1. Energy | 543,427.64 |
| 2. Industrial Processes | 13,986.47 |
| 3. Solvent and Other Product Use | |
| 4. Agriculture | |
| 5. Land Use, Land-Use Change and Forestry | -1,994.75 |
| 6. Waste (incineration) | 441.29 |
| 7. Other | |
| Total Net Emissions | **555,860.65** |

Source, NAEI 2008.

### 3.3.4.2 Carbon dioxide and dairying

The above figures show that for agriculture and for dairying in particular, the emissions of carbon dioxide are generated firstly in the energy sector in the form of electricity supplied to farms, by the use of fuel for farming operations and by land use change. In the case of the latter, this refers to the establishment or ploughing up of grassland, where the former becomes a sink of carbon dioxide and the latter causes that sink to release carbon dioxide.

These emissions could be reduced by improving the efficiency of energy use on dairy farms, avoiding plough cultivation and reducing fuel use where possible and avoiding very short term temporary grassland.

In their report on Greenhouse Gas Inventories for England, Wales, Scotland and Northern Ireland, AEA state that GHG emissions from agriculture comprise entirely methane, ($CH_4$) and nitrous oxide ($N_2O$) (AEA 2007). The estimates are consistent with the United Nations Framework Convention on Climate Change (UNFCCC) reporting guidelines and Choudrie et al. (2008). Whilst no carbon dioxide emissions are reported for the industry due to their small scale compared with major emitters, details of all contributing sectors are in fact recorded. Baseline energy-$CO_2$ emissions from all sectors of agriculture and horticulture have been estimated using UK Greenhouse Gas Inventory (GHGI) figures and for dairying they are 3,638 kg $CO_2$e per cow stocked at 1cow/ha plus followers (Defra 2006a).

Defra project ISO0205, (Defra 2006b) reported a total of 25,200 MJ energy use per 10,000 litres of milk produced, which translates into 1,184 kg $CO_2$e per cow producing 6,500 litres. However, taking the figures from ISO0205 and using them in the same calculation as that used in WT0706, produces an output of 3,120 kg $CO_2$e per cow producing 6,500 litres, a difference from WT0706 of -14.2%. On a per hectare basis, the output is 6,239 kg $CO_2$e/ha at a stocking rate of 2 cows per hectare.

The difference between WT0607 and ISO0205 appears to be because in ISO0205 a lower energy figure for concentrate feed was used, which was only one third of the value for electricity consumed and no figure appears to be used for the value of milk fed to the replacement calf.

The mean dairy farm $CO_2$ emissions shown by Natural England (2008) using the CALM tool is 1,250 kg/ha, but no equivalent per cow is provided.

A further source of $CO_2$ emissions is that of agricultural liming, for which UKGHGI figures state that 0.31Mt $CO_2$e are produced from liming applications to grassland (NAEI 2008). Clearly, there may be an opportunity to reduce $CO_2$ emissions by soil analysis and carefully targeting the use of agricultural lime.

#### 3.3.4.3 The Milk Roadmap

The Milk Roadmap presents the following targets for the dairy sector in England looking ahead to 2020 (Dairy Supply Chain Forum, 2008):

- reduce the GHG emissions balance by 20-30%,
- use 40% of energy from renewable sources, and
- recycle/recover 70% of non-natural on-farm waste.

Table 18. Long Term Actions by 2020.

| Target | Current Level | How? | Measure | Limitation |
|---|---|---|---|---|
| 20 - 30% reduction in GHG (carbon equivalents including $CO_2$, $CH_4$, $N_2O$) balance from dairy farms between 1990 and 2020. | Methane emissions have fallen by 13.5% since 1990. Carbon emissions have fallen by 23% since 2000 through improved efficiencies at the point of Nitrogen fertiliser production. | New feed technologies, improved yields, greater longevity to reduce culling rates and breeding of replacements. | Currently no industry agreed calculation for GHGs. This figure builds on current performance and is in line with international Kyoto targets. | Currently, only those mitigation methods that involve a reduction in number of animals register as a reduction on the enteric fermentation inventory. A more inclusive and comprehensive inventory calculation |
| 70% of non-natural waste is recycled or recovered as standard practice | Aspirational target subject to change following publication of Producer Responsibility targets and following a review of progress before 2020. | Through compliance with Agricultural Waste Regs and enabled by the introduction of Producer Responsibility Scheme | Compliance with PR scheme and Waste Regulations – Defra monitored | Lack of suitable recycling infrastructure and certain types of material (e.g. fertiliser and feed/seed bags, veterinary medicines and animal health products, baler twine and net wrap) are not recyclable and have to be sent to land fill. |
| 40% of energy used on dairy farms is from renewable sources | This is in line with the Government's commitment to generate 11-15% of the UK's energy from renewable sources, as a contribution to an EU target of 20%. | Generation of energy on farm through Combined Heat and Power installations, Anaerobic Digesters. | Government performance targets on Renewable Energy and NFU member information | |

Sources of greenhouse gas emissions identified by Garnett (2007) are as follows:

Table 19. Livestock Sector and Greenhouse Gas Emissions: Quantifying the Impacts.

| Life stage | Process creating emission | Type of emission |
|---|---|---|
| 1 Production of animal feed; silage production; grassland maintenance | Production of nitrate and other fertilisers; agricultural machinery; pesticides and other inputs | $CO_2$; $N_2O$ emissions from grazing land, fodder crops and fertiliser production |
| 2 Animal housing and maintenance, associated machinery | Heating, lighting, milking, etc | $CO_2$ |
| 3 Digestion (ruminants) | Enteric fermentation | $CH_4$ |
| 4 Waste products | Manure and urine | $N_2O$ and $CH_4$ |
| 5 Slaughtering, processing, waste treatment | Machinery, cooking, cooling, lighting, leather and wool production, rendering, incineration | $CO_2$ and refrigerant emissions |
| 6 Transport, storage | Transport, cooling, lighting | $CO_2$ and refrigerant emissions |
| 7 Domestic consumption | Refrigeration and cooking | $CO_2$ and refrigerant emissions |
| 8 Waste disposal | Transport | $CO_2$ and $CH_4$ |

Casey and Holden (2005) estimate on-farm diesel and electricity use to account for 5% of the total $CO_2e$ generated by an average Irish dairy farm. Schils et al. (2005) estimate a very similar 4-5% for a Dutch dairy farm. For pig and poultry systems the relative importance of on farm fuel use will be higher (since methane emissions are lower) although overall fossil fuel use per kg of product is still lower than it is for cattle systems.

Garnett states that farm energy use is not explored any further, since the contribution to Dairy production GHGs is so low.

#### 3.3.4.4 Conclusions and implications for dairy farming

Carbon dioxide emissions from dairying are significant in terms of overall UK emissions. The major source is in terms of land use operations, that is, cultivations involved in cropping and grassland production. Whilst other emissions of carbon dioxide are small, such as those coming from electricity consumption and motor and tractor fuel, all emissions should be treated as significant and measures should be taken to reduce them where possible.

The Milk Roadmap lists a number of objectives to reduce emissions, but others would include improving the efficiency of energy use on dairy farms, avoiding plough cultivation and reducing fuel use where possible and avoiding very short term temporary grassland.

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### 3.3.5 Carbon storage in dairy farming systems

#### 3.3.5.1 Soil carbon storage in grassland systems

There is growing emphasis being placed on soil carbon (C) storage in the mitigation of climate change and various measures are being explored to determine how best soil organic C storage (SOC) levels can be increased (e.g. Defra, 2008). Knowledge of current soil C stocks and ranges in various land-use/soil types is therefore important in order to determine the effect of management practice, land-use and climate on the magnitude and direction of any future changes in SOC. English topsoils (0-15cm) contain a total of 1015 Tg C (Bradley et al*.,* 2005), with *c.*43% in managed grassland soils, *c.*35% in arable soils and *c.*13% in semi-natural grassland soils (with the remainder in woodland, urban soils etc). Scotland, Northern Ireland and Wales have a greater proportion of semi-natural grasslands on peaty and organic soils, so on a UK basis (2543 Tg total soil C) the majority of soil C stocks are in semi-natural grassland soils (37%), followed closely by managed grasslands (35%), with arable soils containing 18% of total UK soil C stocks.

Data from the National Soils Inventory (NSI) indicates that grassland topsoils typically contain 4.2% SOC (1995/96 data), compares with 2.8% in arable/ley soils (Webb et al*.,* 2001), with these levels varying in relation to management, clay content and precipitation (Verheijen et al., 2005). Moreover, there is some evidence that soils in the UK may be losing C, probably as a consequence of land-use change, particularly the drainage of peat soils and a legacy of ploughing out grasslands, and that this could have implications for climate change. For example, Bellamy et al*.* (2005) used data from the NSI collected between 1978 and 2003, which indicated that carbon was being lost from soils across England and Wales at a mean rate of 0.6%/yr (or 13 million tonnes/yr). The relative rate of carbon loss increased with SOC content, being more than 2%/yr in soils with SOC contents greater than 10% irrespective of land use. Also, Webb et al*.* (2001) using NSI data showed that the SOC content of arable/ley and grassland soils in England and Wales had decreased between 1980 and 1995, especially where soils were ploughed out of grassland and on lowland organic and peaty soils in tillage cropping. The mean SOC content of soils in arable/ley cultivation in 1980 was 3.4% compared with 2.8% in 1995, a loss rate equivalent to 0.04%/yr. Similarly, for permanent (managed) grasslands SOC decreased from 5.0% in 1980 to 4.2% in 1985 (a loss of 0.05%/yr). In contrast, data from the Countryside Survey 2000 for a range of semi-natural and agricultural soils suggested that there had been an overall increase (or at least no change) in SOC between 1978 and 2000 (Black et al., 2002).

#### 3.3.5.2 Land use change and its impact on soil carbon storage

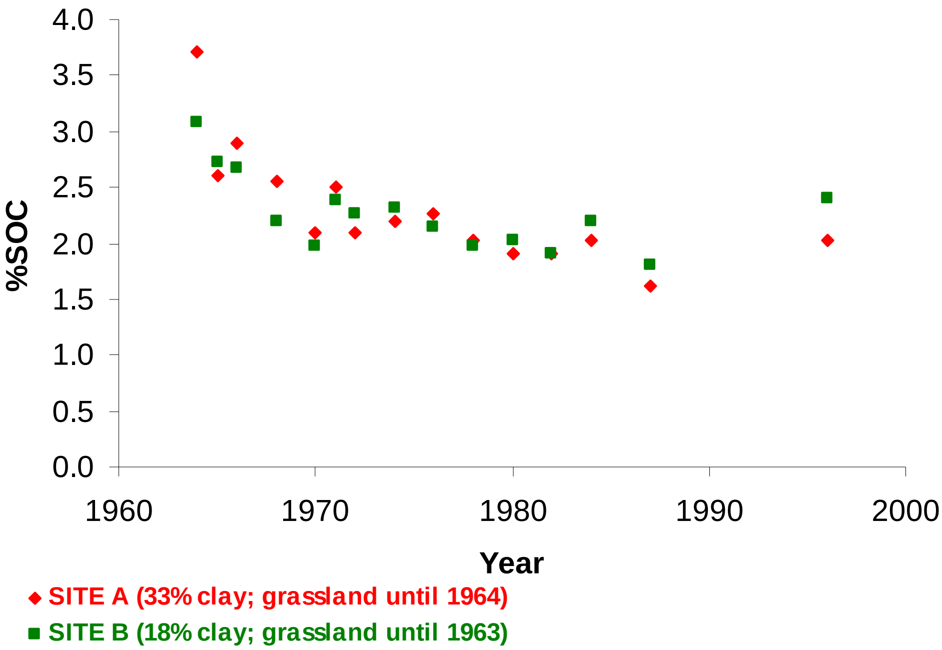
In a comprehensive review of carbon loss from soil and its fate in the environment, Dawson and Smith (2006) provided estimates of the potential gains/losses of soil carbon for a range of land-use changes under temperate conditions. These have been summarised in Table 20 for those land-use changes that are pertinent to dairy farming systems in the UK. It should be noted that there is a high degree of uncertainty associated with these data, either due to lack of relevant studies (e.g. grassland to forest) or variations caused by contrasting management regimes on the same land-use type, particularly tillage land and grassland (Soussana, et al*.,* 2004).

Table 20. Potential soil C gains (+ve) or losses (-ve) resulting from land-use change (adapted from Dawson and Smith, 2006).

| Land-use change | Net C rate of change ($t\ CO_2e/ha/yr$) | Reference |
|---|---|---|
| Grassland to tillage | -3.7 to -6.2 | Smith et al., 1996; Guo and Gifford, 2002; Murty et al., 2002 |
| Tillage to grassland | 1.1 to 7.0 | IPCC, 2000; Vleeshouwers and Verhagen, 2002; Guo and Gifford, 2002; Murty et al., 2002 |
| Permanent crops to tillage | -2.2 to -6.2 | Smith et al., 1996; Guo and Gifford, 2002; Murty et al., 2002 |
| Grassland to forest | 0.4 | Sousssana et al., 2004 |
| Tillage to forest | 1.1 to 2.3 | Smith et al., 2000; Falloon et al., 2004; Guo and Gifford, 2002; Murty et al., 2002 |

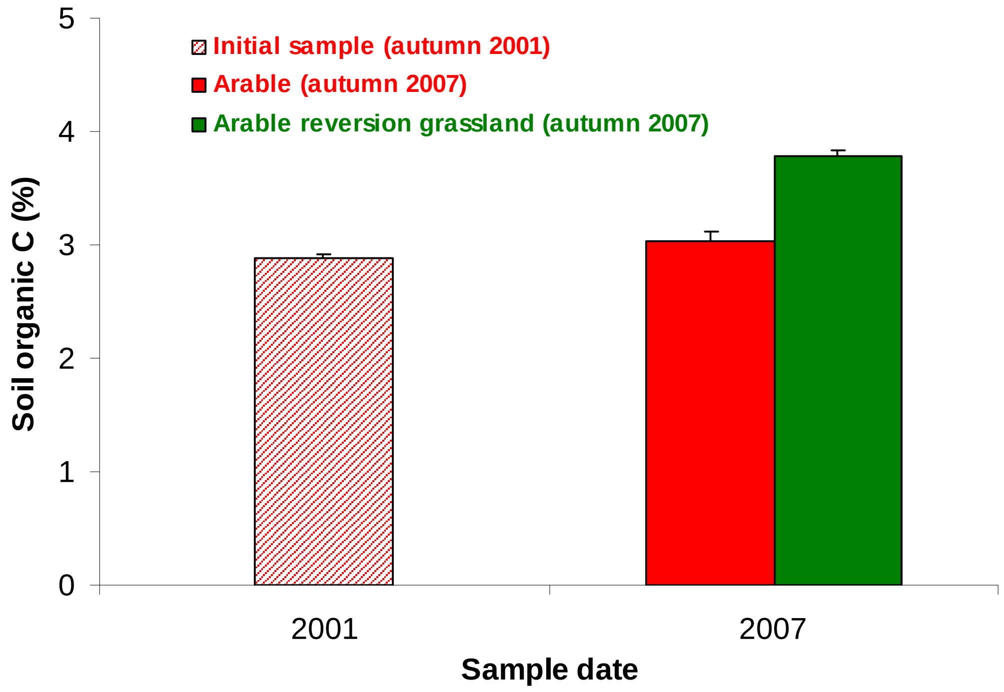
The conversion of grassland or permanent cropping to tillage cropping was estimated to result in C losses in the range 2.2 to 6.2 $tCO_2e/ha/year$ (Dawson and Smith, 2006). These losses were largely due to vegetation clearance, increased soil organic matter decomposition rates upon cultivation and losses of C through erosion (Freibauer et al., 2004). For example, Figure 8 clearly demonstrates SOC loss as a result of ploughing out grassland for tillage cropping at two sites on silty soils in Lincolnshire (Garwood et al., 1998). Here, SOC (0-15 cm) declined by 45% (at site A: 33% clay) and 22% (at site B: 18% clay) over *c.*30 years following ploughing-out. This was equivalent to 33 t C/ha and 13 t C/ha (i.e. 3.8 and 1.5 $tCO_2e/ha/year$), respectively.

Figure 8. Decline in topsoil (0-15 cm) organic carbon following the ploughing-out of long-term grassland in Lincolnshire, UK (Garwood et al., 1998).




In contrast, the conversion of tillage land to grassland can result in increased SOC storage in the range 1.1 to 7.0 $tCO_2e/ha/year$ (Dawson and Smith, 2006). For example, recent results from a medium-term arable reversion experiment at Faringdon in Oxfordshire (Williams et al., 2008) showed a 24% increase in SOC (0-15 cm) after 6 years of arable reversion to grassland (Figure 9). The increase in SOC storage was equivalent to 15 t C/ha (i.e. 9.2 $tCO_2e$/ha/year).

Figure 9. Changes in SOC on arable reversion grassland plots at Faringdon (Oxfordshire) between 2001 and 2007.




With all estimates of potential changes in SOC storage, it must be recognised that annual rates of accumulation (or depletion) change over time. After a change of management or land use, SOC content tends to move towards a new equilibrium value (after 100 years or more) that is characteristic of the soil type, land use and climate (Johnston and Poulton, 2005). Consequently, annual rates of SOC accumulation (or depletion) change over time and gradually decline as the new equilibrium is approached, when they become zero. The greatest rate of increase/decrease will therefore occur in the early years following change, with *c.*50% of the long-term (*c.*100 years) SOC accumulation occurring in the first 20 years (Johnston and Poulton, 2005). A result of this 'diminishing return' situation is that, whatever annual rate of SOC increase/decrease is estimated, it must not be assumed that it will continue indefinitely. Eventually (e.g. after a period of *c.*100 years) the annual rate of SOC increase/decrease will be zero.

Soil carbon accumulation is also reversible, maintaining SOC at the new equilibrium level is then dependent on continuing the new management practice/land use indefinitely. Indeed, SOC is lost more rapidly than it accumulates (Freibauer et al., 2004). Only if land is taken *permanently* out of cultivation (i.e. to permanent grassland or woodland), will the benefits of soil C accumulation and storage be realised over the long-term. This obviously has implications for rotational cropping.

When considering the carbon storage potential of various agricultural practices, the overall effect of the practice on all greenhouse gases (i.e. $CO_2$, $N_2O$ and $CH_4$) needs to be considered. Changes in other greenhouse gas emissions (i.e. direct and indirect $N_2O$ and $CH_4$) following land-use change, have not always been extensively quantified, leading to further uncertainty.

The current UK Greenhouse Gas (GHG) Inventory (Choudrie et al*.,* 2008) provides estimates of GHG emissions from agriculture and land use change and forestry (LULUCF), based on IPCC 1996 Revised Guidelines and Good Practice Guidance (IPCC, 1997 a,b,c). For LULUCF, estimates are made of changes in soil and biomass C stocks resulting from changes in land-use between the following sectors: forestry, grassland, cropland and settlements. Over the period 1990 to 2006, the LULUCF sector in the UK was a net $CO_2$ sink (since 1999), with an estimated 1.99 MT $CO_2$e stored per annum in 2006 (Choudrie et al*.,* 2008).

The CALM carbon accounting calculator (Carbon Accounting for Land Managers; CLA, 2008) is a tool that uses the UK GHG inventory methodology to work out the balance of GHG emissions from a farming business and C stored in trees and soils. By estimating the GHG emissions and C storage that arise from a land management business, the tool can be used to identify areas where either emissions (from energy and fuel use, livestock, cultivation and land use change, or fertiliser and lime application) can be reduced, or C storage in soils and trees enhanced. Three possible *increased* C storage options are identified within the tool, viz: land use change (gain in soil C from conversion of tillage land to grassland or woodland, or grassland to woodland); farm woodland (gain in biomass C) and; commercial forestry (gain in biomass C). The potential for C storage through the creation of farm woodland is discussed in Section 5.3 below. Also, soil C storage *losses* are identified through land use change (e.g. conversion of grassland or woodland to tillage land, woodland to grassland and loss of land to development) and the drainage and cultivation of organic/peaty soils.

#### 3.3.5.3 Woodland carbon storage

One of the main mechanisms of increasing C storage within any farming system is to take land *permanently* out of food production by the creation of farm woodland. Such a land use change has been estimated to increase soil C storage in the range 1.1 to 2.3 t$CO_2$e/ha/year following the conversion of tillage land to forestry, with a lower estimate for grassland conversion (0.4 t$CO_2$e/ha/year) (Dawson and Smith, 2006). However, there will also be considerable gains in above ground biomass C (Smith et al., 2000), with estimates for C gains in woodland vegetation in Northern Europe ranging from 0.1 to 5.6 t$CO_2$e/ha/year (Liski et al., 2002). In a recent study (Defra, 2007), the potential increase in biomass C resulting from the planting of an extra 7,000 ha of woodland in the UK (Oak and Sitka at a ratio of 4:1, and producing 5 t/ha wood per year) was estimated to result in an additional 64 kt $CO_2$e/yr (9.2 t $CO_2$e/ha/yr). Over the following 20 years, this would amount to an additional 13 MT $CO_2$e, as the increase would be annually cumulative. This would be in addition to the existing UK forest biomass C storage estimated at 16 MT $CO_2$e (from 8,000ha forestry).

Smith et al*.* (2000, 2001) examined the non-$CO_2$ impacts of various carbon mitigation options, including re-forestation. It was acknowledged that, as would be intuitively expected, woodland would normally emit less $N_2O$ than tillage land. However, the literature showed that although $N_2O$-N emission rates (0.2-1 kg $N_2O$-N/ha/yr) from European forest and woodland sites (un-affected by high N deposition rates) were at the lower end of the range of estimates for fertilised cereal crops (0.2-3.7 kg $N_2O$-N/ha/yr), it was considered not to be appropriate to estimate the impact of tillage land to woodland reversion in terms of reduced net $N_2O$ emissions, because of the emission overlap. Smith and Conen, (2004) also reviewed the effect on $N_2O$ and $CH_4$ emissions of land use change from forestry or grassland to tillage agriculture. It was estimated that the effect of a change in land use from forestry or semi-natural grassland to tillage agriculture would reduce the $CH_4$ sink (median value of 1-2 kg $CH_4$/ha/yr) by 1.5 to 2-fold. They suggested that this reduction in the $CH_4$ sink was likely to be due to the land use change itself, rather than different management techniques following the change. It was also suggested that cultivating grassland or forest land in temperate environments would enhance $N_2O$ emissions due to the mineralisation of organic N, but for only a few months. The $N_2O$ loss was assessed to be comparable in size to that estimated using the IPCC default value for direct $N_2O$ emissions from agricultural soils i.e. 1.25 ± 1.0% of the total N mineralised (IPCC, 1997 a;b;c); or 1.0% using the 2006 guidelines (IPCC, 2006).

#### 3.3.5.4 Conclusions

In summary, the biggest opportunities to *increase* C storage within dairy farming systems (i.e. to change from the present day baseline) are through the:

- Conversion of tillage land (and to a lesser extent, grassland) to farm woodland (this action will be recognised within the IPCC inventory; IPCC, 2006).
- Conversion of tillage land into *permanent* grassland.

In contrast, situations that are likely to *decrease* C storage within dairy farming systems (i.e. to change from the present day baseline) include:

- Clearance of farm woodland and conversion to tillage land or grassland
- Cultivation of permanent grassland and conversion to tillage land.

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3.3.6 The structure of the UK dairy industry

### 3.3.6.1 Dairy farm and cow numbers

The number of dairy farmers has declined significantly in the past decade (Table 21). The greatest reduction occurred in England (51%) and the least in Scotland (29%). Defra (2004) predicted that there would be a further overall reduction in dairy farm numbers in the UK of 8% by 2010, but that numbers in Scotland and Northern Ireland would increase marginally.

Table 21. UK Dairy Farm Numbers (from June Census 1997 2006 2007).

| | 1997 | 2007 | % reduction 1997 - 2007 | 2010* |
|---|---|---|---|---|
| England | 21,664 | 10,577 | 51 | 9,175 |
| Wales | 4,446 | 2,290 | 48 | 1,581 |
| Scotland | 2,009 | 1,429 | 29 | 1,523 |
| Northern Ireland | 5,233 | 3,619 | 31 | 4,200 |
| UK | 33,352 | 17,915 | 46 | 16,479 |

Source: Dairy Statistics: An insider's guide 2008. DairyCo.
*Projected figures from "The Future of UK Dairy Farming" Report – Defra 2004.

In line with the reduction in dairy farms, there was a reduction in the number of dairy cows. Although the number of adult dairy cows in the UK remained steady for most of the latter part of the last century at around three million head, they have declined steadily in the last decade (Table 22). The greatest proportional reduction has been in England (26%) while dairy cow numbers increased slightly (2.5%) in the same period in Northern Ireland.

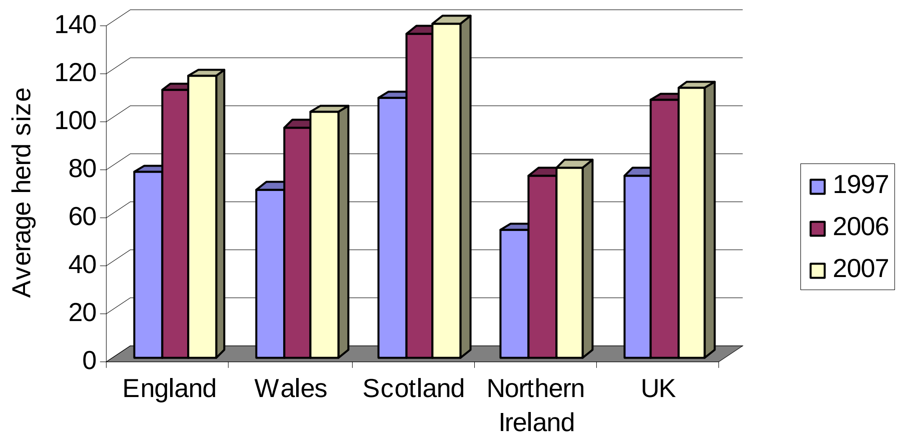
Table 22. UK dairy cow numbers ('000 head).

| | 1997 | 2007 | % reduction |
|---|---|---|---|
| England | 1,672 | 1,236 | 26 |
| Wales | 311 | 234 | 25 |
| Scotland | 217 | 198 | 9 |
| Northern Ireland | 279 | 286 | +2.5 |
| UK | 2,479 | 1,954 | 21 |

Source: Dairy Statistics: An insider's guide 2008. DairyCo.

As a result of this substantial restructuring of the dairy industry, the average herd size has increased in all of the countries of the UK in the past decade (Figure 1).

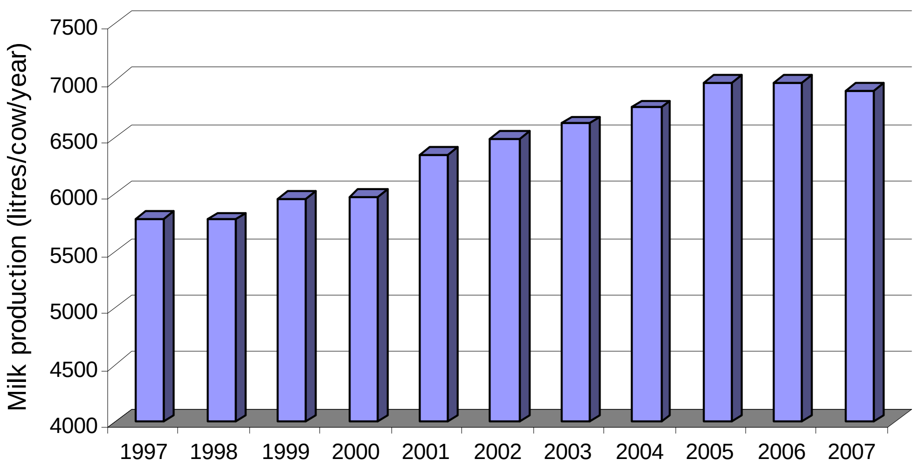
Figure 10. UK Average dairy herd size.




#### 3.3.6.2 Milk yields

In the last decade, milk production per cow has increased by 19%, with an average in 2007 of 6908 litres/cow (Figure 11). These improvements are the result of developments in breeding, nutrition and management. Pryce et al. (2004) estimated that the improvement in milk yield over the past 25 years has been 3-4%, of which up to 50% was directly attributed to improvements in genetic merit. Based on trends over the past 25 years, Garnsworthy and Thomas (2005) predicted that average UK yields would be 7236 litres/cow/year by 2010. Over the longer term these authors predict that, subject to any major market changes, average milk yields in the UK will increase to 9,000 litres/cow/year in 2030 and 10,760 litres/cow/year by 2050. Results for the past two years suggest that the rate of increase may have declined, although this may be too short a time-frame on which to base any firm predictions.

Figure 11. Average milk yield, litres/cow/annum[1].




As these figures demonstrate, there has been a considerable change in the structure of the UK dairy industry. As a result of changes both in total cow numbers and milk producers the number of dairy cows per holding has almost doubled over the past decade, and this trend is

[1] Figures are estimates made by MDC and DairyCo based on average dairy cow numbers and gross milk production excluding any suckled milk, as estimated by DEFRA.

expected to continue.

### 3.3.6.3 The output of GHG and ammonia on UK dairy farms

There is considerable variation between dairy farms in the factors that might be expected to influence GHG and ammonia output. This review examines some of the management, agronomic and nutritional factors that may influence GHG and ammonia emissions.

#### 3.3.6.3.1 Stocking rates

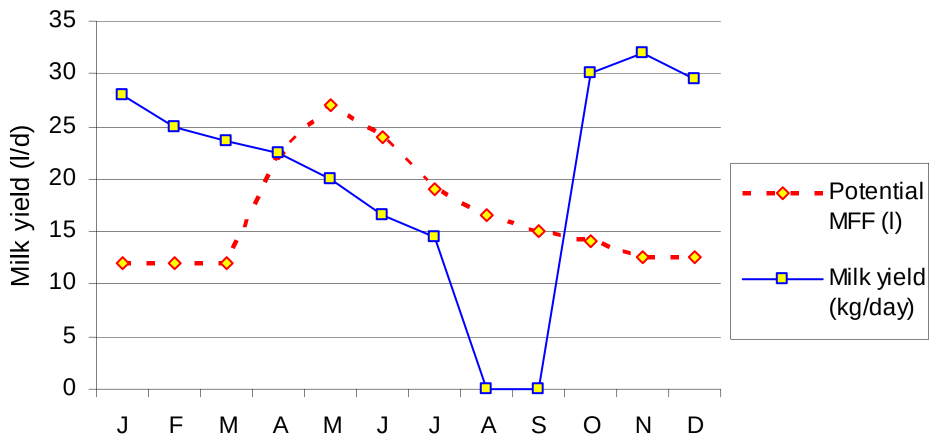
Stocking rates, which tend to be positively associated with overall farm profitability (Turner and Robbins, 2006), vary significantly between dairy farms in the UK. Nix (2009) has suggested that the average stocking rate is 2 cows per forage hectare. However, stocking rates have declined in recent years, largely in response to lower profitability, and in practice relatively few dairy farmers have stocking rates in excess of 2 cows/ha[2]. Furthermore, the 2007 Farmers Voice survey by Temple et al. (2007) reported that over the next five years specialist dairy farms anticipate reducing dairy cow numbers by 10%, largely as a result of CAP changes.

One of the consequences of reducing stocking rates, particularly where forage yield/ha is maintained, is an increase in the amount of forage available per cow. Where milk output is maintained, it follows that there will be more forage available per cow, and therefore less reliance on concentrate feeds. As discussed in section 3.3.1.3 in this review, changes to the relative proportions of forages and starch-rich feeds in dairy cow diets can influence levels of methane production.

#### 3.3.6.3.2 Seasonality

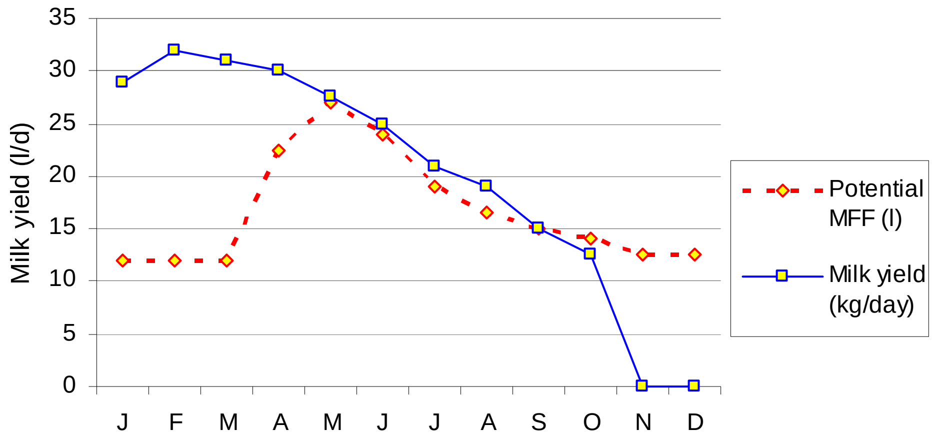
The date of calving can have an important bearing on milk production from forage (MFF) and particularly on milk from grass. As illustrated in Figure 12 and Figure 13, a 7000 litre cow calving in January (Figure 13), calving closer to the start of spring grass growth, utilises more of the potential production from grass than an autumn calving cow[3]. The lactation curve of a lower yielding cow, calving later in the spring, i.e. March, will more closely match the potential milk production from grass.

Figure 12. Potential milk from forage with the 7000 litre October calving cow.




[2] John Morgan, *personal communication*
[3] Source: DARDNI (http://www.ruralni.gov.uk/index/publications/information_booklets/high_forage/grass_utilisation.htm)

Figure 13. Potential milk from forage with the 7000 litre January calving cow.



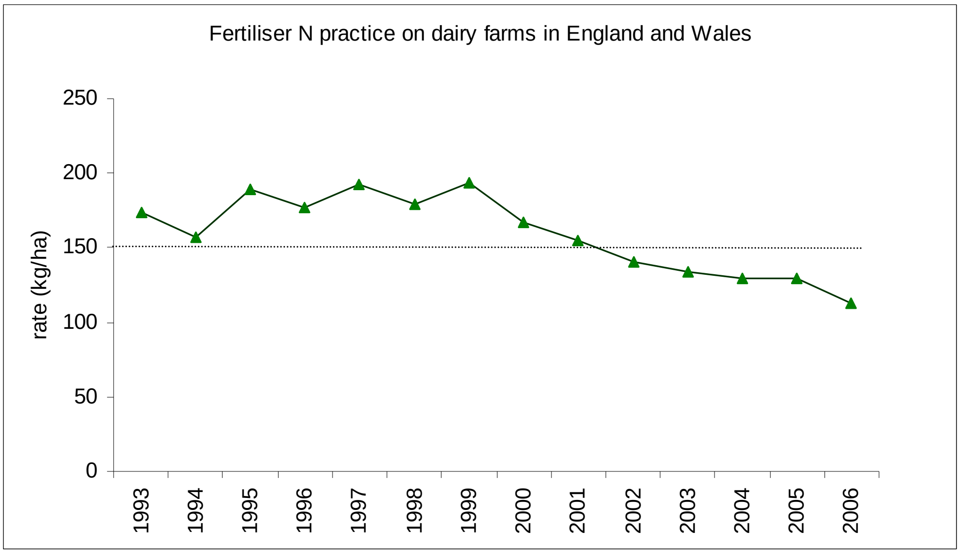

Fresh and conserved grasses vary considerably in their nutritional content, and in particular the amount and form of the protein. Since the form in which forage protein is supplied can have a significant impact on the protein N utilisation, and on both the amount and form of the N excreted, it follows that seasonality of milk production could have an effect on GHG and ammonia emissions by a dairy herd.

3.3.6.3.3 Fertiliser practice

The UK has a unique climate and land distribution which favours grass production, particularly in the west. As a result, grass both grazed and conserved, represents the main source of nutrients to support milk production by dairy cows. The high levels of forage required to support these increasing levels of production have been achieved from a combination of improvements in grass varieties, grassland management and the use of fertiliser.

In order to achieve maximum yields, grass requires a supply of nutrients, but it has long been recognised that nitrogen (N) is the most important in determining yield. Over the past five years there has been a steady reduction in the total fertiliser applied to grassland (Figure 14). Undoubtedly some of this decline reflects pressures on dairy farmers to reduce the unit cost of production, together with an increasing awareness of the contribution of nutrients from manures and hence a greater potential for savings in annual fertiliser inputs on the farm. The continuing decline in dairy cow numbers in the UK, resulting in lower stocking rates, will also have been a major factor.

Figure 14. Fertiliser N application (kg/ka) to grassland on UK dairy farms[4].




One of the effects of this reduction in N application has been a reduction in the N content of forages. In 1998, the average N content of first cut grass silage (1124 samples) was 28.5 g/kg DM; in 2007 it was 22.1 g/kg DM (1300 samples)[5]. Similar differences were observed for second cut and big bale silages. Data are not available on the N content of grazed grass, although it may be reasonable to assume that there has been a similar reduction in the N content of that as well.

As discussed elsewhere in this review, these differences would be expected to be reflected in both the total amount of N excreted and the relative proportions of N excreted in urine and faeces. A number of studies have demonstrated that as N intake increases, there is an increase in the proportion of urea N:total N excreted. The significance of this is discussed in more detail below, but in summary urea N is more prone to volatilisation than faecal N, and likely to make a greater contribution to ammonia and nitrous oxide emissions. In view of the reduction in the N content of conserved forages, it may not be unreasonable to assume that both total N and urine N output has declined as a result of reductions in fertiliser N application to grassland.

### 3.3.6.4 Feeds and feeding systems

As discussed elsewhere in this review, the production of GHGs and N excretion are closely linked to diets fed. While dry matter intake has an overriding effect on the amount of methane produced by dairy cows, this can be modified by the forage composition and the proportion of concentrates in the diet (Yates et al., 2000). N excreted by dairy cows is related to N intake and protein degradability (Kebreab et al., 2002), which in turn impacts on ammonia emissions, together with housing and manure handling systems (Misselbrook and Smith, 2002). Therefore, there is a potential to decrease atmospheric emissions by changes in diet formulation.

Milk producers in the UK use a wide range of feeding systems, certain characteristics of which are discussed below.

[4] Source: British Survey of Fertiliser Practice
[5] Source: ADAS laboratories (1998 data); Eurofins (2007 data)

#### 3.3.6.4.1 Types of forage

The UK has around 6.5 million hectares of grass, excluding rough grazing. Grass, mainly perennial ryegrass, is the main crop for both feeding and conservation. It is widely acknowledged that forage N is generally poorly utilised. On dairy farms, between 20 and 35% of the N consumed by the herd is retained in the protein of the milk and meat produced, with the remainder excreted in manure (van Vuuren and Meijs, 1987; Charmley et al., 1988; Tamminga, 1992; Dou et al., 1996). The low efficiency with which N is converted to milk is largely attributed to an imbalance in the supply of fermentable energy and degraded N, both of which are required by rumen micro-organisms for the synthesis of microbial protein. As a result, strategies aimed at improving nitrogen utilisation efficiency (NUE) have tended to focus on ways of reducing this imbalance. Two general approaches have been identified to improve N utilisation, namely the use of alternative forages and diet manipulation.

A number of studies have suggested that higher sugar contents in grass may decrease the asynchronous supply of N and fermentable energy for ruminal microbial protein synthesis, and hence reduce rumen N losses and increase efficiency of N use. In an attempt to increase the supply of fermentable energy, plant breeding programmes have been undertaken to develop ryegrass varieties that express consistently higher water soluble carbohydrate (WSC) levels than conventional varieties. The potential for such ‘high sugar grasses’ (HSG) to improve NUE appears to be high. New perennial ryegrass (PRG) varieties have been produced which accumulate on average 280g WSC/kg DM (range 150-370) which, when evaluated throughout two growing seasons, consistently yielded 3.6% units more WSC than conventional ryegrass species. Similar improvements in WSC content and yield have been noted for newer Hybrid ryegrass varieties (Wilkins and Lovatt, 2004). However, Tas et al. (2006) have reported that cultivars with an increased WSC content do not consistently result in a significantly increased microbial protein flow to the duodenum and efficiency of microbial protein synthesis, and did not result in an increased efficiency of N utilization in dairy cows. Clearly further research is needed in this area if they are to be reliably used as a means of improving NUE and reducing N excretion.

#### 3.3.6.4.2 Legumes

Legumes are generally regarded as being agriculturally beneficial due to their ability to ‘fix’ atmospheric nitrogen, rendering both the plant and the rhizosphere high in N. Indeed, white clover (WC) grass mixes have been shown to support similar levels of ruminant production (Davies and Hopkins, 1996) to that of grass fertilised with 200kg N/ha. The use of legumes therefore provides an opportunity of reducing use of synthetic fertiliser.

Forage legumes such as birdsfoot trefoil (*Lotus corniculatus*) and sainfoin (*Onobrychis vicifolia*) contain condensed tannins that bind to proteins, protecting them from rumen proteolysis. The tannin/protein complex dissociates post-ruminally, rendering the protein available for use by the animal. Thus, tannins in fresh *L. corniculatus* have been shown to reduce rumen ammonia by 40-50%, increase NAN flow to the small intestine by 40% (Waghorn et al., 1987), increase feed intake and milk yield in dairy cows, (Woodward and Reed, 1989) The beneficial effects of condensed tannins are only observed at low levels (10-55g tannin/kg DM) of dietary inclusion; higher levels can reduce both feed intake and diet digestibility (Barry and Duncan, 1984; Barry et al., 1986).

Although red clover does not contain condensed tannins, it contains a protein protectant, polyphenol oxidase (PPO), such that proteolysis of protein during ensilage was 7-40% compared with 45- 90% of that of ensiled Lucerne (Jones et al., 1995; Cavallarin, et al; 2002a). The efficiency of MPS *in vitro* was enhanced when RC silage (RCS) was mixed with grass silage (70:30) by 22% (Merry, et al., 2002). The NUE of RCS for milk has been shown to be higher than from Lucerne silage (Broderick, 2002) and from four forage legumes

(Bertilsson et al*.,* 2001), but lower than that of grass silage. Mixing RCS with grass silage raised the NUE for milk by 11-15% compared to when RCS was the sole forage (Bertilsson et al*.,* 2001), whilst maintaining the higher DMI and milk yields observed when RCS rather than grass silage was fed as the sole forage. Likewise, when compared with grass silage, mixtures of RCS and maize silage (25:75) resulted in increased DMI, improved milk yield (7.5 kg/d) with the NUE for milk being raised by 14%. This represented a 35 g N/d reduction in urinary N loss per animal, (Dewhurst et al., 2005) which, for the entire 2002 British dairy herd, is equivalent to a reduction of 18 677 t urinary N per annum. RC also contains formonetin, a phytoestrogen which has been associated with increased lamb LWG (Fraser, 2004; Moorby et al., 2004), but may reduce ewe fertility.

A number of other forage crops have also been considered as sources of 'bypass' protein. The British acreage of white lupins has increased to *ca.* 20,000 ha, as it is perceived that they are a valuable home-grown protein source. While most lupins currently being grown will be harvested for their seed, there has been some interest in growing lupins as a forage crop for ensiling, and subsequent use as a feed for ruminants. The feed quality of forage lupins is similar to that of peas (15% protein), but DM yields are higher, around 11 t/ha (Sheldrick, et al., 1980, 1985). Furthermore, protein from lupins is less rapidly degraded in the rumen than that from peas, and could form an effective accompaniment to grass silage or maize silage. There is also reduced soil accumulation of N associated with lupins compared with peas (Haynes et al*.,* 1993) and lupins 'scavenge' soil P, thus potentially reducing the polluting effects of excess levels of these elements.

Compared to grass, forage peas contain higher levels of N and starch, and milk yields by dairy cows fed pea/ wheat silage and 4 kg of concentrates were equivalent to those fed grass silage plus 8 kg concentrates. However, NUE was lower for the mixed crop silage (Adesogan et al., 2000). Forage peas are a short-lived crop enabling succeeding crops to be grown in the same season, and thus kale may follow peas for autumn and winter feeding, using the excess N in the system. Kale contains significant levels of N and WSC and is highly digestible (Young et al., 1997), but when ensiled is low in residual WSC and true protein (Fraser et al*.,* 1999). However, when ensiled as a bi-crop (kale: barley 20:80) and fed to dairy cows, Moorby et al., (2003) reported reduced urinary N outputs and greater milk N contents compared to when grass silage was fed.

There are a number of reasons why forage legumes are not more widely used than they are. These include difficulties of establishment, lack of persistency and the general fact that they do not readily fit into many dairy farming systems. It should also be noted that the perception that biologically fixed N has less environmental impact than that derived from synthetic fertilisers is unsound, as N losses are related to the total N input rather than its origin. The N pollution emanating from legume forages can be significant. Nitrogen leached from soil beneath stands of red clover (RC) or WC was slightly greater than that from grassland fertilised with 200kg N/ha (Scholefield et al*.,* 2001). When unfertilised grass/clover swards were compared with N-fertilised grass there were little differences in the quantities of leached nitrate (Cuttle et al., 1992). Therefore, the combination of high levels of N leaching from forage legumes and their low NUE by livestock suggests that they may not be suitable as the major portion of the forage ration for ruminant systems designed to maximise NUE and minimise the environmental burden. Their judicious admixture with high WSC/low N forages may be more appropriate in this regard. However, greater NUE can be obtained from legumes containing substances that reduce N degradation in the rumen.

#### 3.3.6.4.3 Conserved forages

Approximately one third of grassland area (excluding rough grazing) is used for silage production, with dairy farms the principle silage producers and users. Grass, mainly perennial ryegrass, is the main crop for conservation. While most silage comes from first cuts and is stored in clamps, the production of big-bale silage has become increasingly popular, and now accounts for 25–30% of all grass silage made.

The type and amount of forage fed can have a significant effect on protein intake and N excretion by cattle. Grass silage is normally very high in degradable protein; in contrast, maize silage is low in total protein, and this protein is generally less degradable. There is now considerable evidence that partial substitution of grass silage with maize silage results in a reduction in N excretion (Verité and Delaby, 2000; Kebreab et al., 2001). Whole-crop cereal silage is also increasingly used as winter forage for lactating dairy cows.

#### 3.3.6.4.4 Grass silage

On the majority of UK dairy farms, grass silage is the main, and frequently the only, conserved forage. There have been significant developments in recent years in both machinery and products for silage making, all aimed at minimising field and in-silo losses and improving the quality of the feed to maximise intake potential, although the extent to which this rate of development will continue is not clear.

Additives have had a major role in improving the silage quality, particularly under difficult silage making conditions. The use of additives has resulted in improved fermentation, reduced in-silo losses, greater aerobic stability at feeding and improved nutrient utilisation and animal production under a wide range of conditions. Such has been the success of silage additives in reducing ensiling losses and improving production that by the mid 1990s there were over 150 different products on the market.

The first additives used commercially in Europe were based on mixtures of sulphuric and hydrochloric acids, and for many years thereafter the use of organic acids or their salts were widely used. These had the effect of augmenting the acids produced as a result of natural fermentation of the grass sugars present in the grass. Significant improvements in silage additive applicators were made in the 1960's, and as a result a number of chemicals that had previously only been shown to be effective in small-scale trials were successfully introduced. The most widely used of these was formic acid, used either alone or in mixtures. Other products marketed as silage additives included mixtures of organic acids and formalin, other sterilising agents and sugars. The use of these additives increased substantially, and they were undoubtedly a key factor in the increase in silage making that took place in the 1970s and 1980s.

Since then, a plethora of biological additives have been developed. Bacterial inoculants were first imported into the UK in the early 1980's but they were relatively ineffective, probably due to inappropriate application rates or poor product quality. Today they represent the largest number of products currently on the UK register of silage additives, with the main active ingredient being *Lactobacillus plantarum*. Products tend to be differentiated by the strain of the bacteria; although all aim to ensure that at least 1 million viable (colony forming) bacteria are applied to each gram of fresh forage ensiled.

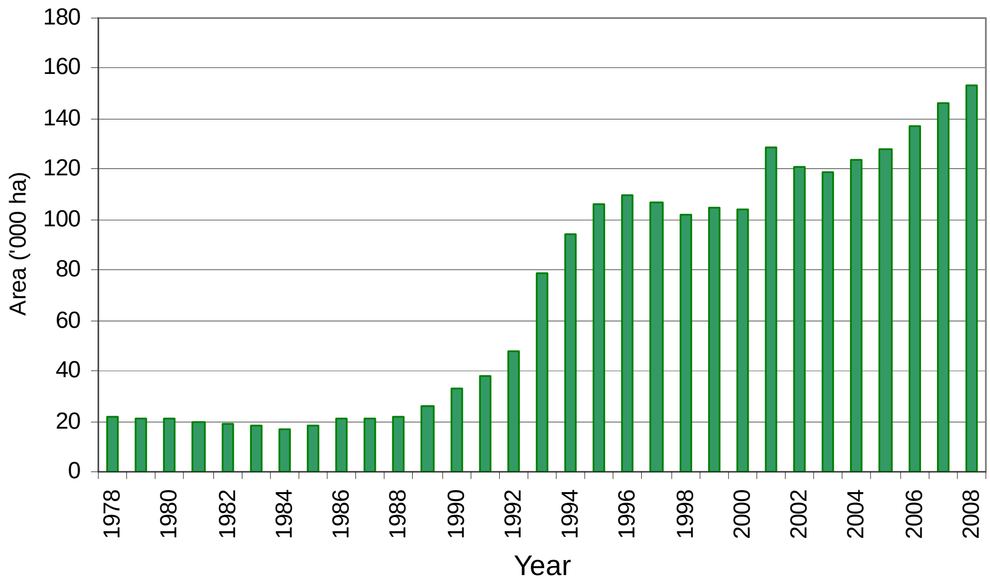
The primary benefits for dairy farmers from using additives are that they improve the efficiency of production by reducing waste and improving animal production. This improvement in animal performance may be associated with improved efficiency of microbial protein. Sharp et al. (1994) reported an improvement of 33% in the efficiency of microbial protein synthesis when silages made with a biological inoculant were compared with well preserved untreated silages.

Recently, EU legislation has been introduced which requires that all silage additives must undergo authorisation by October 2010[6]. This will apply both to products currently in use and any new products prior to being marketed[7]. The approval process, which will include a requirement to demonstrate safety and efficacy, will have significant cost implications for silage additive manufacturers, and it remains to be seen whether this legislation results in reductions in both the number of additives available to silage makers and development of new improved additives.

#### 3.3.6.4.5 Maize silage

Although grass silage remains the main farm-produced feed for winter feeding, alternative forages such as maize silage and whole-crop cereal silages have been increasingly used. Figure 15 illustrates how the area sown to forage maize has increased over the past 30 years. Although it is an excellent feed for beef cattle, most of the silage produced will have been used for dairy cows.

Figure 15. Area ('000 ha) sown to forage maize in the UK (Source: Agricultural Statistics, MAFF and Defra).




Over this period, the number of dairy cows has declined while yields of dry matter (t/ha) have increased. As a result, it is estimated that nationally the amount of maize silage produced/cow has increased from about 1.5 to 2.5 tonnes per cow.

The rapid expansion during the 1990's was due to a combination of factors, including the introduction of new varieties which were better adapted to UK conditions, warmer weather and the introduction in 1993 of an arable payment scheme under the Common Agricultural Policy. In future, the area under production might be expected to increase if predicted increases in temperatures are realised. However, elevated temperatures will favour higher maize yields only where there is sufficient soil moisture. Simulation studies have predicted that higher temperatures accompanied by a 10% reduction in precipitation would result in a significant reduction in DM yields in the South and South-east of England (Cooper and McGechan, 1996; Davies et al., 1996). Since the value of the crop is insufficient to justify

[6] Regulation (EC) No. 1831/2003 of the European Parliament and of the Council.
[7] All current silage additives will need to have been approved for use as such by 2010. Regulation (EC) No 1831/2003 requires that, from 2010, all silage additives marketed for sale in the EU must be authorised by the European Commission. Authorisation will be based on a dossier.

irrigation, expansion in the production of maize may be concentrated in the western and northern regions of the UK, which receive higher rainfall but are currently too cool for maize production. Indeed this trend has already commenced; as illustrated in Table 23, there has been a significant change in where forage maize is grown in England, with a move away from the East and South East and a greater proportion grown in the West of the country. As a consequence, more maize will be grown in those areas of the UK with the greatest concentrations of dairy cows. The potential for maize to reduce methane production and improve nitrogen utilisation, discussed in more detail elsewhere in this review, is therefore likely to be enhanced.

Table 23. Breakdown of silage area (% of total area in England) by region[8].

| | 1970 | 2008 |
|---|---|---|
| North West Region | | 1 |
| Yorkshire and Humber Region | 8 | 3 |
| Eastern Region | 35 | 6 |
| East Midland Region | 5 | 7 |
| North West Region | 3 | 9 |
| South East Region | 28 | 16 |
| West Midlands Region | 5 | 14 |
| South West Region | 14 | 44 |

#### 3.3.6.4.6 Whole crop cereal silages

The early harvesting of whole-crop cereals offers a practical way of providing an economical source of home produced forage for feeding to ruminant livestock. They enable farmers outside the geographical limits for forage maize production to produce alternative conserved forage to grass silage. Yields of over 10 t DM/ha can be achieved, and since the crops are harvested at DM contents of between 300 and 600 g/kg, no wilting is required. The inclusion of an additional dietary component, such as whole crop cereal silage constitutes an option to increase energy intake (Phipps et al*.,* 1995). Whole-crop may be either ensiled or preserved with an alkali. A number of factors are likely to be influential in the use of whole-crop cereals as feeds for ruminant livestock, including developments in additives to both improve the ensiling process and the stability of the silage once it has been exposed to air, the cost of production relative to costs of other forages, and changes in support under the Common Agricultural Policy.

#### 3.3.6.4.7 Forage legumes

The main forage legumes now grown in the UK include white clover (in mixtures with grass), red clover, Lucerne and Sainfoin. Each of these has quite different characteristics, and there are particular niches (environment and management) to which each is suited, but lack of persistence can be a disadvantage compared with a grass sward.

Lucerne is tolerant of the winter temperatures experienced in England and Wales and has been grown successfully in the dryer East and South East of the UK, but only on the right type of soils. It is a deep rooting crop which can sustain dry matter production at times of low rainfall, and may therefore become increasingly important in view of likely rising temperatures due to climate change. Where it is successfully grown, it has a high dry matter yield with a high protein content. Moreover, the protein is utilised more efficiently than that of grass silage, and therefore N excretion may be reduced. Although it can be fed green to cattle when other fodder is not available, in the UK lucerne is best made into silage for milk

[8] Source: Ministry of Agriculture (1970 data), Defra (2008 data)

production.

Red clover has potential to produce good yields under a range of soil and weather conditions. It can play a role in short-term leys in livestock/tillage farms or as a “cash crop” on nearby tillage farms. Red clover is best suited as a conservation crop with 3-4 cuts of silage taken each year allowing 6-8 weeks between cuts.

An EU-funded study[9] on the use of forage legumes concluded that forage legumes can play an increasing role in developing more sustainable farming practices. Because they capture nitrogen from the air, they reduce the need for inorganic fertilisers. In addition, using these crops as silage can also reduce the need for high-protein concentrate feeds on the farm. The results of this study have confirmed that silage made from legumes, or from legume-grass mixtures, outperforms silage made from grass alone, and that using silage made from forage legumes can significantly increase the profitability of dairy systems.

#### 3.3.6.5 Methods of feeding conserved forages

The impact on GHG and ammonia emissions resulting from the way in which feeds are fed has received rather less attention than the effects of the feeds themselves. The main methods of feeding conserved forages in the UK are (a) directly from a silage clamp (or ring feeder, in the case of big-bale silage), (b) from a trough (for silage) or rack (for hay) or (c) as part of a complete diet.

Where cows self-feed from the silage clamp, the compacted nature of silage in a clamp reduces the rate of eating by the cow, and if maximum intakes are to be obtained, the cows will need to spend a greater amount of time eating. Where self-feed silage clamps are not under cover, there is the potential for greater ammonia losses. In contrast, where silage is removed from the clamp prior to feeding, it is usually fed in troughs or passageways under cover.

In contrast, complete diet feeding involves mixing the forage and other ingredients into a well-mixed blend, to which the cows are given free access. Mixing the diet in this way has often been shown to increase daily intake and to increase the solids content of the milk of cows. Since the introduction of complete diet feeding in the mid 1970’s, an increasing number of farmers have come to rely on total mixed ration (TMR) feeding in preference to feeding manufactured compound feeds. Although there are no statistics on the number of TMR feeders in the UK, industry sources suggest that 35-40% of dairy herds in the UK are fed in this way[10]. One of the claimed benefits of feeding a TMR is that it allows producers to tailor the ration more accurately for a particular group of cows than would be possible where cows are fed compound feed; if this is true then this offers scope for formulating diets to minimise GHG and nitrogen emissions.

#### 3.3.6.6 Dairy cow fertility and GHG and ammonia emissions

For many years, the declining reproductive performance of high-yielding dairy cattle has been a major problem facing the UK farming sector. Poor reproductive performance results in longer calving intervals, resulting in considerable additional costs to dairy producers. These are largely due to the need to rear more replacement cattle, since fertility has a major effect on the number of heifer replacements required to maintain herd size for a given level of herd milk production. Garnsworthy (2004) has estimated that that up to 27% of methane produced on farms with commercially common fertility levels is produced by herd replacements, and it follows that improvements in fertility would result in lower methane

[9] Legumes for silage in low input systems of animal production (LEGSIL)
[10] Source: Max Ford, Keenan Ltd., *personal communication*

emissions. In the model developed by Garnsworthy (2004), reducing herd fertility levels to 1995 levels would reduce methane emissions by up to 11%. However, there was scope for reductions of up to 24% if fertility were to be improved to ‘ideal levels’.

Improving reproductive performance can also improve N use efficiency. In the USA, Rotz et al. (1999) estimated that reducing the replacement rate of a dairy herd from 35 to 30% would reduce the required replacement animal numbers by 15%, and on a farm where all replacements are raised, the manure N excretion for the herd will be reduced by about 5% with about 6% less N loss from the farm (Rotz et al., 1999). Under UK conditions, Garnsworthy (2004) estimated that a return to 1995 fertility levels would result in a reduction in ammonia emissions of about 9%, but that there was scope for reducing this still further – to 17% - from further improvements in dairy cow reproduction.

In summary, enhancing ruminant productivity generally requires simultaneous improvements in nutrition, genetics and management. While single factor changes such as a better balance of protein or rectifying a mineral deficiency can significantly improve productivity, they soon run into the next limiting factor if these are not being addressed simultaneously.

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3.3.7 On-farm assessment of carbon footprint and GHG emissions in the UK

### 3.3.7.1 A standardised model of carbon footprint and GHG emissions

With the recent focus on the environmental credentials of dairy farms, there has been much interest in models to estimate on the carbon footprint of dairy farm including GHG emissions. A single standard has recently been developed with BSI involvement. Publically Available Specification 2050 (PAS 2050) attempts to consider a wide range of factors that influence the environmental performance of a business. Whilst PAS 2050 approval is not a statutory requirement, it is likely that there will be a commercial advantage to gaining the accreditation. Retailers, processors and other customers will be wary of carbon footprint data calculated from software packages without this stamp of approval.

PAS 2050 is based where appropriate on IPCC guidelines for calculation of GHG emissions. Therefore, any strengths or weaknesses associated with this methodology will follow through into PAS 2050. However, there is scope to update the standard as new information becomes available. Taken from the PAS 2050 literature:

> *"BSI reserves the right to withdraw or amend this PAS on receipt of authoritative advice that it is appropriate to do so. This PAS will be reviewed at intervals not exceeding two years, and any amendments arising from the review will be published as an amended Publicly Available Specification and publicized in Update Standards."*

IPCC methodology is also gradually updated. It is not clear whether PAS 2050 will follow updates to IPCC or whether there may be divergence in the future. It is well known that there are several shortcomings with the IPCC approach to estimating GHG emission from UK dairy farms. For example there is a very limited representation of the effects of diet composition on methane emission. Any software based around the Tier 2 IPCC approach is likely to be of limited value in suggesting dietary improvements due to the insensitivity of the underlying model. Therefore, if IPCC recommendations are slow to change or remain unable to account for the factors of interest, it would seem sensible for PAS 2050 to develop independently where appropriate data are available. PAS 2050, therefore, does make recommendations on how to calculate emissions but it is not sufficiently prescriptive to list the equations involved for all aspects of the dairy farm. Its emphasis tends to be on identifying the factors to be considered, but for many of these factors there is considerable ambiguity in the calculation of the effects.

Within the dairy industry there already exist several on farm tools for estimating carbon footprint and GHG emissions. Initially there was concern over the proliferation of these software packages with an apparent lack of validation or method of comparative evaluation. However, from this point forward PAS 2050 forms a standard against which the Carbon Trust (and possibly other accredited bodies in future) can judge existing estimates from emissions models. Of those software packages commercially available, one is Carbon Trust certified and another is due to complete certification and PAS 2050 approval shortly. In theory, all software developed to Carbon Trust/PAS 2050 approval should deliver similar estimates of GHG emission for a given business. There may be small areas of technical difference where elements not currently considered as part of PAS 2050 are included in some software packages, but it is likely that the principal difference will be one of user experience and presentation of results. Therefore, whilst PAS 2050 may not at this stage consider the full range of factors at play in determining GHG emission from a dairy enterprise, it does at least have industry acceptance. All future software built to the PAS 2050 standard will have to update in line with any revised standards to maintain its certification. Rather than trying to coax industry into using one specific piece of freely available software, the adoption of a single standard (PAS 2050) to which commercial packages can conform will almost certainly prove to be a more effective way of harmonising the approach to estimate GHG emissions. It follows that it will be very important for the industry to have an input into the specification of PAS 2050 as it develops.

PAS 2050 is broadly based and not designed specifically for agricultural or horticultural production but aimed at food products. The pre-farm gate component of this approach is most relevant to the current DairyCo project. Milk was one of the food products included in the project designed to test the validity of the LCA approach in GHG footprinting. Emissions of $N_2O$ are based on the highest Tier approach used by any given country, so in the case of the UK this is Tier 1.

#### 3.3.7.2 Software tools to estimate carbon footprint and GHG emissions

A number of farm-scale benchmarking greenhouse gas emissions calculators have been developed over the past few years. Most are based on the IPCC methodology for the country in which they were developed. However, some are not. Below we briefly review each of them and a summary is shown in Table 24. They are categorised by country of origin.

##### 3.3.7.2.1 United Kingdom

###### *3.3.7.2.1.1 E-$CO_2$ (Energy and carbon assessment) (CMS/Kite consulting)*

Kite consulting and CMS have combined forces to produce a business to business model of carbon footprint for the dairy industry. By the end of January 2009 this tool should receive Carbon Trust certification/PAS2050 approval. The two companies have a large customer base comprising dairy farmers, processors and retailers. Both companies have experience providing business and environmental audits to the industry (e.g. water and energy audits for Defra). The E-$CO_2$ model has been designed to be PAS 2050 compliant. As such they are keen to be involved in the development of PAS 2050, thereby allowing advances in knowledge to be integrated within the system. The underlying model to the E-$CO_2$ software is spreadsheet based, but the software is packaged with a user friendly interface. The software is designed to be used on farm with the help of a Kite/CMS consultant. There is a strong emphasis on the quality of the data gathering process. Farms are graded (1-4) according to how robust their input data appears to be. This allows the subsequent analysis of combined data from different herds to be filtered and weighted accordingly. Rogue data can therefore be identified and treated accordingly. The software is not web based as it would limit the control over the input of data.

The model fits with the business-to-business approach of PAS 2050. For example, moist byproduct feeds such as brewer's grains are carbon credited to the farm business at a 50% rate to allow the supplier to claim the remaining carbon credit for avoiding landfill with what would otherwise be a waste product. The environmental costs of haulage are included where possible. All field operations and cropping are included within the model, as are a large number of other factors including waste management, rainwater recycling, milk cooling technology and machinery usage. Different farm management strategies are considered, including the likely differences between machinery usages for contracted operations versus those conducted in house. All utility bills are analysed to determine energy usage and information from costing is used to determine herd performance (production and feed utilisation).

Results are displayed with an emphasis on comparing results against other farms. Performance is therefore considered to be relative to others in the industry. This allows farmers to interpret emissions in a more meaningful way. A basic report is left on farm immediately following the consultant's visit. A more thorough analysis is delivered at a later date, once the data have been checked. The results have a strong economic element. Suggested changes to management practice are expressed as financial savings against the status quo. As well as carbon emissions, nitrogen pollution is also considered within the framework of the NVZ regulations. Data is also gathered on HLS/ELS conservation schemes and any potential carbon credits could be integrated in the future.

*3.3.7.2.1.2 $CO_2$ Emissions footprint tool (Dairy Crest Direct) (incorporating Forum for the Future model of GHG emissions)*

The Dairy Crest Direct $CO_2$ emissions software is a tool that has been developed by collaboration between the following organisations:

- Centre for Sustainable Energy
- Forum for the Future
- Farming and Wildlife Advisory Group (FWAG)
- Institute of Grassland and Environmental Research (IGER)
- AEA Energy and Environment (Jim Webb)

The original Forum for the Future GHG calculator was developed several years earlier than the CLA CALM calculator, but is based on very similar principles. The spreadsheet model was aimed at Dairy enterprises and is based on the UK IPCC emission factors from the 1996 guidelines. It comprised an excel-based tool with a series of questions on several worksheets (livestock, land, electricity, transport fuels, renewable energy and carbon) designed to extract the information on livestock numbers and categories as well as land use (cropping types) and N fertiliser use. It was strong in the area of energy use, and includes $CO_2$ emissions as a result of electricity and diesel use with an aim of helping to identify savings. Methane, $N_2O$ and $CO_2$ emissions as well as C storage are calculated at the farm level and summarised in a final worksheet where data are expressed as tonnes of $CO_2$e/yr or as kg $CO_2$e/litre of milk or per dairy cow. A list of management practices which would result in C storage are included in one of the worksheets, e.g. producing energy crops (SRC Willow, oilseeds for biodiesel), woodland generation, straw incorporation and manure applications. Indirect emissions are defined as $CO_2$-energy used to produce N fertiliser, and these are estimated and can be included in the farm total emission.

The Dairy Crest Direct software integrates this spreadsheet model with a user friendly web based interface along with updates to move the calculations from IPCC (1996) to IPCC (2006). The software is designed to allow farmers to enter their business details into the system with default values applied for areas of missing information. The user interface aims to be as descriptive as possible in an attempt to avoid ambiguity. The user guide lists the information sources that are required to complete the exercise as follows:

1. Livestock
    a. Average numbers and sizes of livestock on the unit
    b. Yearly milk production
    c. Information about livestock waste management
2. Land
    a. Average amount of nitrogen applied per hectare
    b. Nitrogen applied from manure and slurry over the area, or estimates
3. Electricity
    a. Electricity bills to cover the time period you would like to look at
4. Transport fuels
    a. Fuel bills for the same time period or estimates of the amount used in litres
5. Renewable energy and Carbon

a. Areas over which various cultivation practices take place which help add to organic matter

Once this information has been entered by the user, the tool generally applies the IPCC methodology to calculate GHG emissions, for example using a fixed 6.5% emission factor for methane based on estimated gross energy intake. However, there are some deviations or extensions to the IPCC recommendations as with the re-categorisation of manure handling systems.

The broad approach adopted by the Dairy Crest Direct model includes an account of the embedded GHG emissions from growing and transporting feedstuffs (bought in or grown on farm). In a similar manner, emissions associated with purchased livestock are also considered. Nitrous oxide emissions from the land are calculated in a similar manner to IPCC guidelines with the use of emission factors, although recent evidence on the magnitude of these factors has been incorporated in a deviation from existing IPCC recommendations. Emissions associated with electricity usage are estimated from the annual accounts detailing the total spend by the business. Purchased fuel is also directly accounted for. Fuel use by contractors is also estimated based on a proportion of the total cost to the business over the year. Soil carbon sequestration is considered based on changes in land use and certain management practices including, woodland management and manure application. Energy costs (and therefore emissions) are considered from water usage and these are offset against any on-farm waste water treatment activity.

Overall the Dairy Crest Direct software is well presented and considers a wide range of factors influencing GHG emissions from the farm. The software is not currently Carbon Trust certified. However, discussions between the relevant parties and the Carbon Trust are taking place. The calculations involved in the model are readily available through the associated documentation together with a list of academic and industry references. The software takes a different approach to data gathering to that of the Kite/CMS software in that it relies on direct interaction between end users and the software via the internet. Therefore, there are advantages in terms of ease of access, but the ability to quality control the data entry process is limited to automated parameter boundaries.

#### *3.3.7.2.1.3 WhiteGold (AB Agri)*

White Gold Carbon Footprint initiative was commenced two years ago and is currently the only available software tool for calculating GHG emissions on-farm with Carbon Trust certification. The AB Agri system was developed in conjunction with Sainsbury's and Kingshay Farming Trust and aims to understand and improve the carbon footprint and environmental performance of dairy farms. It is currently being used exclusively with the Sainsbury's milk supply pool where the trained White Gold assessor visits each farm. The assessment benchmarks the current practice on-farm using an environmental scorecard and an electronic GHG model. The environmental scorecard is a subjective questionnaire with scores weighted according to the response based on best practice. The questions cover 12 key areas including, livestock, nutrition, manure, fertiliser, energy, water and fuel. The second part of the assessment involves the collection of data on farm which is then returned to the central office for input into the electronic GHG model. The model considers direct GHG outputs such as those arising from livestock, manure, manure spreading, fertiliser, electricity, fuel, contractors etc and indirect sources arising from purchased feed, fertiliser manufacture, bedding and others. The model and its functionality are covered by intellectual property and confidentiality agreements. Nonetheless, models submitted for Carbon Trust certification have to be in spread sheet form and predominantly based on IPCC methodology. Where the model does extend IPCC recommendations is through the provision of carbon footprint data for a comprehensive range of feed ingredients and compounded feeds which allow the model to calculate carbon footprints for any diet the farmer may be feeding. The model calculates a quantitative assessment of GHG emissions and provides the farmer with a total $CO_2$ output per cow and per litre of 4% FCM. The outputs from the model are sent back to the farm as a full report. The software delivers a list of actions that the farmer can put into place to reduce emissions. The cyclical process of 'measure, review, improve – measure, review, improve', aims to reduce the GHG emissions resulting in benefits to dairy farmers and the environment. When an individual assessment has been completed the farmer then receives a 'traffic light' style result sheet which allows them to benchmark their farm against others in the pool. This highlights areas for potential improvement and cost savings. AB Agri's next objective is to make the model laptop based so that it can be used directly in the field.

*3.3.7.2.1.4 $SIMS_{DAIRY}$ (North Wyke Research)*

$SIMS_{DAIRY}$ (del Prado and Scholefield, 2008) is a modelling framework, which integrates existing models for nitrogen, phosphorus and economics. It is capable of describing all of the complexity of the interacting nutrient transformations within the soil, plant and animal components of a dairy system to calculate levels of losses to water (nitrate and phosphorus) and the air (nitrous oxide, methane and ammonia), in response to nutrient inputs, farm managements (stock, swards and manure) and specific site conditions (climate and soil). At the same time it calculates the levels of production and farm profitability within its economic module so that pollution and farm viability can be explored within the same system. In doing this $SIMS_{DAIRY}$ has the capability to search for and specify optimal conditions for satisfying environmental and production criteria. Additionally, $SIMS_{DAIRY}$ has the unique capability to integrate the effects of management on some of the more socio-economic strands of sustainability that are not yet amenable to most modelling systems and to quantify how changes to these criteria might affect profitability. These include animal welfare, food quality for human health and product saleability, landscape aesthetics and biodiversity. Since the model calculates ammonia volatilisation and nitrate leaching, indirect $N_2O$ emissions are also accounted for. Thus, $SIMS_{DAIRY}$ can be used to explore management options that reduce emissions of $N_2O$ and $CH_4$, whilst assessing any trade-offs or win-wins. Emissions of $N_2O$ or $CH_4$ are expressed as kg/ha/yr or kg/litre of milk, or as $CO_2$e/ha/yr or kg/litre of milk. The model would require further work, e.g. development of a user friendly interface, before being of use to advisors and farmers.

*3.3.7.2.1.5 CALM (Carbon Accounting for Land Managers) (www.calm.cla.org.uk)*

The Country Land and Business Association CALM calculator is a recently developed web-based tool aimed at generating a greenhouse gas/carbon inventory at the farm level. It has been developed to take into account all farm types, hence it includes dairy systems. It was developed using UK IPCC default emission factors using the revised 1996 guidelines. There are a series of questions on several worksheets that extract all the necessary information about the farming system. The information required is generally known by the farmer, e.g. number and categories of stock, quantity of N fertiliser used, land and cropping areas etc. The tool calculates total annual emissions of $N_2O$, $CH_4$ and $CO_2$, and takes into account C storage on the farm, e.g. in woodland areas, resulting in a final total of tonnes of $CO_2$e per farm. Some externalities are also included in its calculations, e.g. the energy used in producing N fertiliser. The tool also accounts for indirect $N_2O$ emissions associated with nitrate leaching and N deposition.

A recent review of the CLA CALM model by Natural England concluded that farmers would consider annual assessment of their emissions. However, the reviewers noticed that many management practices that farmers suggested they might undertake to reduce emissions would not necessarily address the principal sources of greenhouse gases on their farms. They tended to suggest practices that would reduce the highly visible inputs that farmers are familiar with e.g. energy and fuel use on the farm, suggesting less awareness of the non-$CO_2$ greenhouse gas emissions and no direct cost savings associated with tackling them.

The CALM calculator does not present results on a per litre of milk or a per hectare basis. Whilst this calculation could be estimated by the user, the absence of this information does provide a barrier to uptake on farm.

*3.3.7.2.1.6 Carbon calculator (www.cplan.org.uk)*

CPlan is a web-based calculator that is presented in two versions as follows:

- A free carbon equivalent calculator – which allows you to calculate the carbon equivalent foot print for your business. The calculator is anonymous and has no registration, your data will not be stored; it will disappear when you leave the page.
- A members section with:
  - Facility to calculate and store specific data for your business.
  - Test alternative scenarios (e.g. reduce your fertiliser application and see the difference to your calculation).
  - Access to a farmer's forum to hear innovative ideas from others in the industry.
  - Access to a wide array of mitigation advice.

  These additional features have associated costs so a small charge is made if a farmer wishes to become a member.

The current C-Plan calculations follow IPCC 2006 methodology. However, C-Plan is aware of the potential shortcomings of this approach when applied to dairy farms. They state that many of their members and visitors to the site have commented that IPCC methodology is not totally appropriate at the farm scale. Therefore they have been working with other research scientists and have produced a discussion document which examines the issues which have been raised.

3.3.7.2.2 New Zealand

*3.3.7.2.2.1 OVERSEER (www.agresearch.co.nz/overseerweb/default.aspx)*

The OVERSEER® Nutrient Budgets model has been developed by AgResearch, New Zealand and combines nutrient budgets with indices derived from these nutrient budgets, to provide users with a tool to examine the impact of nutrient use and flows within a farm (as fertiliser, effluent, supplements or transfer by animals) on nutrient use efficiency and possible environmental impacts. The model also provides a means to investigate mitigation options to reduce the environmental impact of nutrients (including greenhouse gas emissions) within a land use.

The greenhouse gas component of OVERSEER® is based on models and algorithms used for New Zealand's greenhouse gas national inventory, but with improvements to include on-farm management practices. Methane emissions are based on a metabolic energy intake model developed by Clark (2001). $N_2O$ emissions are based on the New Zealand IPCC-based inventory, which includes the use of emission factors for direct $N_2O$ losses from excreta, fertiliser and effluent, and indirect losses from leached N and volatilised ammonia. Leached N and volatilised ammonia are estimated from the associated N budget model. Methane and nitrous oxide emissions are converted to $CO_2$ equivalents.

$CO_2$ emissions from fuel and electricity, processing and some indirect contributions (e.g. fertiliser manufacturing) are largely based on the data of Wells (2001). In many settings, default values are presented, but can be overridden by the user if required. Processing data for dairy factories was extracted from surveys. $CO_2$ emissions generally refer to total embodied emissions, that is, total $CO_2$ cost associated with the use and production of fuel, electricity, fertiliser, etc.

#### 3.3.7.2.3 The Netherlands

##### *3.3.7.2.3.1 DairyWise (Schils et al. 2007) (Also known as BBPR)*

DairyWise is a whole-farm dairy model, similar to $SIMS_{DAIRY}$. It is an empirical model that simulates management, environmental and financial processes on a dairy farm. Central to the model is the FeedSupply model that balances animal requirements for a given level of productivity. The outputs of this model act as inputs to the environmental and economic sub-models, which simulate nitrogen and phosphorus cycling, nitrate leaching ammonia, $N_2O$ and $CH_4$ emissions, energy use and financial farm budget. The model is based on empirical data from crop and animal studies conducted in the Netherlands on the main Dutch soil types (clays, sands and peats), hence its applicability is restricted to these regions. Emissions of $N_2O$, $CH_4$ and $CO_2$ are calculated with emission factors used in Dutch emission inventories, but related to soil type and ground water level, with generally higher emissions on organic soils and wetter soils. Indirect $N_2O$ emissions associated with nitrate leaching and N deposition are also included. Emissions of $CO_2$ related to fossil fuel based energy use are also included. Typical outputs are kg $CH_4$ or $N_2O$/ha.

##### *3.3.7.2.3.2 KLIMAATLA (Climate yard stick of CLM (Centre for Agriculture and Environment) www.clm.nl; www.klimaatlat.nl)*

This is an on-line tool to raise awareness of the major sources of greenhouse gases on dairy farms and which measures can be used to reduce these emissions. Dairy farmers enter their farm data and the tool calculates emissions of greenhouse gases. Entering all the data required is not supposed to take more than 15 minutes. A second module determines the effect of prevention measures. Emissions from each source are expressed as kg $CO_2$e/kg milk and can be compared with average emissions. The comparisons module is not publicly available, but is used for farmer study groups. It is not clear from the information available whether this is an IPCC emission factor based tool. Suggestions to reduce emissions include more careful practice of N fertiliser, keeping less stock, adjusting the feed ration or cooperation with a neighbouring farmer to invest in an anaerobic digester. CLM suggest that Dutch dairy farms can lower their emissions for greenhouse gases by about 10%.

#### 3.3.7.2.4 Denmark

##### *3.3.7.2.4.1 FarmGHG (Olesen et al., 2006)*

FarmGHG is a flow-based model for estimating greenhouse gas emissions from livestock farms. It calculates nitrogen and carbon budgets and allows for imports and exports off the farm. Its aim is to allow quantification of all direct and indirect gaseous emissions from dairy farms and allow the assessment of the impact of mitigation methods. It is made up of a number of submodels: system, external (handles emissions associated with import of energy fertiliser, pesticides, feedstuffs etc.), farm, crop rotation, housing, field, animals, feed storage and manure storage. Emissions of $N_2O$ and $CH_4$ are generally calculated using IPCC methodology (revised 1996 IPCC Guidelines).

#### 3.3.7.2.5 Republic of Ireland

*3.3.7.2.5.1 MDSM-GHG (Shalloo et al., 2004)*

The Moorepark Dairy Systems Model (MDSM) was linked to a GHG model to develop a systems approach to quantify GHG fluxes from Irish pastoral dairy production. The model was developed to determine what effect management practices would have on the emissions of $N_2O$, $CH_4$ and $CO_2$ from typical Irish dairy farms. It simulates emissions from the farm, as well as associated 'off-farm' emissions associated with imports/inputs onto the farm and indirect emissions associated with nitrate leaching and N deposition. The model was developed to take explicit account of mitigation strategies at the farm level capable of reducing emissions per litre of milk.

The MDSM is used to define herd size, milk yield, land use, manure and fertiliser applications etc. for a given 12 month period. On farm emissions of $N_2O$, $CH_4$ and $CO_2$ are estimated using emission factors sourced from the literature with relatively few emission factors taken from IPCC (1996 Guidelines). Off-farm emissions of $CO_2$ associated with diesel, electricity, lime and fertiliser production are sourced from the literature, whilst calculations of emissions associated with supplementary concentrate feedstuffs (crop cultivation, transportation, processing etc) were calculated by Shalloo et al. (2004). Account is also taken of soil oxidation of methane (Boeckx and van Cleemput, 2001). On-farm and total GHG emissions are expressed as Mg $CO_2$e/yr. The relative strength of each source is also represented.

The authors summarise that Irish dairy production has a heavy reliance on grazed pasture to provide the majority of annual energy intake. Therefore the potential to reduce GHG emissions through the use of enteric fermentation modifiers, farm waste treatment facilities or the manipulation of basal ration is limited. The model was therefore developed to evaluate the potential effect of varying levels of concentrate feeding and differing genetic potential for total GHG emissions as well as farm profitability.

### 3.3.7.3 References

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Table 24. Summary of farm-scale carbon footprint and GHG emission software.

| **Model/tool** | **Country** | **Sector** | **Capacity** | **Carbon Trust certified (PAS2050)** | **$CH_4$ method** | **$N_2O$ method** | **$CO_2$ method** | **$NH_3$** | **Comments** |
|---|---|---|---|---|---|---|---|---|---|
| E-$CO_2$ software (Kite/CMS consulting) | UK | Dairy | $CH_4$, $N_2O$, $CO_2$, Energy | Due Jan 09 | IPCC (2006) | IPCC (2006) | IPCC (2006) | NA | Available via Kite/CMS through a consultant service. |
| Emissions Footprint Tool (Dairy Crest Direct) | UK | Dairy | $CH_4$, $N_2O$, $CO_2$, Energy | Discussions ongoing | IPCC (2006) | IPCC (2006) | UK GHG inventory | | Forum for the Future spreadsheet model has been integrated into the Dairy Crest Direct software |
| WhiteGold (AB Agri) | UK | Dairy | $CH_4$, $N_2O$, $CO_2$, Energy | YES | PAS2050 | PAS2050 | PAS2050 | NA | |
| $SIMS_{DAIRY}$ (North Wyke Research) | UK | Dairy | $CH_4$, $N_2O$, $CO_2$, $NO_3$, P, sediment, productivity, welfare, biodiversity, landscape | NO | Literature values | Literature values | N/A | | Still require a simplified interface for non-experts. |
| CALM (Country Land and Business Association | UK | All | $CH_4$, $N_2O$, $CO_2$ (energy) | NO | IPCC (1996) | IPCC (1996) | IPCC (1996) | | |
| Carbon Calculator (CPLAN) | UK | All | $CH_4$, $N_2O$, energy, $CO_2$ | NO | IPCC (2006) | IPCC (2006) | IPCC (2006) | NA | Web based tool |

Table 24 (cont). Summary of farm-scale carbon footprint and GHG emission software

| Model/tool | Country | Sector | Capacity | Carbon Trust certified (PAS2050) | $CH_4$ method | $N_2O$ method | $CO_2$ method | $NH_3$ | Comments |
|---|---|---|---|---|---|---|---|---|---|
| OVERSEER | NZ | Dairy | $CH_4$, $N_2O$, $CO_2$ (energy), embodied $CO_2$, $NO_3$, $NH_3$, productivity, nutrient budgets | N/A | NZ IPCC | NZ IPCC | NZ IPCC | | |
| DairyWise (BBPR) | NL | Dairy | $CH_4$, $N_2O$, $CO_2$ (energy), $NO_3$, $NH_3$, productivity | N/A | Dutch Inventories (IPCC) | Dutch Inventories (IPCC) | Dutch Inventories (IPCC) | | |
| Klimaatlat | NL | Dairy | $CH_4$, $N_2O$, $CO_2$ | N/A | | | | | |
| FarmGHG | DK | Dairy | $CH_4$, $N_2O$, $CO_2$ (energy), $NO_3$, $NH_3$ | N/A | IPCC (2000) | IPCC (2000) | IPCC (2000) | | |
| MDSM-GHG | IRL | Dairy | $CH_4$, $N_2O$, $CO_2$ (energy), productivity | N/A | Literature values and IPCC | Literature values and IPCC | Literature values and IPCC | | |

# 4 APPENDIX

## 4.1 FUTURE RESEARCH

### 4.1.1 Quantifying GHG emissions from UK dairy farms

**1**. Defra's annual greenhouse gas emissions inventories are calculated using IPCC revised 1996 methodology (year 2000 Good Practice Guidance Document). However, as the review has demonstrated, there exist a number of more recent data sets and statistical models that could be used to provide more accurate estimates of emissions from the dairy sector. Not only could accuracy of current emissions be improved, but projections could now be made that factor in the likely efficacy of specific mitigation strategies, if the structure of the inventories facilitated it. Such valuable information could be used to promote the progress being made by the dairy industry and it would highlight where further significant reductions could be made. An emissions inventory calculated using the best available scientific knowledge would be a logical starting point for any analysis of mitigation strategies at the national level. Use of the existing Defra statistics based on IPCC 1996 methodology introduces a large error at the commencement of any modelling study and it is an urgent priority to provide industry wide statistics that are a better representation of current performance against environmental targets. The inaccurate and crude nature of the currently available statistics have led to widespread belief that trends in emissions are purely a function of changes in the size of the national dairy herd ($CH_4$) or total fertiliser application ($N_2O$). Whilst these effects are likely to be important, the IPCC 1996 guidelines are too blunt an instrument to meet the requirements of the dairy industry (and agricultural industry as a whole) as it looks to meet its objectives on the way to a more sustainable future. A new 5-year Defra funded project (AC0112) has recently been commissioned (NWRes led) to improve the structure of the nitrous oxide and methane inventories and better reflect country specific emission factors and potential mitigation methods. The project will also link the greenhouse gas inventories to the national ammonia ($NH_3$) inventory. We believe that this is a great opportunity to introduce new modelling approaches, particularly for estimating methane emissions from dairy cows. This Defra project will deliver its main output in 5 years time. However, an initial project could involve calculating GHG emissions from either the dairy industry or the whole agricultural sector (with other levy bodies) over the period 1990-2006 using alternative methodology which is currently available and comparison with the current inventory values. This output and lessons learnt from this more timely project could then feed into the main Defra project.

**2**. A significant limiting factor in the industry's quest for reduced emissions of GHG is the lack of available data regarding current feeding practice. For fertilisers, Defra sponsor an annual Survey of Fertiliser Practice which provides data on changes in fertiliser use in GB. This allows us to claim, with some confidence, that over the past 10 years fertiliser use on grassland has declined, resulting in lower N contents in herbages and lower N intakes by grazing livestock. However, there is no equivalent information base on how farmers feed their dairy cows. This review has identified a number of strategies for reducing GHG emissions on dairy farms, but there is no central body of data to inform on the extent to which these strategies have been adopted, if at all, or the potential for their uptake. Consequently, there is no baseline by which to judge the potential for reducing GHG emissions from the strategies identified in this review.

This review highlights specific knowledge gaps for the individual GHGs, for example, the need to identify the potential of supplemental fat in dairy rations as a strategy for reducing methane emissions. The need to fill such a gap is real, but unfortunately apart from anecdotal evidence we have no information on how much or what type of fat is currently being added to dairy cow diets. Data are available from the Defra monthly survey of compound feed manufacture on the total amount of oils and fats added to compound feeds, but not the type, nor the category of compound (dairy, poultry, pig etc) to which the fat has been added. Therefore, it is not possible to establish from these data the amount or type of fat added to dairy cow rations, and hence the potential for further reductions in methane production by the inclusion of fat. Moreover, the survey only applies to compound feed manufacture. Commercial suppliers of fat supplements have data on the use of their products, but the information is highly confidential in a very competitive market. A significant proportion of cows (exact numbers unknown) are fed complete diets that are prepared from straights on farm and don't include compound feeds, and as a result are not covered by the Defra survey. There is no central body of data on the number of cows or herds that are fed complete diets, let alone what they are fed.

The lack of this information is a major barrier to identifying the potential for reductions in GHG emissions using the strategies we have identified. With this information, DairyCo would be better placed to identify which strategies are likely to have the greatest marginal return and ensure better focus for both extension services and further research. Much of the data necessary to develop a baseline of feeding practice may already be available, but in a range of disparate sources. An initial project would involve collating data currently available, identifying the remaining critical gaps and ascertaining the appropriate methods to obtain the necessary data.

### 4.1.2 Methane

Experimentation

Whilst the available knowledge base concerning the factors controlling methane emissions from dairy farms has been expanding greatly over recent years, there still remain some important areas where information is scarce. For example, an analysis is urgently required of the effects of nutritional mitigation strategies to reduce enteric emissions (e.g. added fat, increased starch), on the composition of manure and subsequent emissions from storage or spreading. In addition to an experimental programme, models need to be developed that are capable of predicting emissions from other farm sources, such as ‘unmanaged’, non-productive areas, e.g. poached areas around drinking and feeding troughs, where urination and defecation result in large fluxes of nitrous oxide and methane. At present the absence of information has led these sources to be omitted from farm level GHG inventories.

The scope for breeding animals which produce less methane per day without any associated losses in production has not been adequately assessed especially given its large potential impact. It is speculated that animals with reduced methane emissions but uncompromised performance occur within a population due to a more efficient feed conversion ratio, smaller frame size and faster digesta kinetics or altered digestive function. Work is therefore required to determine whether these differences are related to intake behaviour, or to potential anatomical and physiological differences. This work is likely to involve the analysis of GHG emissions from different dairy breeds.

Further research is also necessary on potential biological control organisms targeting rumen methanogens or the protozoa which they associate with. In addition, the adaptation of methanogens to chemical inhibitors is not understood and monitoring and assessment of adaptation is required for novel control strategies such as vaccines or biological controls. The role of natural bacteriocins in inhibiting rumen methanogenesis looks promising but needs further evaluation. Investigations are underway as part of Defra project AC0209 to establish the effects of some of the naturally occurring compounds on rumen function. However, there will still remain gaps in our knowledge due to the large number of potential compounds. In particular, there is a limited amount of information on the range, activity and hydrogen threshold of naturally occurring gut-dwelling reductive acetogens.

Modelling

Further work is required to target specific gaps that have been identified as a result of this review and these are prioritised below.

**1.** As work is published regarding the quantitative effect of supplemental fat in dairy cow diets on methane emissions, our biological understanding of the mechanism involved has advanced. However, models of ruminant metabolism have failed to keep pace with such developments and as discussed earlier, it seems unlikely that the more basic statistical models will provide a quick solution to quantifying the effects of supplemental fatty acids. A major research initiative in this area is therefore required to collate data on the relationship between the intake of specific fatty acids and methane emission. This will involve the detailed analysis of existing experimental data across a number of studies to determine the intake of individual fatty acids and their relationship with methane output whilst accounting for nutrient interactions due to variation in the basal diet. Typical fatty acid profiles for supplemental fat sources could be established and combined with the statistical models arising from this research to suggest the most effective application of supplemental fat as a methane mitigation strategy.

**2.** A significant barrier to applying any model that attempts to relate available nutrition to methane emissions has been the poor description of intake. Many statistical models of intake in ruminants are available, but by their nature they are tied to the specific environment in which they were created. Therefore, both mechanistic and statistical models of methane output have tended to require intake to be defined as an input to the model. This is not a problem where intake has been measured directly for a given diet, as is often the case in studies directed at model evaluation, but it does create substantial problems where models are required to predict future emissions with no estimates of likely intake, such as is the case for inventory purposes. Given the overwhelming importance of intake as a descriptor of methane emissions, a statistical modelling exercise is required to update and redefine suitable intake prediction equations for UK dairy cows.

**3.** The review has described the concept of sources and sinks for excess reducing power produced during the anaerobic fermentation process with conversion to methane acting as the final sink following all other transactions. Our knowledge of fermentation biology is more advanced than the framework adopted by existing mechanistic models would suggest, with several other sources and sinks likely to play a role in the overall process. Dynamic models have yet to include these elements and progress is likely to remain limited in these areas without additional quantitative observations. A desktop study is therefore required to categorise and quantify the missing sinks and sources in the existing models.

**4.** It has been shown that as the rate of fermentation increases due to the feeding of increased readily fermentable carbohydrate, the rate of methane emission declines per unit of feed degraded (Pelchen and Peters, 1988). This fits neatly with observations of reduced methanogenic activity as rumen pH declines. However, existing models do not account directly for this effect, with even the most advanced mechanistic schemes including only a crude effect of pH on cellulolytic activity (Dijkstra et al., 1992). Simulating the effects of rumen pH on the diverse microbial groups present in the rumen remains a challenge, but perhaps an even greater task is to model adequately the diurnal fluctuations in pH itself.

**5.** Non-nutritional factors including the effects of direct pharmacological interventions such as rumen modifiers (e.g. ionophores) and selective vaccination have yet to be incorporated into mechanistic models. The lack of distinct subdivisions between microbial groups in existing models has thwarted what could have been a relatively straightforward application of these simulation models whereby one or another group is selectively limited or destroyed. A desktop modelling exercise is required to expand the representation of microbial groups within current dynamic models.

**6.** Existing modelling research has focused upon animals adapted to their diet for good reason, namely because this is quantitatively more important as far as emission inventories and animal production are concerned. However, the process of adaptation from one diet to the next is of significant interest. A quantitative description of the transitional phase would further our understanding of gastric fermentation, possibly identifying key control mechanisms in the shift from one type of fermentation to another. A model capable of accurately simulating transient changes in emissions through dietary manipulation will have to account for specific inter-microbial relationships and a modelling project to provide such a description could be combined with the need to model the effects of non-nutritional factors (see point 5). Therefore, a study is needed to deliver an improved mathematical description of multiple microbial groups in the rumen.

Other gaps
Problems arise when attempts are made to model the long term consequences of intervention strategies. Extant models of rumen function have been developed to describe the steady state biology as characterised by an animal well adapted to its diet and environment. In practice, intervention studies rarely consider treatments in these terms leading to questions about the longer term efficacy of these methods. Indeed Johnson et al. (1994) indicated that the effect of monensin and lasalocid was not sustained beyond 16 days of treatment. This is most likely the result of adaptation by the rumen microorganisms given their short generation time and genetic diversity. If enough data were available to describe the long term effects of pharmacological treatments, there would still exist an opportunity to model the adaptation to these treatments. This applies equally to the potential to model the transitional response following a conventional nutritional change. Until further experimental data becomes available following long-term intervention studies, the models will remain as predictors of steady state systems.

A longer term view is also important when considering how to account for changes in methane emissions with the inevitable progression from one physiological state to another. As time advances, animals grow and develop and they move from juvenile to adulthood. Subsequently they will also experience physiological changes as they progress through pregnancy and lactation. Homeorhetic control throughout this development is brought about through the endocrine system, itself a function of an individual's genes. Extant models assume these factors will ultimately manifest themselves as effects of intake and nutrition. However, it is conceivable that other modes of action to affect methane emissions are possible. For instance, it is known that intestinal morphology can change substantially following parturition with Gibb et al. (1992) demonstrating a 15% increase in small intestinal length between calving and mid-lactation in Holstein dairy cows. The same study showed that an even greater (50%) increase in mass was observed for rumen tissues and rumen contents. These changes will affect the extent of degradation in the rumen and also the quantity of nutrients available for fermentation in the hind gut. Unfortunately, the invasive and expensive nature of the experimentation required to provide further quantitative estimates of such physiological changes will limit the available data for model construction an evaluation.

### 4.1.3 Nitrous oxide

**1.** The current IPCC $N_2O$ inventory methodology for the UK needs to be further developed to include country specific emission factors which better reflect the UK's soil, climate and management systems. This means development to Tier 2 and eventually Tier 3 methodology. Further review and experimental work is required to generate country specific emission factors for the main agricultural sources of $N_2O$. The new five year Defra funded inventory project will work towards this. However, further experimental work may be required to generate country (perhaps site specific) $N_2O$ data to validate tier 3 modelling. Emission sources that may require validation/empirical data include; grazing land, inorganic and organic N fertilisers, and crop residues. The influence of N rate and timing of N applications/deposition will also influence $N_2O$ emissions. Hence, any Tier 3 modelling will need to take these factors into account. Activity data at the relevant spatial scale will also be required.

**2.** Mitigation methods need to be developed, tested and validated in the field, and the IPCC inventory developed to reflect the main mitigation methods. The new five year Defra funded inventory project aims to provide a framework that acknowledges key mitigation methods. However, experimental work will be required to determine the efficacy of a number of key mitigation strategies, including the use of nitrification and urease inhibitors with a range of N sources (urine, inorganic N fertilisers and manures) applied to land. The delivery of nitrification inhibitors also needs to be assessed e.g. applied directly to the soil, mixed with slurries or coated on or included in the mix of fertiliser N prills. Other delivery mechanisms for nitrification inhibitors to urine patches include the use of rumen boluses or tail operated 'spray-cans' (NZ research).

**3.** The effect of diet on N excretion and subsequent losses of $N_2O$ and $NH_3$ is still not fully understood, and obviously linked to effects of dietary manipulation to control methane emissions. Some key information about the potential for diet selection and dietary additives to reduce methane emissions and N excretion will be provided by the Defra project AC0209. However, further experimental work will be required to follow the impacts of the urine (and faeces) on emissions of $N_2O$ and $NH_3$ after urination or manure spreading.

**4.** Crop residues are a significant source of $N_2O$ in the UK agricultural inventory. Of most relevance to the Dairy industry are residues from fodder crops and maize silage production. Further $N_2O$ measurements are needed to determine country specific emission factors for crop residues as there are few data to support current emission factors used in the IPCC $N_2O$ inventory.

**5.** Maize silage production is key to the economic sustainability of many dairy farms. Inclusion of maize silage in the dairy cow diet is also seen as an approach to reduce methane emissions. However, the losses of nitrate and ammonia (indirect sources of $N_2O$) and $N_2O$ (direct) from the fertiliser and manure applications to maize land could off-set any reduction in methane emissions from the dairy cows fed the silage maize. A whole system approach to the production and use of maize silage is required to determine best practices ensuring overall reduced greenhouse gas emissions.

**6.** Anaerobic digestion can be used to reduce the methane emissions from the management of livestock manures. In order to promote this technology as an effective greenhouse gas mitigation strategy, it is important to know that the resulting digestate can be used for agronomic benefit without detrimental impacts on the environment. Research is required to benchmark the agronomic effects of anaerobic digestate from a range of sources (on farm without food wastes to centralised AD plants that use food wastes) with conventional manure types, such as undigested dairy slurry. Since anaerobic digestion increases the pH and the availability of nitrogen in the digestate compared to the input slurry, there is potentially a greater risk of ammonia volatilisation and nitrate leaching (indirect sources of $N_2O$ emissions) following application to land compared to conventional slurry application. On the other hand, if most of the available carbon has been used up within the digester (to produce methane), then the digestate could result in lower $N_2O$ emissions compared to undigested slurry. A system based analysis of the use of digestate on agricultural land is required.

**7.** We know that $N_2O$ emissions are controlled by temperature, soil anearobicity (water content) and availability of nitrogen. More tactical application timings of fertiliser N could result in reduced $N_2O$ emissions. Further research is required to optimise the timing of fertiliser N applications to reduce $N_2O$ emissions, whilst maintaining acceptable agronomic yields.

### 4.1.4 Ammonia

Significant advances have been made in recent years in improving our understanding and ability to model ammonia emissions from agricultural sources. However, a number of knowledge gaps within the context of dairy production remain, requiring further research effort:

**1.** Lack of data regarding current feeding practices (as mentioned under GHG above) makes it difficult to estimate nitrogen excretion rates of livestock and, in particular, the form of nitrogen excreted, which are key input requirements for ammonia emission modelling. There is a growing body of literature and ongoing research pertaining to the relationship between diet and nitrogen excretion, but the potential impact of dietary manipulation methods on ammonia emissions from the dairy sector is difficult to assess without good prior knowledge of current feeding practices. Of course, the difficulties of collating robust N intake information or influencing dietary N intake with largely forage-based diets should not be underestimated.

**2.** Despite a large database of experimental measurements, predictions of ammonia emissions following slurry application to land are still associated with high uncertainty. While our predictive capability has improved by establishing relationships between emission and slurry, soil, crop, meteorological and management factors, a key uncertainty which has not yet been well characterised is the infiltration rate of applied slurry into the soil. Limited research has demonstrated the importance of slurry dry matter content and soil moisture content on the rate and extent of infiltration, but further research is required in order to develop robust relationships which can be applied within decision support systems and predictive tools.

**3.** Nitrogen losses and transformations within solid manure management systems are subject to large uncertainty, particularly losses of nitrogen as dinitrogen through the denitrification process. The nature and extent of these losses and transformations will influence the emissions of ammonia at the housing, storage and land spreading stages of solid manure management, and further research is required to better characterise these. This will also improve our knowledge of the factors influencing nitrous oxide emissions from solid manure systems.

**4.** A robust emission factor for slurry lagoons, expressed as a percentage of ammoniacal nitrogen available in the lagoon is required. The current emission factor is a factor of ten greater than that for above ground tanks, but based on very few measurements.

**5.** Bringing together the ammonia and nitrous oxide emission (and methane) inventories into a single model to ensure consistency in underlying data and a commonality in the approach taken to nitrogen flows through the system will improve the accuracy and robustness of both inventories. This is one of the objectives of current Defra funded project AC0112.

**6.** Robust farm management activity data across the range of practices employed and uptake of mitigation methods are required for a robust national assessment of ammonia emissions from the dairy sector. Running the annual Farm Practices Survey (since 2004) has greatly improved the quantity and quality of data available (for England, at least), but improvements in our modelling abilities require improvements in the type and resolution of activity data collated.