SERPECcC

Sectoral Emission Reduction Potentials and Economic Costs for Climate Change (SERPEC-CC)

Agriculture: methane and nitrous oxide

October 2009

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AEA Energy & Environment

Financial support from the DGRTD (European Community Sixth Framework Programme) and DGENV of the European Commission as well as of the Dutch and German ministries of Environment (VROM and BMU) is acknowledged. The SERPEC paper reflects the opinion of the authors and does not necessarily reflect the opinion of the European Commission, VROM and BMU on the results obtained.

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Executive Summary

The aim of the project Sectoral Emission Reduction Potentials and Economic Costs for Climate Change (SERPEC-CC) is to identify the potential and costs of technical control options to reduce greenhouse gas emissions across all European Unions sectors and Member States in 2020 and 2030.

In this SERPEC sectoral report, we determine the abatement potential of non-CO_2 greenhouse gases from the agricultural sector across Europe. Over the period 1990 to 2005, emissions from this sector in the EU27 fell by around 15% and in 2005 represented approximately 10% to the overall greenhouse gas (GHG) emissions in the EU27. Within the sector, agricultural soils, enteric fermentation and manure management are the main emissions sources. Projections show European agricultural emissions declining through to 2015, after which they remain relatively stable out to 2030. Against projected baseline emissions (see Figure 1), we demonstrate that the maximum technically available potential in 2020 is 160 Mt CO_2-eq/year, which would represent a 35% reduction against European agricultural sector emissions in 2005 (see Figure 1)

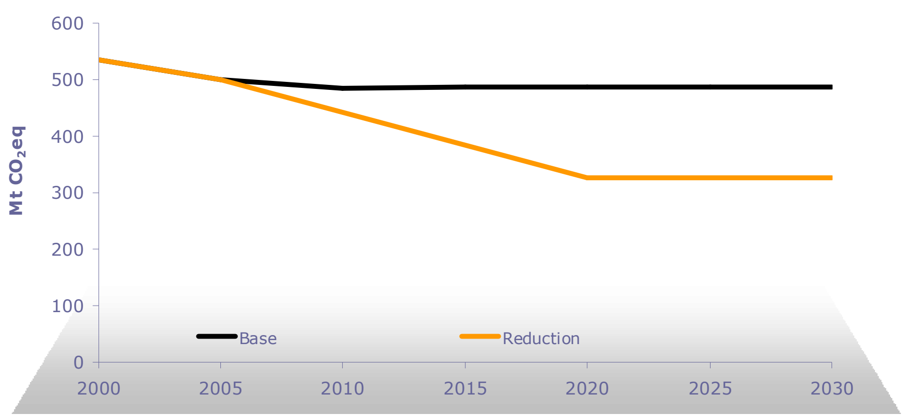

Figure 1: Historical agricultural emissions (1990-2005), business as usual baseline emissions and maximum identified abatement potential.

For the year 2020, 47 Mt CO_2-eq/year of emissions reductions was identified at zero or negative cost (Figure 2) compared to baseline emissions. This amounts to a reduction of 10% against 2005 levels which is consistent with the EU target for emission reduction across the non-traded sector of the European economy. The set of most cost-effective measures (<20 €/tCO_2-eq) is shown in Table 1 and includes: precision farming (reducing N_2O from soils), centralized anaerobic digestion of manure (reducing N_2O and CH_4 emission that would otherwise occur from manure storage or application), improvement of lifetime and efficiency of livestock (long term management and use of genetic resources that reduces enteric CH_4 emissions) and addition of nitrification inhibitors to soils (reducing N_2O emissions from soils). The overall cost-abatement curve for European agriculture in the year 2020 is shown below in Figure 2.

Finally, we point out that the abatement identified in this study is 'technical' abatement and in order to deliver the maximum proportion of this potential, a strong policy and regulatory framework should be established.

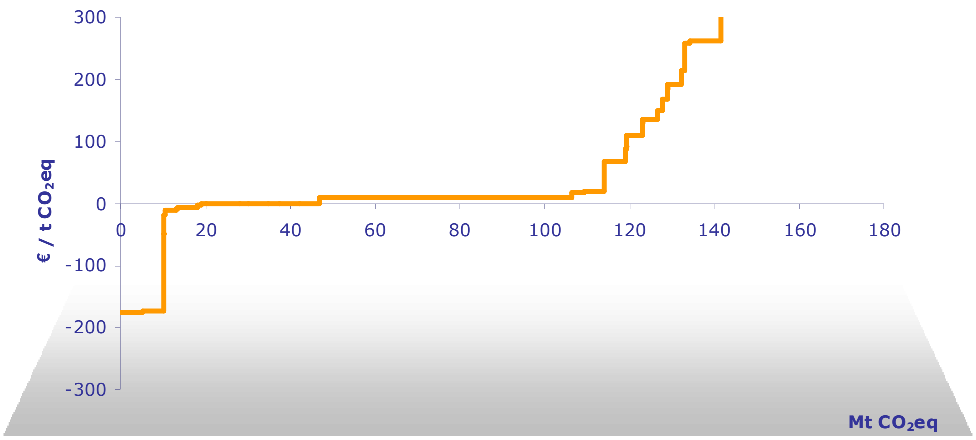

Figure 2: Abatement potential and specific costs of abatement options in the agriculture sector in the EU27 in 2020 (note, approximately 15 Mt of reductions with costs >300 €/tCO_2 are not shown).

Table 1: Agriculture abatement measures, specific costs, abatement potentials and cumulative abatement potential.

	Measures	**Specific costs (€/tCO_2)**	**Mton $CO_{2\text{-eq}}$/ year)**	Cum. Mt CO2-eq year
Soils	Reduce N application through precisison farming	-175	5	5
Soils	Reduced N application through improved spreader maintenance	-173	5	10
Manures	Centralized anaerobic digestion: dairy - West (warm)	-48	0	10
Manures	Centralized anaerobic digestion: dairy - East (temperate)	-19	0	10
Manures	Centralized anaerobic digestion: dairy - West (temperate)	-10	3	13
Manures	Centralized anaerobic digestion: pigs - East (temperate)	-8	0	14
Manures	Centralized anaerobic digestion: pigs - West (temperate)	-7	5	18
Manures	Centralized anaerobic digestion: pigs - West (warm)	-3	1	19
Soils	Reduce N application through fertiliser free zone	-1	1	20
Enteric	Long term management and use of genetic resources (Non-dairy cattle Eastern Europe)	0	5	25
Enteric	Long term management and use of genetic resources (Dairy cattle Eastern Europe)	0	2	27
Enteric	Long term management and use of genetic resources (Dairy cattle Western Europe)	0	3	30
Manures	Reducing the rate of microbial action	0	5	35
Manures	Removal of the gas source	0	8	43
Enteric	Long term management and use of genetic resources (Non-dairy cattle Western Europe)	0	5	48
Soils	Addition of Nitrification inhibitors - Mineral	10	30	78
Soils	Addition of Nitrification inhibitors - Manures	10	29	107
Soils	Reduced grazing on wet areas	18	3	110
Soils	Reduce N application through enhanced distribution geometry	20	5	115
Soils	Reduce N application through allowance for manure/residual N	67	5	120
Manures	On farm anaerobic digestion: dairy - West (warm)	77	0	120
Manures	On farm anaerobic digestion: dairy - East (temperate)	88	0	120
Manures	Centralized anaerobic digestion: dairy - East (warm)	93	0	120
Manures	On farm anaerobic digestion: for pigs - West (warm)	111	4	124
Manures	Centralized anaerobic digestion: pigs - East (warm)	130	0	124
Enteric	Adding oils and oilseeds (Dairy cattle Western Europe)	137	4	128
Manures	On farm anaerobic digestion: dairy - West (temperate)	150	1	129
Enteric	Adding oils and oilseeds (Dairy cattle Eastern Europe)	168	1	130
Manures	On farm anaerobic digestion: pigs - East (warm)	183	0	130
Manures	On farm anaerobic digestion: dairy - East (warm)	186	0	130
Manures	On farm anaerobic digestion: pigs - West (temperate)	193	3	133
Manures	On farm anaerobic digestion: pigs - East (temperate)	214	1	134
Enteric	Adding oils and oilseeds (Non-dairy cattle Eastern Europe)	258	1	135
Enteric	Adding oils and oilseeds (Non-dairy cattle Western Europe)	262	7	143
Enteric	Replacement of roughage with concentrates (Dairy cattle Western Europe)	1,222	3	145
Enteric	Replacement of roughage with concentrates (Dairy cattle Eastern Europe)	1,497	1	146
Enteric	Replacement of roughage with concentrates (Non-dairy cattle Eastern Europe)	2,297	1	147
Enteric	Replacement of roughage with concentrates (Non-dairy cattle Western Europe)	2,338	5	152
Soils	Better livestock nutrient use efficiency - grazing	2,624	3	155
Soils	Better livestock nutrient use efficiency - fertiliser	3,432	6	161

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1 Introduction

1.1 The SERPEC project, agriculture sector

The SERPEC project

The aim of the project Sectoral Emission Reduction Potentials and Economic Costs for Climate Change (SERPEC) is to identify the potentials and costs of technical control options to reduce greenhouse gas emissions across all European Union sectors and Member States in 2020 and 2030. The results are presented in so-called marginal abatement cost curves (MACCs) that provide a least-cost ranking of options across technologies and sectors in the EU. In general, MACCs provide strategic information for policy makers.

All identified abatement options refer to technologies that are applied already today, or will become commercially viable in the near future. To identify their abatement potentials we estimated the maximum feasible implementation rates, often governed by the rate of turnover of existing technology stocks. Costs of mature technologies were assumed constant over time, whereas costs of relatively new technologies, e.g. wind turbines, were allowed to decrease over time due to economies of scale and technology learning.

The agriculture sector

In this SERPEC sectoral report, we determine the potentials and costs of control options in the agriculture sector. This sector contributed approximately 10% to the overall greenhouse gas (GHG) emissions in the EU in 2005. Within the sector, agricultural soils, enteric fermentation and manure management are the main emissions sources.

This report examines abatement options for emissions of non-CO_2 greenhouse gas emissions from agriculture. The three areas of standard agriculture that give rise to non-CO_2 greenhouse gases are: agricultural soils, enteric fermentation and manure management.

We used the most recent Global Warming Potentials (GWP) from the fourth IPPC assessment report to calculate the CO_2-equivalent emissions in the baseline and of the abatement measures. These values, 25 for methane and 298 for nitrous oxide, are slightly different from the older GWP values (21 respectively 310) on which some of the literature to which this report refers is based.

1.2 Historic emissions

1.2.1 Soils

Emission mechanisms

Agricultural N_2O emissions derive from three principal sources (IPCC, 2006):

- direct emissions from soil nitrogen e.g. applied fertilisers (both manures and artificial), excreta voided by livestock while grazing, the mineralisation of organic soil organic matter and crop residues;
- emissions from livestock manures in store;
- indirect emissions from nitrogen lost to the agricultural system e.g. through leaching, runoff or atmospheric deposition.

Nitrous oxide is produced by the processes of denitrification and nitrification. Denitrification is the microbial reduction of nitrate or nitrite to dinitrogen or N-oxides and occurs in anaerobic, flooded soils, and in anaerobic microsites in otherwise aerated soils.

Nitrous oxide is readily soluble, and surface water draining from agricultural fields contains dissolved N_2O, which is later denitrified or lost to the atmosphere. Leached nitrate can be denitrified in ground or surface waters to provide a source of N_2O as large as direct emissions from soils (Bouwman, 1990).

Nitrification is the oxidation of ammonium to nitrite or nitrate by the bacteria *Nitrosomonas* and *Nitrobacter*.

The principal environmental parameters affecting N_2O emissions are the availability of a nitrogen source, moisture and temperature, with nitrogen availability being the most important (Colbourn, 1993). Microbial diversity may also influence temporal and spatial variability in the processes, which govern N_2O emission rates. Agricultural management has a major influence on nitrogen availability and environmental conditions, through, for example, fertiliser applications, livestock waste handling, residue management or operations affecting the structure, aeration and pH of soils.

At present the recommended methodology for estimating N_2O emissions from soils (IPCC, 2006) assumes that 1% of the nitrogen added to soil as mineral fertilisers, manures, crop residues etc., is released directly as N_2O, with further N_2O emissions arising indirectly from volatilisation and subsequent deposition of NH_3 and NO_x from the application of manures and fertilisers. The IPCC

(2006) methodology also assumes that 2% of the nitrogen voided directly to pastures by cattle is directly emitted as N_2O, with 1% of the N deposited by sheep. In addition, 0.75% of the N in nitrate lost by leaching is considered to be emitted indirectly as N_2O from the denitrification of the nitrate in slow-moving waters. The uncertainty surrounding this estimate is high meaning that estimates of emissions have a relatively high level of uncertainty.

There is a considerable amount of ongoing research into improving understanding of emissions mechanisms, and examining the impact that different agricultural practices might have on N_2O emissions at the field and farm scale. Recent data suggests that emissions do vary with both environmental factors (climate, soil organic C content, soil texture, drainage and soil pH); and (2) management-related factors (N application rate per fertiliser type, type of crop, with differences between legumes, non-leguminous arable crops, and grass). Establishing such relationships, as well as improving emissions estimates, would also allow the impact of mitigation measures to be established with more certainty.

Emission in EU-27 Croatia and Turkey

Figure 3 shows how emissions of N_2O from agricultural soils have varied since 1990. The general trend has been one of falling emissions, which reflects the reduction in nitrogenous fertiliser use. N_2O emissions from soils in the EU27 were 305 Tg CO_2 eq. in 1990 falling to 238 Tg CO_2 eq. in 2006. Dissagregating these numbers, the largest emissions of N_2O from soils in 2006 arose from France (47 Tg CO_2 eq.), followed by Germany (38 Tg CO_2 eq.) and then the UK (24 Tg CO_2 eq.). Emissions from Croatia were consistently between 1.9 and 2.6 Tg CO_2 eq. over the period 1990 to 2006, whilst no emissions of N_2O from soils were reported over this period for Turkey.

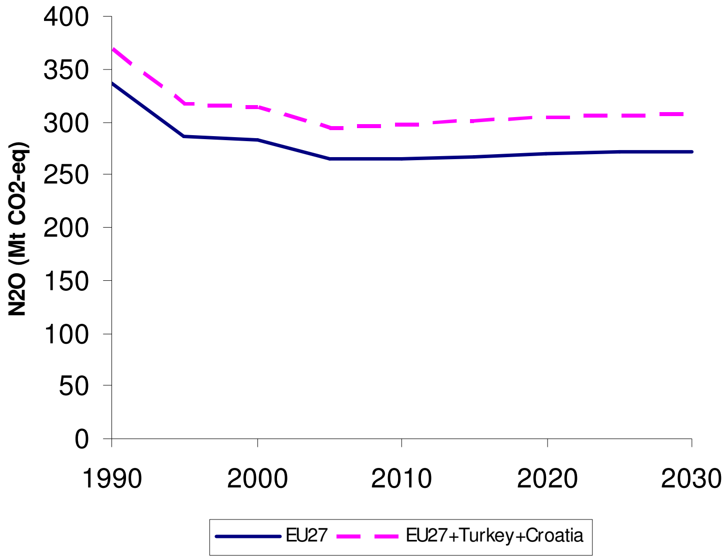

Figure 3: Time series of N_2O emissions from agricultural soils in the EU 27 MS, Croatia and Turkey (1990 and 2000 emissions for Turkey were assumed similar to 2000 emissions).

1.2.2 Enteric fermentation

Emission mechanisms

Enteric fermentation is the anaerobic fermentation of cellulose and other .components of animal feeds in the gut of ruminant animals (the rumen), e.g. cattle and sheep, by micro-organisms. The ruminant diet typically consists of fresh grass and silage together with prepared animal feed, although some in-door beef rearing systems are almost entirely cereal and maize silage based.

The stomachs of ruminants contain four divisions, the rumen, reticulum, omasum and the abomasum. When food is eaten it first enters the rumen. Here it is retained and anaerobically fermented by the rumen's large and diverse microbial population. These microbial organisms break down the compounds in food to produce volatile fatty acids (VFA), carbon dioxide, methane, cell material, heat and ammonia. The VFAs pass through the rumen wall into the circulatory system and are oxidised in the liver, supplying a major part of the energy needs (60 to 70%) of the animal; they may also be directly utilised as building blocks for synthesis of cell material. The carbon dioxide and methane are mainly removed by eruction (through the mouth or gut of the animal) with a small proportion of methane absorbed in the blood and eliminated through the lungs. Fermentation is also coupled to microbial growth and the microbial cell protein synthesised forms the major source of protein, rumen digestible protein, for the animal. Since microbial organisms provide such an important source of protein, the aim of most commercial feeding systems is to maximise microbial growth within the rumen while also providing good sources of rumen undegradeable protein which is absorbed by the abomasum.

Enteric emissions depend on the average daily feed intake and the percentage of this feed energy which is converted to methane. Average daily feed intake for any particular livestock type can vary considerably and is related to, amongst other things, the weight of the animal (and the energy required to maintain it), its rate of weight gain, and for dairy cows, the rate of milk production. Methane conversion efficiency depends on rumen efficiency and the quality (digestibility and energy value) of the feed. Rumen efficiency depends on the diversity, size and activity of the microbial population in the rumen, which are largely determined by diet. Between 4-10% of the energy in feed is lost through conversion to methane and is not available for digestion.

Emissions in the EEA MS

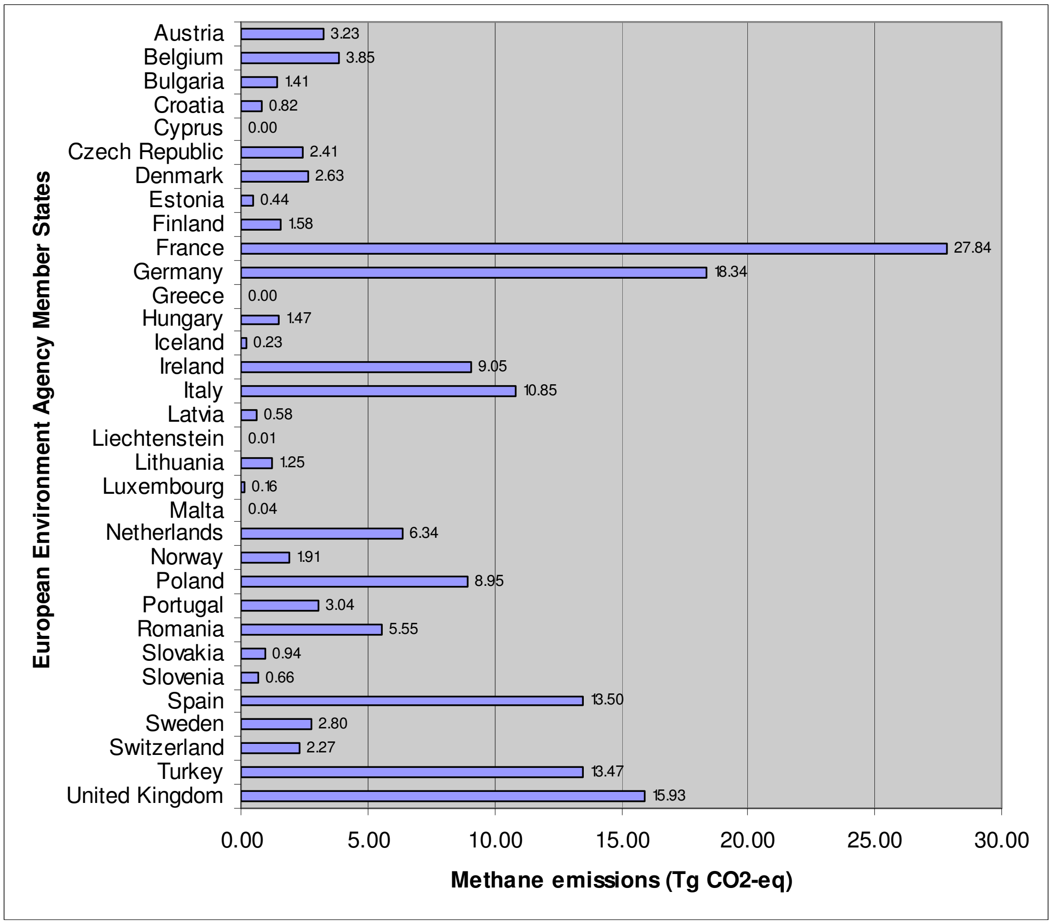

Figure 4: Emissions of methane from enteric fermentation in EEA Member States for 2005 (exceptions: for Turkey 2004 values are presented; for Malta 2000 values are presented)[1].

The leading Member States for enteric CH_4 emissions are France, Germany, UK and Spain (Figure 4). For the EU27 as a whole enteric CH_4 emissions were 145 Tg CO_2 eq. in 2005, slightly lower than N_2O emissions from soil (240 Tg CO_2-eq in 2005). In comparison, emissions arising in Croatia were 0.8 Tg CO_2 eq., whilst emissions from Turkey were 13.5 Tg CO_2 eq. in 2005. The relative magnitudes of methane emissions from the different member states shown in Figure 4 reflect the head of ruminant livestock in each nation.

1 UNFCCC, accessed March 2008. See: http://unfccc.int/di/FlexibleQueries/Setup.do

1.2.3 Manures

Emission mechanisms

Animal manures contain relatively complex organic compounds such as carbohydrates and proteins that are broken down naturally by bacteria. In the presence of oxygen, the action of aerobic bacteria results in the carbon being converted to CO_2 and, in the absence of oxygen, anaerobic bacteria transform the carbon to CH_4. Whereas the CO_2 that is evolved is part of the natural cycling of carbon in the environment and results in no overall increase in atmospheric CO_2, the carbon released as methane, has a higher global warming potential.

When livestock are in fields and their manure ends up being spread thinly on the ground, aerobic decomposition usually predominates. However with modern intensive livestock practices, where animals are often housed or kept in confined spaces for at least part of the year, manure concentrations will be higher and manure will often be stored in tanks or lagoons where anaerobic conditions generally predominate and methane will be evolved.

In summary, methane emissions from manure depend on:

- The quantity of manure produced, which depends on number of animals, feed intake and digestibility;
- The methane producing potential of the manure which varies by animal type and the quality of the feed consumed;
- The way the manure is managed (e.g. whether it is stored as a liquid or spread as a solid);
- The climate: the warmer the climate the more biological activity takes place and the greater the potential for methane evolution. Also, where precipitation causes high soil moisture contents, air is excluded from soil pores and the soils become anaerobic increasing the potential for methane release (for wastes which have been spread).

Animal manures also contain nitrogen in the form of various complex compounds, and if applied to land, then this nitrogen enters the nitrogen cycle, as various bacteria in the soil break down these nitrogen containing compounds. N_2O production only takes place under specific conditions since it results from combined aerobic and anaerobic processes: nitrification, the transformation of ammonium to nitrate (aerobic); denitrification, the formation of nitrogen gas from nitrate reduction (anaerobic). Consequently, N_2O emission is influenced by oxygen status, temperature, moisture content and antecedent soil conditions. Typically, conditions in manure are strictly anaerobic and, so, both nitrification and denitrification will occur. However, under certain conditions such as under forced and controlled aeration of liquid or solid manure, denitrification occurs after aeration (Monteny et al., 2006). Alternatively, passive aeration may occur in manures to which straw or bedding is introduced, giving rise to nitrification and denitrification conditions.

Nitrous oxide has an even higher global warming potential than methane. Therefore, any measures to reduce methane releases to atmosphere resulting from animal manures, should attempt to avoid creating conditions that give rise to greater nitrous oxide release.

Emissions across EEA MS

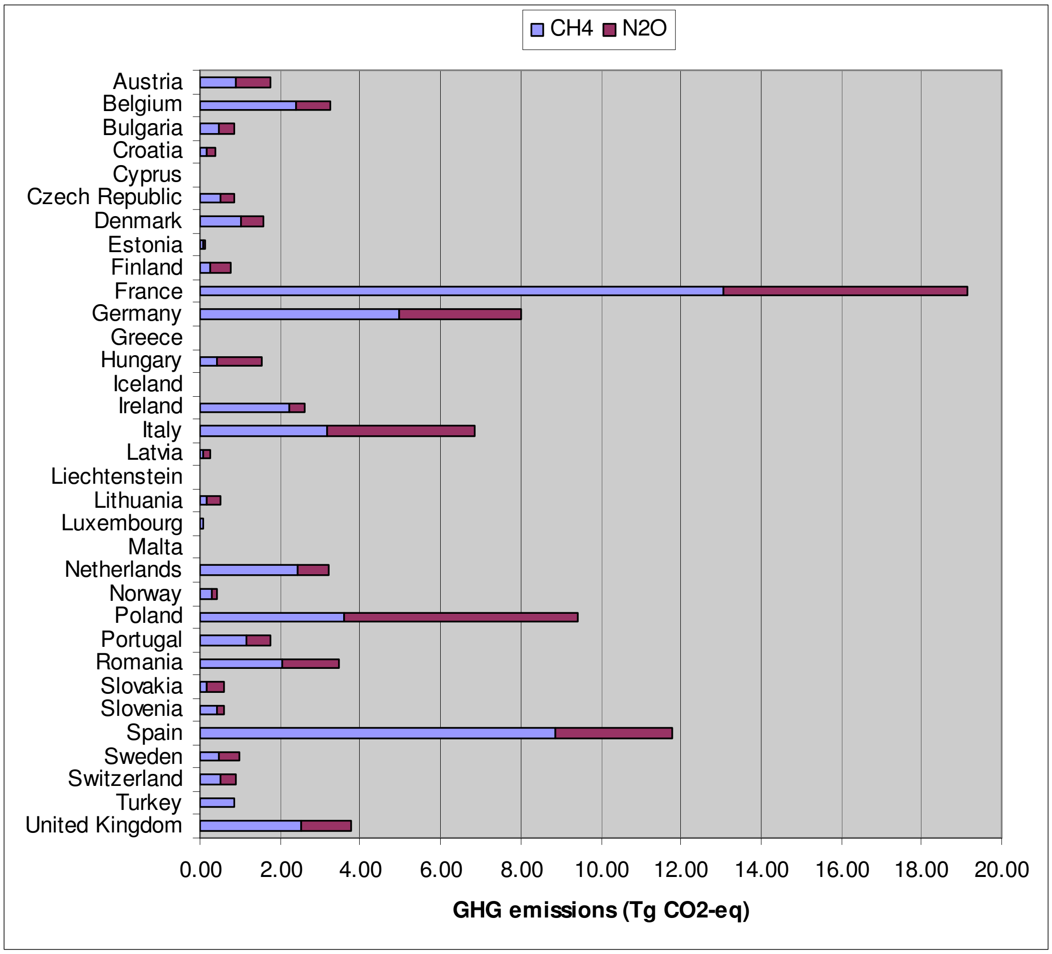

Figure 5: Greenhouse gas emissions from manure management in 2005 for EEA Member States (exceptions: for Turkey 2004 values are presented; for Malta 2000 values are presented)[2].

2 UNFCCC, accessed March 2008. See: http://unfccc.int/di/FlexibleQueries/Setup.do

CH_4 and N_2O releases from manures can be significant although the magnitude of the emissions is variable (IPCC, 2007). Figure 5 illustrates the variation in CO_2 equivalent greenhouse gas emissions from EEA members' states. As is the case for enteric fermentation emissions, France dominates GHG releases from manure management, however here Poland, Spain, Italy and Germany are all important emitters. N_2O is a significant emission from livestock manures, accounting for around 40% of the total CO_2 equivalent emissions from all EEA MS. GHG emissions from manure management total 86 Tg CO_2-eq across the EEA, slightly over 50% of those from enteric fermentation. Across the EU 27, GHG emissions of 85 Tg CO_2 eq. arose from manure management in 2005, whilst in Croatia and Turkey emissions were 0.4 and 0.84 Tg CO_2 eq., respectively.

1.3 Baseline emissions

Historic emissions for the agriculture sector are taken from UNFCCC reporting. Projections of emissions from agriculture are made using RAINS model outputs for livestock numbers per MS and consumption of nitrogenous fertilisers. This data is used in conjunction with standard IPCC emissions modelling parameters and emissions factors. These calculations yield emissions from agricultural soils, enteric fermentation and manure management for the years 2005 to 2030 in five yearly intervals, which form the baseline, or business as usual projection against which abatement is compared.

Projections of N_2O emissions from agricultural soils include the impacts of:

- Direct and indirect emissions from mineral and manure fertiliser application. The two fertiliser types together represent the principal driver of N_2O emissions from soils and are taken from the following sources:
    - Mineral fertiliser application taken directly from IIASA-RAINS modelling.
    - Manure fertilser application calculated based on animal numbers calculated in IIASA-RAINS modelling, IPCC parameters for excretion rates of livestock and proportion of manure applied to land as fertilser.
- Emission from excretion of manure by grazing animals.
- Removal of N by N fixing crops and that stored in crop residues.
- Emissions associated with animal manure management.

Projections of CH_4 emissions include emissions from enteric fermentation and manure management of:

- cattle (dairy and non-dairy), sheep, goats, horses, swine, poultry and other livestock

CH_4 emission projections are calculated by scaling 2005 emissions by the change in livestock numbers over the period to the projected year.

Baseline emissions estimate for the total agriculture sector are shown in Figure 6. The trend is generally downwards to 2010 and then stabilises. The baseline established in this study is fully comparable with the baseline that IIASA established as input to the European Commissions 2008 Climate package (IIASSA, 2008a,b) and is based on Member States national agricultural projections provided for the revision of the NEC Directive. For those countries that did not provide a national projection, the baseline assumes the default the agricultural development as outlined by the CAPRI and European Fertiliser Manufacturers Association (EFMA) agricultural and fertiliser projections. Where these are unavailable, projections developed by the Food and Agriculture Organisation are used.

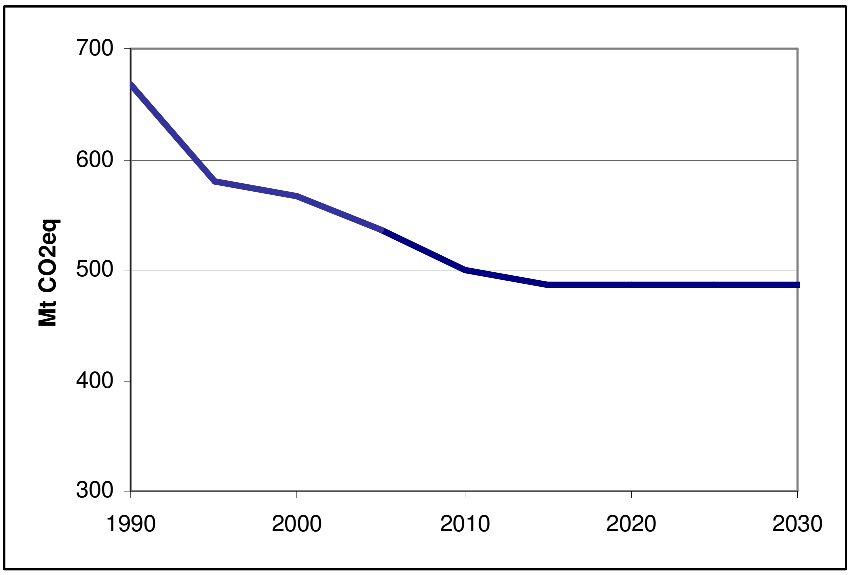

Figure 6: Baseline GHG emissions from the agricultural sector.

1.4 Abatement measures

For each of the emission sources, we identified measures to further reduce emissions relative to the base line shown in Figure 6.

- Chapter 2 considers the measures available for abating N_2O emissions from cultivated soils, presents the abatement potential and cost assumptions for calculating the cost effectiveness of each measure.
- Chapter 3 considers the options for abating CH_4 emissions from enteric fermentation along with the abatement and cost assumptions for each.
- Chapter 4 considers abatement measures for CH_4 and N_2O emissions from manure management.
- Chapter 5 presents cost curves for the years 2020 and 2030 as well as discussing where the measures presented in sections 3 to 5 sit within those curves.

Data used to generate abatement measure characteristics is taken largely from public sources such as Eurostat or Faostat unless otherwise specified. We do not take into account detailed country specific values such as the number of days dairy cows are housed nor changes in milk yields unless these measures are included in modelling used to generate cost estimates for example.

Nonetheless, the implementation level of the measures was calculated separately to the best of our ability for each Member State, taking into account the available national constraints on application of each measure according to their characteristics. For measures targeting:

- Emissions of N2O from agricultural soils, implementation levels were assumed to be 100% in 2020 and 2030 for all but reduced grazing on wet areas, for which implementation was calculated based on cattle numbers, agricultural area and annual average rainfall rate.
- Emissions of CH4 from enteric fermentation, implementation was assumed to be 100% in 2020 and 2030 as measures were considered applicable to all relevant head of livestock.
- Emissions from manure management, implementation was calculated according to national characteristics such as livestock density, proportion of cattle on liquid manure management systems.

Measures refer either to further implementation of technologies that are already partly included in the baseline, or application of new abatement techniques that are available on the market today. For all, the implementation proportion reflects the technical potential for application.

1.5 Abatement costs

The general method to calculated specific abatement costs is explained in the *textbox* below.

The specific costs of measures in € per tonne CO_2-eq abated

Abatement costs of measures are calculated from the sum of annualised investment costs and annual operating and maintenance costs (these could have negative –savings- numbers) divided by mean annual emission savings of the measures:

$$\text{specific costs} = \frac{\text{annualised capital costs} + \text{annual O\&M}}{\text{Annual abated CO2 - eq. emissions}}$$

Capital costs are annualised over the technical lifetime of the measure using a discount rate of 4%. This value is similar to government bond rates. The annual operation and maintenance costs are assumed to remain fixed over the depreciation period.

The costs refer to the *extra* costs compared to the reference situation. In the case of so-called retrofit measures, the extra costs are the same as the overall costs of the measures, because the reference is 'not taking the measure'. In the case of new stock, e.g. a new car or new production capacity, the extra costs are a case specific estimate.

The overall costs calculation is also referred to as 'social costs'. The method allows for comparison of the 'bare' costs of technologies, across measures, sectors and countries. A negative cost number indicates that from a social perspective there will be a net welfare gain from taking these measures; a positive cost number indicates a net welfare loss.

Note, that the so-called 'end-user' perceives higher energy prices and discount rates. As a result the cost-curve from an end-users perspective looks different.

The agricultural baseline used here includes the effects of policies such as IPPC and the Nitrates Directive. Therefore, it is reasonable to consider that the costs of all measures identified are attributed to greenhouse gas reduction.

2 Options to reduce N_2O emissions from soils

Introduction to the measures

In general terms, N_2O emissions from the application of both mineral N fertilisers, and organic N in animal manures can be decreased, by management practices (Mosier et al., 1998) that:

- optimise the crop's natural ability to compete with processes whereby N available to the plant is lost from the soil-plant system (e.g. by volatilisation as NH_3, denitrification and leaching);
- directly lower the rate and duration of the processes by which N is lost as N_2O.

A variety of strategies can be used to an increase the overall efficiency with which N is used by crops. These are mainly based around accurately controlling the supply of N and matching it to crop demand. This can be done by synchronising N supply with plant demand by maintaining low levels of mineral N in the soil when there is little or no plant growth and providing sufficient N to meet plant requirements during periods of rapid development. Multiple rather than single applications of fertiliser N can also be a more efficient way of supplying N to crops, although this may not always be practical. Soil mineral N can also be managed during fallow periods, e.g. by using interseasonal cover crops to reduce nitrate production.

Preventing the formation of N_2O can be done by

- Improving the aeration of soil, for example increasing drainage and preventing soil compaction.
- Use of N-transformation inhibitors such as nitrification inhibitors. These can be used both for arable and grazing land.
- Rewetting of farmed organic soils (peatlands), or the maintenance of a shallow water table in combination with a ban on root crops. This can reduce both N_2O and CO_2 emissions. The drainage and cultivation of peatlands allow oxygen to enter the soil, enhancing soil organic matter decomposition, which delivers nutrients to the crop and is the source of N_2O and CO_2 emissions. Emissions from farmed organic soils are much higher than from farmed mineral soils and tackling emissions from this source could therefore be a very effective way of reducing emissions.

The impact of techniques which reduce total N inputs to crops are readily captured even by the IPCC Tier 1 methodology and so will be reflected in national totals.

Options which could reduce N_2O emissions from the animal wastes from grazing animals and the application of animal wastes from housed livestock to land are also discussed here. Options to reduce emissions from storage of manures are discussed in the section on manure management as such techniques can also impact on CH_4 emissions from manure management.

2.1 Improvements in fertiliser practice

The use of farm management techniques to reduce nitrogen losses from the agricultural system through improved fertiliser efficiency has been evident for some time, as a result of efforts to maximise crop productivity and the profitability of farm enterprises (rather than specifically reduce N_2O emissions), and also to reduce nitrate leaching to waters. The succession of CAP reforms which have moved away from direct price support to crops has encouraged this trend and in some countries there are mandatory limits on fertiliser-N application in order to comply with the Nitrates Directive.

A number of options that could be used to maximise fertiliser use efficiency are discussed below. These are intended to provide examples of the types of measures available to farmers. An additional benefit of measures which are designed to match N supply to crop N may be a reduction in the amount of nitrate leaching. Similarly, measures introduced specifically to reduce nitrate leaching and pollution (e.g. under the Nitrates Directive) are likely to help to reduce N_2O emissions.

The underlying principle for improving fertiliser practice is to reduce the risk of exceeding crop N requirements. There are many ways to reduce this risk, mainly through monitoring manure and fertiliser application and comparing this to crop nutrient requirement. This principle is applicable to all of the "improvements in fertiliser practice" measures. Options identified in the previous Sectoral Objectives Study (Bates et al. 2001) and subsequent studies (e.g. Moorby et al., 2007; Monteny et al., 2006) include:

- making effective allowance for manure N and residual N
- improved maintenance of fertiliser spreaders
- maintaining fertiliser free zones on the edge of fields
- optimisation of fertiliser distribution geometry
- precision in fertiliser application

There are many recognised fertiliser recommendation systems (eg, RB209 (MAFF, 2000), PLANET (Planning Land Applications of Nutrients for Efficiency and the Environment)[3] and other supplementary guidance (e.g. on precision farming, canopy management techniques) to plan fertiliser applications to all crops. This guidance takes into account crop requirements; soil nutrient requirements and supply; the amount of nutrients supplied by applied manure both recent

[3] www.planet4farmers.co.uk

and from previous seasons and soil pH. All of these when taken into account can reduce the risk of over application of N fertiliser.

When applying fertiliser, farmers should take into account any manure that has been recently applied to the soil. Manure analysis should be carried out to assess the nutrient content. Taking the nutrient content of manure into account when applying mineral fertilisers should reduce the amount of fertiliser required. Accurate records should also be kept of manure and mineral fertiliser applications on individual fields. This can reduce the risk of applying slurry too soon after mineral fertiliser which increases denitrification and therefore N_2O emissions. A reduction in N_2O emissions of approximately 5% has been seen from making full allowance for manure (Cuttle et al., 2007).

Whether any reduction in N_2O emissions from adopting these measures will be reflected in national emission inventories will depend upon whether they lead to reductions in fertilizer-N use and hence may be recognised when using a Tier 1 methodology to calculate national emissions. Such reductions are likely to accrue from making effective allowance for manure N and residual N and maintaining fertiliser free zones on the edge of fields. However, techniques such as optimisation of fertiliser distribution geometry will less obviously lead to reductions in fertilizer-N use.

2.1.1Cost and abatement information for fertiliser spreading

Abatement potential for improved fertiliser spreading is taken as an improvement over baseline mineral fertiliser emission rates of:

- 5% for improved spreader maintenance (Anon, 2005a)
- 100% for fertiliser free zones
- 5% for improved spreader distribution geometry (Anon, 2005a)

Increased efficiency of fertiliser use has been analysed previously in Bates (2001) based on calculations developed in Worrell (1994). Because there has been little development of this work, the cost information used for these measures here updates the information generated previously, converting it to $€_{2005}$ prices and accounting for changes in the price of fertiliser.

The possible increase in organic farming (and concomitant decrease in mineral fertiliser use) in the future is only taken into account to the extent that the RAINS model projections for mineral fertiliser application take this effect into account.

The main assumptions originally used to derive costs in Worrell (1994) of the options are shown in Table 2. A discount rate of 10% was also used in that study.

Table 2 Main assumptions for options.

Spreader maintenance	Good maintenance saves approx. 50 kg N/ha on current average levels (of 228 kg N/ha); annual maintenance costs increase by 64 € per spreader; uniform spreading increases crop production by 26 €/ha/year for grass and 86-300 €/ha/year for other crops.
Fertiliser free zone	It is assumed that a 50 cm unfertilised strip will save 4% on fertiliser use but will lead to a yield reduction equivalent to 11-15 €/ha/year for grass and 9 –13 €/ha/year for other crops.
Fertiliser distribution geometry	Adjustment of spreader costs 120 € per spreader and spreader has a lifetime of 10 years; fertiliser losses to ditches are 50 kg N/km per year.

The cost-effectiveness is shown in Table 3. All costs are in $€_{2005}$ and have been adjusted from Bates (2001) to reflect current fertiliser prices of 669 € per tonne of N.

Regional dissagregation in overall cost was calculated according to the current area of the different crop types in each MS.

It is assumed feasible to implement improved spreader maintenance and spreader distribution geometry to 100% of mineral fertiliser application. Fertiliser free zones are assumed to be implemented to a level that avoids 1% of fertiliser use. There is current estimate of fertiliser free practice and the implementation of the measure would likely need to be agreed by the EU or MS.

Table 3: Cost information for fertiliser efficiency measures.

Measure	Crop type	N fert use	Potential savings		Cost saving through reduced fertiliser use	Investment cost	O & M costs	Loss of income from yield reduction	Total costs per t Nf saved	Total costs per t Nf applied	Reduction already achieved	Fertiliser use reduction already achieved	Total costs per t Nf saved, adjusted by reduction already achieved	Total costs per t Nf applied
		kg Nf/ha	kg Nf/ha	%	€/t Nf	€/t Nf	€/t Nf	€/t Nf	€/t Nf/year	€/t Nf/year	kg Nf/ha	%	€/t Nf/year	€/t Nf/year
Spread Maintenance	Grass	314	47	15%	669	-	87	-	-582	-103	-	0%	- 582	-103
	Maize	97	15	15%	669	-	87	-	-582	-103	-	0%	-582	-103
	Potato	196	14	7%	669	-	87	-	-582	-44	-	0%	-582	-44
	Sugar Beet	148	5	4%	669	-	87	-	-582	-24	-	0%	-582	-24
	Wheat	191	7	4%	669	-	87	-	-582	-24	-	0%	-582	-24
	Barley	95	8	8%	669	-	87	-	-582	-51	-	0%	-582	-51
Fertiliser Free Zone	Grass	314	2	1%	669	-	-	57	-612	-6	47	15%	-602	-7
	Maize	97	1	1%	669	-	-	90	-579	-6	15	15%	-562	-7
	Potato	196	1	1%	669	-	-	86	-583	-6	14	7%	-576	-6
	Sugar Beet	148	-	0%	669	-	-	62	-607	-	5	3%	-605	-
	Wheat	191	1	0%	669	-	-	47	-622	-	7	4%	-620	-
	Barley	95	1	1%	669	-	-	115	-554	-6	8	8%	-543	-6
Distribution Geometry	Grass	314	10	3%	669	813	-	-	144	4	49	16%	295	11
	Maize	97	3	3%	669	813	-	-	144	4	16	16%	305	11
	Potato	196	5	3%	669	813	-	-	144	4	15	8%	212	7
	Sugar Beet	148	2	1%	669	813	-	-	144	1	5	3%	173	2
	Wheat	191	3	1%	669	813	-	-	144	1	8	4%	180	2
	Barley	95	3	3%	669	813	-	-	144	4	9	9%	229	8

2.1.2 Cost and abatement information for Precision farming

The abatement potential identified for precision farming is a 5% reduction in the baseline emission rate for mineral fertiliser application (Anon, 2005a).

Costs for precision farming are poorly defined, therefore we have updated the costs used in Bates (2001), based on field trials of systems on large farms in Germany. Costs are inclusive of: computer, planning, mapping and decision support software, monitoring and recording facility for tracer and yield mapping ability for combine, variable rate device for sprayer and for fertiliser application and GPS system, and operation and maintenance (Schmerler and Basten, 1999). Capital costs for the system are 39 €/ha, whilst maintenance costs are estimated at 20 €/ha per year. A 10 year lifetime is assumed for the farming system. The underlying cost savings associated with the precision farming system are shown in Table 4. These costs have been updated for changes to nitrogen fertiliser and crop prices, however the seedling costs have only been adjusted from $€_{1990}$ to $€_{2005}$.

Table 4: Cost data for precision farming.

	Fertiliser saving	Cost savings		
		Fertiliser use	Yield	Seedling
	kg/ha	euro/ha	euro/ha	euro/ha
Wheat	20	13	15	3
Barley	10	7	18	3
Maize	15	10	4	51

Cost data was translated into costs per tonne of nitrogen fertiliser and disaggregated according to the area of wheat, barley and maize within each MS (agricultural area of different crop types taken from the Eurostat database).

2.1.3 Cost and abatement information for making effective allowance for manure N and residual N

Abatement provided by making effective allowance for manure and residual N is taken as a 5% reduction to the baseline emission rate for mineral fertiliser application (Moorby, 2007).

The main cost for making effective allowance for manure N and residual N is farmer's time. It is assumed that 1 day of farmer's time is required for every 100 ha. Taking a salary of €20,000 as representative of EU average farmer's wage, this yields 55 €/day to implement the measure. However, allowance for manure and residual N could yield fertiliser savings. In addition to these costs, benefits will be generated through fertiliser savings consistent with the overall abatement for this measure of 5% (Moorby, 2007).

Costs are disaggregated across the EU using the total area of fertilised land and the total mass of fertiliser applied to that land (total area of fertilised land calculated from area of crops: grass, maize, potato, sugar beet, wheat and barley, taken from Eurostats[4] database; and fertiliser application rates as presented in Table 3). The measure is considered implementable for all mineral fertiliser application.

2.2 Spreading slurry or poultry manure at appropriate times of the year

There has been a large amount of research into the effects of spreading slurry and poultry manure at appropriate times of the year. Care should be taken not to apply manure to land at times when there is no requirement, for example late in the growing season or when there is no crop on the soil. Liquid manure has readily available inorganic N (Chambers et al., 2000), this increases the risk of NO_3 leaching, a proportion of which will be lost as N_2O. The farms at highest risk of losing N through leaching are those in high rainfall areas or with soils with small clay content (ADAS, 2001; CEH, 2004). Late winter applications can also reduce N_2O emissions as the lower temperatures slow the rate of ammonium conversion to NO_3 and therefore denitrification. Manure application in late winter or spring will reduce indirect emissions of N_2O due to reduced NO_3 leaching; there is also a possibility of a slight increase of CH_4 emissions due to a longer storage period of manure. N_2O emissions following manure applications, are however very dependent on subsequent weather conditions and it is difficult to generalise or guarantee the particular impact that timing applications will have. This measure is therefore not considered in the agricultural cost curve.

[4] ec.europa.eu/eurostat/

2.3 Matching Fertiliser Form to Conditions

Matching fertiliser N type (nitrate or ammonium based fertiliser) to season or general weather pattern, for example using urea on grassland in the spring instead of ammonium nitrate has been reported to significantly decrease N_2O emissions (Dobbie & Smith 2003). The rationale behind this approach is that urea does not contain NO_3 and hence urea-N will not be denitrified. Thus to use urea on wet grassland in spring should lead to less N_2O emissions than the use of nitrate-containing fertilizers. Conversely the use of ammonium nitrate, which contains less NH_4-N than urea, should lead to less N_2O emissions than the use of urea on dry soils when the dominant process of N_2O loss is nitrification. However, this approach may be confounded if the weather does not follow the predicted pattern, e.g. as in summer 2007. Moreover, the IPCC recently concluded (IPCC, 2006) that there is still insufficient evidence to conclude that form of fertilizer-N has a significant impact on annual N_2O emissions.

2.4 Use of nitrification inhibitors

The application of nitrification inhibitors reduces the rate of conversion of NH_4 to NO_3 (Moorby et al., 2007). NO_3 is formed, through nitrification at a slower rate, which the crop can use more efficiently. This reduces the production of N_2O and also NO_3 leaching.

Nitrification inhibitors are effective with fertilisers, slurries and urea-N so can be used both on arable soils as well as those used for livestock grazing. Extensive research in New Zealand (Di et al., 2002, 2003, 2004a, 2004b, 2004c, 2006) has shown that using dicyandiamide (DCD) on a variety of soil types and climatic conditions, significant N_2O emission reductions of up to 85% from urea patches on grazed land can be achieved (Table 5). Ditter et al. (2001) found that if slurry is injected with the nitrification inhibitor dimethylpyrazole (DMPP), N_2O emissions can be reduced by up to 32% when compared with non-treated slurry. In the laboratory, nitrification inhibitors have been shown to potentially reduce N_2O emissions by close to 100%, while reducing typically approximately 30% under field conditions (Hatch et al., 2005).

Some reviews (e.g. Edmeades, 2004), have however found more mixed results. While most studies found them to be beneficial, others report nil or negative results on nitrogen losses and plant yields, and toxic effects on some plants. Some report increasing NH_3 volatilisation and it has even been suggested that inhibitors can have a priming effect on the net mineralisation of organic nitrogen in the soil. This could lead to more nitrogen losses in the long term.

Table 5 Nitrous Oxide Emission Reductions reported from Inhibitor Use.

Soil Type	**Decrease (kg N_2O – N/ha)**	**% Reduction**	**Source**
Lismore Stony Silt Loam (urine patch)	37.5	82%	1
Lismore Stony Silt Loam (autumn urine patch)	19.1	72%	2
Lismore Stony Silt Loam (spring urine patch)	14.1	78%	2
Lismore Stony Silt Loam (autumn urine patch)	16.2	70%	3
Lismore Stony Silt Loam (spring urine patch)	22.6	73%	3
Templeton Fine Sandy Loam (autumn urine patch)	22.8	61%	3
Lismore Stony Silt Loam (autumn urine patch)	5.8	67%	4
Templeton Fine Sandy Loam (autumn urine patch)	15.2	73%	4
Taupo Pumice Sand (spring urine patch)	0.7	69%	4
Horotiu Silt Loam (autumn urine patch)	3.8	61%	4
Tokomaru Silt Loam (urine patch, 600kg N, 7kg DCD/ha)	1.79	53%	5

2.4.1 Cost and abatement information

The abatement offered by nitrification inhibitors is taken as a 30% reduction on baseline emissions from fertiliser application.

The cost of this option is derived from Bromilow (2007) who estimated 20 £/ha (28 €/ha) as the outlay for applying effective nitrification inhibitors. In the absence of substantial cost information regarding the application of nitrification inhibitors we have taken this value as the uppermost cost for nitrification inhibitors per tonne of mineral fertiliser N applied. The cost has not been differentiated between MS.

The measure is considered to be implementable with all fertiliser application.

2.5 Improvement of soil physical condition

Increased soil wetness can significantly increase emissions of N_2O. Furthermore, soil compaction, especially when the soil is wet and most susceptible to structural change creates more anaerobic conditions and therefore denitrification rates (Dobbie & Smith, 2003b, Moorby et al., 2007). Soil compaction by treading of cattle has been seen to increase N_2O emissions by a factor of two (Oenema et al., 1997).

During times when soils are wet the number of livestock per unit area or the time spent grazing by livestock should be reduced significantly. This would help to reduce soil compaction. This would also reduce the addition of dung and urea to the vulnerable area which would increase N_2O generation (Moorby et al., 2007). It was estimated by Clark et al. (2001) that avoiding compaction by decreasing the amount of time spent on the soil by livestock could reduce N_2O emissions by up to 3%.

Achieving the reductions in N_2O emissions that are suggested are likely to rely on education of farmers. The application of this method would not be at a great cost to the farmer as long as the option for moving stock to another field or dividing a field using temporary fencing was available.

Implementation will be easier on farms with access to freely-drained land which can provide alternative grazing during wet periods. Farms without access to such freely-drained areas may need to house animals for longer winter periods than those with. This will increase the amount of manure that would need to be managed. N_2O reductions could be affected by methods used to manage the additional stored manure. Additional manure management activities would need to be implemented with this extra manure to avoid a potential increase in both NH_3 and CH_4 emissions. Possible secondary effects include a reduction in water pollution from reduced surface runoff.

2.5.1 Cost and abatement information

Abatement due to reduced grazing on wet ground is estimated as a 3% reduction against the baseline emission rate of N_2O per unit manure N excreted to grazing land (Clark et al. 2001).

The cost of this option is based on estimated costs for winter grazing. Winter grazing costs are in turn banded according to the density of grazing livestock excretion. From Nix (2005), the charge for winter grazing in the UK is €$_{2005}$ 5 /ha, this forms the central banding price. The upper band is taken as €$_{2005}$ 10 /ha and the lower band as €$_{2005}$ 0.15/ha to fit with the range of grazing livestock manure densities in the study MS.

Grazing livestock manure density was calculated for each MS by dividing the total area of permanent grassland and meadow (from Eurostats database) by the total grazing excretion, based on standard IPCC parameters (discussed in section 1.2.1).

The implementation fraction of reduced grazing on wet areas is also banded according to the density of grazing livestock excretion. However, this is also adjusted by a rainfall dependent banding, based on the volume of rainfall (Eurostat: Annual precipitation volume) per unit area of agricultural land in each MS (Eurostat: agricultural land area). 50% of the UK's grazing land is thought to be at risk of poor drainage (Robinson and Armstrong, 1987) and this is taken as the value for the high rainfall depth MS. The middle banding is taken as 25% of grazing land and the lower band, for drier MS (such as: Spain, Greece and Portugal). For Cyprus and Malta, it is assumed that no grazing land is suitable for implementation of this measure. Drainage crucially depends on soil type as well as rainfall there fore this rainfall based approach is used as a first approximation and will only give a broad indication of the potential implementation level. In order to make a better estimate, soil drainage class would need to be taken into account. This would be a substantial task requiring fairly detailed information on the drainage class of soils across the EU overlain with data on rainfall. Such a task is beyond the remit of this project.

2.6 Increase livestock nutrient use efficiency

Dietary nitrogen (N) is inefficiently utilised by farm animals for the production of milk, meat and eggs, and significant quantities are excreted in manure. If N could be used more efficiently, and N excretion reduced, then emissions of N_2O from manure management should also be reduced together with emissions from grazed pastures.

Nutritional models of feed requirements suggest that N supply exceeds requirements by 25% (Anon, 2005a). However, estimates of intakes, and intakes of forage by ruminants in particular, are notoriously difficult to obtain, and it is likely that these values significantly underestimate the excess supply.

The proportion of dietary N in useful food products retained in some pig and poultry systems is around 35-40%. In extensive beef and sheep systems retention is lower at 10% or less (Anon, 2005a). This inefficient use of N may lead to unnecessarily large emissions of N_2O, while the enteric fermentation of feeds is also a major source of CH_4. However, a significant proportion of the N excreted is the result of obligatory losses associated with normal digestion, metabolism and physiological processes, and hence the maximum theoretical N efficiency is approximately 50%. Therefore the scope for reducing N excretion may not be as great as overall efficiency values may suggest. Nevertheless, until recently, nutrition management of farm animals has been used as a tool to maximise the output of useful products such as milk and meat, with little or no attention to emissions. However, its potential as a tool to help control environmental pollution has received increasing attention in recent years.

Two general strategies can be used to improve N utilisation and reduce N excretion. The first is to improve animal productivity. As more milk, meat, or eggs are produced per animal, the maintenance requirement of protein per unit of production is reduced. Significant increases in N utilisation have occurred in the last 50 years or so as a result of breeding for greater feed efficiency, and further improvements may be expected in future years. The second approach is to improve the match between the quality of protein quality fed and that required by the animal.

A number of strategies for improving N utilisation by pigs and poultry have been identified. Breeding lower protein cereals, with improved essential amino acid profiles, could result in greater N utilisation efficiency, although the benefits would largely be at a farm level – because less fertiliser N inputs are required for lower protein crops – rather than at an animal level. Synthetic amino acids are increasingly being used in pig and poultry rations, and their inclusion results in significant improvements in N utilisation at an animal level because essential amino acid supply can be more closely matched to requirements. Other strategies, such as phase feeding allow further matching of dietary amino acid supply with requirements, resulting in reductions in overall dietary N concentration and N excretion. Although soya bean meal remains the preferred protein supplement in pig and poultry diets, alternative proteins (e.g. peas and beans, lupins) provide some scope as alternative protein sources. However, because of the quality and concentration of their protein, N utilisation efficiency is reduced when they are included in significant quantities.

In the short term the very cheap and plentiful supply of oilseed meals and other protein-rich by-products on world markets provides only limited incentive for industry to invest in projects aimed specifically at improving the utilisation of feed proteins. Availability of co-products produced during processing of crops for food (e.g. vegetable oil) and industrial uses (e.g. alcohol) will continue to increase and to be a major source of feed protein.

For cattle and sheep, the greatest potential for improving feed N utilisation appears to be associated with improvements in the supply of energy to rumen micro-organisms. Increasing use of high-sugar grasses or forage maize (as maize silage) is likely to offer the greatest potential for improving N utilisation by ruminants. Substituting conventional grass varieties with high-sugar grass in sheep and beef cattle diets could result in reductions approaching 2.5% of N_2O emissions from agriculture (Anon, 2005a). There is also potential for reducing N_2O emissions as a result of feeding maize silage to dairy cows and beef cattle, although to achieve any meaningful reduction would require a substantial increase in the area sown to the forage maize.

The use of silage made from legumes, or from legume-grass mixtures can significantly increase N utilisation efficiency *at a farm level* - largely as a result of the ‘capture’ of atmospheric N by the legumes, and the reduced amounts of N fertilizer used. However, N intakes on legume-based diets tend to be greater than on grass silage-based diets, so that efficiency *at an animal level* is reduced. 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 N utilisation efficiency and minimise the environmental burden.

2.6.1 Cost and abatement information

Abatement associated with increased nutrient use efficiency is estimated at 6% against baseline emissions from use of manures as fertilisers and excretion of manures to grazing land.

Costs of $€_{2005}$ 188 per dairy cow (Anon, 2001) are used here and converted to a cost per unit of manure either excreted to grazing land or applied to soils as fertiliser, using IPCC parameters for excretion rate and the fraction excreted to grazing land that is applied to soils as fertiliser. The cost per dairy cow is apportioned to grazing manure and fertiliser according to the excretion rate.

The measure is assumed to be applicable to all manure fertiliser and manure excreted to grazing land.

3 Options to reduce enteric CH_4 emissions

It can be seen then that the problem of reducing enteric methane emissions is dependent on methane emissions per head of livestock and the total number of livestock. Clearly, enteric methane emissions can be reduced through a cut in the total head of livestock. The problem of reducing the methane emissions per head of livestock can be addressed by increasing the productivity of ruminants. In other words, reducing the methane emissions per unit animal produce. A reduction in CH_4 emissions per unit animal product can be made via any process that increases the ratio of livestock 'production' to 'maintenance' (Monteny et al., 2006).

Practices for reducing CH_4 emissions from enteric fermentation can be grouped into three general categories:

- Improved feeding practices;
- Specific agents and dietary practices;
- Long-term management changes and use of genetic improvement.

3.1 Improved feeding practices

Bacteria are the principal micro-organisms that ferment carbohydrates in the rumen, and the type of bacteria required depends on the animal's diet. The composition of micro-organisms in the rumen is also important in determining the composition of products from the fermentation process. For example, certain bacteria can shift the fermentation process from less-reduced to more-reduced end products, reducing the amount of methane produced. Finally, the presence of protozoa, another type of micro-organism, in the rumen may also be important, although its role is not clear. Some studies have indicated that removal of the protozoal population from the rumen (defaunation) may lower the amount of hydrogen in the system and thereby reduce methane production (Vermorel and Jouany, 1989).

Changing the VFA composition of the rumen so that less acetate and more propionate are formed, will lead to reduced methane emissions per unit of animal product. Replacement of roughage, which contains a high proportion of structural carbohydrate (fibres), with concentrates, can improve propionate generation in the rumen and decrease emissions of methane.

3.1.1 Replacing roughage with concentrates

One of the values of ruminants is that they can utilise what other, monogastric,

livestock cannot: fibrous foods with low energy value. Roughage, which forms a high proportion of the typical diet for livestock, is such a feedstock and contains a high proportion of structural carbohydrate. Replacing it with concentrates can reduce methane emissions by improving propionate generation in the rumen (Bates, 2001). Although concentrates may increase daily methane emissions per animal, emissions per kg-feed intake and per kg-product are invariably reduced (IPCC, 2007). IGER (2001) reported that replacing grass silage with general concentrate led to an increase in CH_4 emissions whereas using a high starch concentrate led to a 7% drop in CH_4 emissions and a 14% increase in milk yield per animal.

The net benefit of the use of concentrates does however depend on:

- Reduced animal numbers or younger age of slaughter;
- How the practice affects land use;
- Emissions from manures and fertilisers used in the production of concentrates;
- Emissions from processing and transporting the concentrates.

3.1.1.1 Cost and abatement assumptions for option

Replacement of roughage with high starch concentrates is taken here to give rise to a 7% reduction against baseline enteric methane emissions from dairy and non-dairy cows.

The replacement of roughage with concentrates is applicable to both dairy and beef cattle whilst they are housed indoors and therefore not grazing. Geographically, this option is therefore suitable for all European members' states where cattle are housed indoors for all or part of the year.

In order to calculate the costs associated with the measure, the following assumptions are made in this study:

- The typical winter feeding regime comprises; 5kg of concentrate, 7.5 kg silage and 2.5 kg maize.
- Typical feeding regime with an increased concentrate proportion comprises: 10kg concentrates, 2.5kg silage and 2.5kg maize.
- Dairy cattle are outdoors for 160 days of the year and beef cattle 183 days.
- The costs of concentrates, silage and maize are taken to be 350 €/tonne, 109 €/tonne and 123 €/tonne, respectively[5].

3.1.2 Adding oils or oilseeds to the diet

The addition of certain oils or oilseeds to the diet can reduce emission by the process of defuanation (reducing the number of protozoa in the rumen). Various fats and oils have been used for this purpose, however, recent research has shown that the addition of coconut oil is particularly effective. Since refined coconut oil is expensive, alternatives such as copra meal (a coconut product) have been investigated. Gerbens (1998) estimated that replacing 'low fat' concentrates with concentrates of a high fat content (about 7%) could reduce reduction in

5 Eurostat (Data extracted 4th Feb 2008). See:
http://epp.eurostat.ec.europa.eu/portal/page?_pageid=0,1136206,0_45570467&_dad=portal&_schema=PORTAL

emissions from dairy cows in Western Europe by 7.3% and in other cattle by 4.3%. This reduction assumes that for every kg of concentrate replaced, CH_4 emissions are reduced by 2%. More recently, Jordan et al. (2006) demonstrate a reduction of approximately 18% in daily CH_4 emissions as a result of the inclusion of 250g of refined coconut oil (RFO) in the diet. This methane reduction is further enhanced by better animal performance with the RFO supplement; average daily gain was around 15% higher and average daily carcass gain 9% above the control, leading to a reduction in the finishing time per animal. Further research indicates that including refined soy oil (RSO) in the proportion 6% of dry matter intake (DMI) of young bulls can lead to a 40% reduction in CH_4 emissions per day (Jordan et al., 2006b). No detrimental impact on animal performance was observed with the treatment.

3.1.2.1 Cost and abatement assumptions for option

Abatement associated with this measure is taken as a 10% reduction against baseline enteric methane emissions from dairy and non-dairy cattle.

The addition of refined coconut or soy oil is considered here for both dairy and beef cattle. The emissions reductions reported for these oils are for beef cattle. They are therefore taken as upper limits for dairy cattle and, to reflect the possibility that emissions are not reduced to the same extent for dairy cattle, values half those found for beef cattle are also considered.

The addition of both types of oil to the feed intake of cattle is considered here and the associated costs are calculated using the following assumptions:

- 250 g of refined coconut oil or 6% DMI refined soy oil is consumed per day.
- The oils replace the same mass of untreated concentrate.
- Dairy cattle are outdoors grazing for 160 days per year, beef cattle for 183 days.
- The costs of concentrates, refined coconut oil and refined soy oil are taken to be 350 €/tonne [6], 772 €/tonne [7] and 488 €/tonne [8] respectively.

The potential impacts on cropping areas have not been taken into account in the analysis.

3.2 Dietary additives

Establishing conditions under which rumen fermentation will be optimised requires an understanding of the nutrient requirements of the mixed microbial population. Growth of rumen microbes will be influenced by chemical, physiological and nutritional components. The major chemical and physiological modifiers of rumen fermentation are rumen pH and turnover rate and both of these are affected by diet and other nutritionally related characteristics such as level of intake, feeding strategies, forage length and quality and forage

6 Eurostat (Data extracted 4th Feb 2008). See:
http://epp.eurostat.ec.europa.eu/portal/page?_pageid=0,1136206,0_45570467&_dad=portal&_schema=PORTAL

7 http://www.commodityonline.com/commodities/oil-oilseeds/newsdetails.php?id=5807

8 http://www.fas.usda.gov/oilseeds/circular/2007/October/oilseedsfull1007.pdf

to concentrate ratios. Although significant advances in knowledge of effects of various combinations of these factors on microbial growth have been made in recent years, there is still insufficient information available to identify and control the interactions in the rumen that will result in optimum rumen fermentation.

One of the options described above indicated how changes in concentrates could increase rumen efficiency. A number of other possible dietary options have been identified which could also lead to increased rumen efficiency, these include:

- hexose partitioning;
- propionate precursors;
- direct fed microbials (acetogens or methane oxidisers);
- an immunogenic approach (vaccinations).

Several of these options are still under research and development. Accordingly, cost information is sparse. However, some discussion of costs is available for propionate precursors, direct fed microbials and vaccinations. These options are therefore discussed in more detail along with the use of growth hormone additives, which can improve animal performance and hence reduce CH_4 emissions per unit of product.

3.2.1 Propionate precursors

Within the rumen, hydrogen produced by the fermentation process may react to produce either methane or propionate. By increasing the presence of propionate precursors such as the organic acids, malate or fumarate, more of the hydrogen is used to produce propionate, and methane production is reduced.

Propionate precursors can be introduced as a feed additive for livestock receiving concentrates. The propionate precursor, malate, also occurs naturally in grasses, and it is possible that plant breeding techniques could be used to produce forage plants with high enough concentrations of malate. Considerable research is needed, but if these techniques were successful then this mitigation option could then also be used with extensively grazed animals.

Recent research suggests that high doses of fumarate are required to elicit a response. There are practical considerations to supplementing livestock diet with high levels of fumarate. Newbold et al. (2005) consider two additive forms, fumaric acid and sodium fumarate. They suggest that feeding high doses of the acid form could be problematic since it would affect rumen pH. On the other hand, using the salt form would deliver potentially toxic levels of salt. There are also concerns over the impact of high propionate levels on feed intake.

3.2.2 Vaccination

Vaccinations against methanogenic bacteria are being developed. Since CH_4 production represents a loss of energy (2-15% of gross energy intake) (Johnson and Johnson., 1995), a reduction in methanogens would increase the efficiency of rumen fermentation. A study from the Commonwealth Scientific and Industrial Research Organisation (CSIRO) in Australia investigated if methane emissions from sheep immunized with an anti-methanogen vaccine were significantly lower than methane emissions from non-immunized sheep (Wright et al., 2004). One group of immunized sheep showed a significant 7.7% reduction in methane production per kg dry matter intake compared to the controls. However, emissions from another group, treated with a different vaccine, were not significantly different to the control group.

The development of a vaccine against methanogenic micro-flora is of great interest, however there are no products commercially available yet (IPCC, 2007).

3.2.3 Ionophores and natural extracts

Fermentation in the rumen can be modulated by the use of feed additives to enhance of inhibit specific microbial populations. Ionophores are antibiotics and chemical feed additives that increase productivity by adjusting several fermentation pathways. Bates (2000) reported that on average, an 8% increase in feed conversion efficiency has been observed and a reduction in methane production of up to 25%. However, the CH_4 reduction effect is apparently transient (4-6 weeks in duration) and can be nullified, since the rumen protozoal population can develop a mechanism to overcome the toxic effects of ionophores.

Since January 1 2006, the use of antibiotics for non-medicinal purposes has been illegal in the EU[9] because of concerns that widespread use could lead to increased antibiotic resistance of pathogens responsible for human diseases. Monensin, an ionophore, was one of the last antibiotics to be removed from feed by this ban and therefore cannot be used to modulate rumen metabolism (Moorby et al., 2007). There is, however, scope to incorporate various novel natural additives that have been suggested to reduce N lost through excretion of ammonia. It is not yet clear though whether natural extracts can decrease CH_4 production.

3.2.4 Bovine somatotropin and hormonal growth implants

Bovine somatotropin (bST) is a growth hormone that although it does not specifically suppress CH_4 formation, can improve animal performance. For example; increasing and maintaining high milk yields in dairy cows, or as a way to increase the rate of live weight gain in beef production, allowing animals to be finished more quickly. The enhancement to animal productivity can reduce the CH_4 emissions per-kg product.

Emissions reductions and abatement costs have been calculated for the use of bST in livestock, however, on animal welfare grounds, the use of bST was prohibited in the EU in

[9] Regulation 1831/2003/EC on additives for use in animal nutrition, replacing Directive 70/524/EEC on additives in feeding-stuffs.

January 2000[10] and it is unlikely that bST or other hormones would become acceptable for EU Member States in the near future. For this reason, mitigation through the use of growth hormones is not considered further in this study.

3.3 Long-term management changes and use of genetic resources

Increased productivity through breeding and better management practices can lead to a reduction in methane output per unit of animal product. The use of plant and animal genetic resources to improve lifetime and efficiency of livestock systems focuses on three elements (Moorby et al., 2007):

- Increasing efficiency of individual animals;
- Use of improved animal genetics for longevity, fertility (including calving ease for dairy cows) and other non-productive traits;
- Use of new forage plant varieties for improved nutritional characteristics.

With increased efficiency, meat-producing animals reach slaughter weight at younger age, with reduced lifetime emissions. However, as IPCC (2007) point out, the whole-system effects of such practices may not always lead to reduced emissions. Historically selection goals have focussed on production and did not include health and fertility traits. The result is less ‘robust’ animals and, in the dairy industry, more replacement heifers are required in the herd (Lovett et al,. 2006).

The lifetime efficiency of a dairy cow depends on age at first calving, the number of lactations, the calving interval, the duration of dry periods and the quantity of milk produced when lactating. All these affect the volume of milk and CH_4 emissions over a lifetime. In the first two years cows develop and are ready for insemination only after this period. Hence these first two years are both economically unproductive and environmentally deleterious since the heifer is producing CH_4 and taking in feed without actually producing milk. The rapid turnover of milking cows due to early mortality or infertility means that energy inputs and CH_4 emissions are ‘wasted’ in the process of rearing heifers before they reach their first pregnancy and lactation. Once maturity is reached, it important to keep the cow milking for as long as possible so that the investment in growth and development pays off and to keep the replacement rate as low as possible. Thus an increase in the number of lactations over the lifetime of a cow has the potential to reduce CH_4 emissions.

Improvements in dairy productivity through breeding strategies have led to higher yielding cows that are more prone to lameness, infertility and illness; they also usually have shorter life spans than lower yielding breeds. Once these factors are taken into account, the improvements in milk productivity (and concomitant decline in CH_4 emissions per unit of

[10] Institute of Food Science and Technology. See:
http://www.ifst.org/uploadedfiles/cms/store/ATTACHMENTS/BST_summary.pdf

output) may in practice be difficult to achieve. As the genotype (milk yielding potential) and level of concentrate supplementation increases, the number of dairy cows required to fill the annual milk quota declines. Since fewer lactating cows are needed to fulfil the milk quota it could be expected that fewer calves will be needed to replace them, consequently follower and calf numbers should decrease as well. However, the turnover rate for higher yielding cows is greater than for less productive animals and this will increase the number of replacement cows needed. Therefore, while the absolute numbers of stock may decrease, the *ratio* of the non-productive younger animals to dairy cows increases. As a result, the population of non-productive cows increases and both on-farm and total life cycle CH_4 emissions increase with increasing genotype potential (i.e. yield) (Lovett et al., 2006).

As discussed above, fertility has a major effect on the number of heifer replacements required to maintain herd size for a given milk quota or number of cows. Taking typical fertility levels for commercial dairy herds, the proportion of total gas emissions produced by herd replacements is up to 27% for methane. Modelling suggests that restoring fertility levels to 1995 levels could reduce methane emissions by 10-11% (Garnsworthy, 2004). More conservative calculations put the reductions at 3% for CH_4 (Moorby et al., 2007). Simulations of dairy herds in the US show that reducing the culling rate by 10% can lead to a very desirable result; providing a 5% reduction in total herd emissions whilst also increasing the profit of the herd.

Today, breeding goals in most species are more balanced to reflect some of these factors, but there is still much scope for health and fertility traits to catch up with yield related factors (Moorby et al., 2007).

Plant breeding programmes have similarly tended to concentrate on agronomic characteristics over nutritional ones. Clearly, traits such as dry matter yield and disease resistance are important, however, greater emphasis on traits such as carbohydrate quality and protein degradability would help improve forage use efficiency by animals (Moorby et al., 2007).

Assigning a cost to the emissions reduction associated with longer term management and use of genetic resources is difficult. However, there is logic is the assumption that costs associated with identifying and trying to improve genetic traits such as health and fertility (as discussed above) would be compensated by cost reductions for example if the number of replacement heifers per herd was reduced. Similarly, the costs of developing forage with greater nutritional value will be compensated by productivity gains that result from the use of that forage. A simplistic argument could be made, based on this logic that the abatement cost for long term management and breeding will be negligible.

3.3.1.1 Cost and abatement assumptions for option

It is assumed that a 5% reduction in methane emissions can be achieved through this option. As discussed above the option is also considered to be cost neutral.

In agreement with previous research, it is assumed here that making better use of genetic resources could lead to a reduction in methane emissions from dairy cattle, non-dairy cattle and sheep. Geographically, the options are considered for all EEA MS.

4 Options to reduce CH_4 and N_2O emissions through manure management

Strategies to target emissions of methane and nitrous oxide from manures focus on housing and storage of manure or on controlled anaerobic digestion of the manure (Monteny et al., 2006). However, some of the proposed methods can only target one of the two greenhouse gases, potentially negating much of the mitigation effort. The following sections will discuss these options and their impacts for CH_4 and N_2O emissions in more detail.

We do not take into account country specific variations in the N-excretion rate of livestock in our analysis; instead we take standard IPCC (2006) excretion parameters.

4.1 Housing and storage

Some of the options proposed for greenhouse gas mitigation from housing of livestock and storage of manures are listed in Table 6.

Table 6: Options proposed for reducing greenhouse gas emissions from manures and effect on CH_4 and N_2O (Monteny et al., 2006; Moorby et al., 2007; IPCC, 2007; Amon et al,. 2006).

Option	Description	CH_4	N_2O
a.	Reduction of gas production through deep cooling of manure to less than 10℃	↓	↓
b.	Removal of the gas source, e.g. by frequent and complete removal of manure from indoor storage pits	↓	↓
c.	Mechanical separation	↕?	↕?
d.	Swapping manure practice from solids to a slurry based system	↑	↓

Mechanical separation of livestock manures (option c.) involves the partial removal of organic and inorganic solids from liquid manure by physical processes. Separation devices typically use one of two techniques: using a screen that collects the majority of the solid fraction but through which the liquid fraction will pass; centrifugation, which uses a cylinder rotating at high velocity to increase the settling rate of suspended particles. After collection the solid fraction is stored and then applied to croplands as a fertiliser. The action of separation has been reported reduce the overall emissions of greenhouse gases from slurry by over 35% (Amon et al., 2006). However, this has been disputed by a laboratory-scale study that showed storage of the separated fractions resulted in a 30% increase in aggregated greenhouse gases compared when compared with untreated slurries (Dinuccio, 2007). Since the exact influence of mechanical separation is not yet established, this option is not considered further as a mitigation measure.

Moorby et al. (2007) identify the change from solid manure to a slurry based system as a potential future mitigation practice to limit N_2O emissions. Results suggest a 15% reduction in N_2O emissions for dairy systems, with a similar reduction likely for pigs. However, they suggest that CH_4 emissions are likely to increase from slurry and the extent to which these enhanced emissions will negate the benefits of reduced N_2O emissions is uncertain. In addition, the costs of slurry storage are high, requiring pumps, alternative spreading equipment and potentially altered building design (slatted flooring, slurry collection pits and storage facilities). Monteny et al. (2006) discuss the addition of high C substrate to solid manure heaps to mitigate emissions of N_2O. Data also suggests that compacting and covering solid manure heaps to reduce oxygen penetration, therefore maintaining anaerobic conditions, might limit N_2O emissions (Chadwick, 2005). However, this can be expected to increase CH_4 emissions.

Until the overall balance of GHG and other gaseous emissions from slurry and solid based manure management systems is quantified it is difficult to factor these options into our analysis. An argument could also be made that it is unlikely that considerable infrastructure investment, such as might be required to implement these management practices, would not be made until the impacts are quantified in detail. Therefore, discussion focuses on options a and b, which are discussed further in the following section.

4.1.1 Reducing the rate of microbial action

Emissions of methane and nitrous oxide from animal manures result from microbial degradation of volatile solids, which serves as an energy source and as a sink for atmospheric oxygen. Microbial action is temperature dependent and cooling the manure can slow the rate at which decomposition occurs.

Pigs respond adversely to extreme climatic conditions: cold dramatically decreases the food conversion effectiveness in pigs being fattened, and hot weather disturbs their reproductive performance. In the cooler countries of the EU, pigs are therefore generally raised in indoor housing, which is heated in the winter. If the floor in the housing is concrete then the manure is usually mechanically removed to a solid storage facility outside, but in systems with a slatted floor, the manure is collected as slurry in a cellar or pit underneath the housing. This type of system is very common in intensive pig farming operations and manure is often stored there for some months. This creates relatively high emissions, as the manure begins to anaerobically decompose, particularly as the housing is often heated.

Model predictions suggest that cooling pig slurry would reduce total annual CH_4 and N_2O emissions by 21% (CO_2-eq.) compared with slurries at typical in-house temperatures (Sommer et al., 2004). Previously it has been found that at a temperature of 10°C emissions from slurry can be 60 to 100% lower than emissions from slurry stored in animal housing kept at 20°C (Zeeman, 1994).

4.1.1.1 Assumptions for this option

It is estimated that reducing the rate of microbial action might lead to a 20% reduction in emissions from intensive pig rearing systems. Following the reasoning of Bates (2001), the costs associated with implementing this option are considered to be negligible since it is assumed that all countries in which it is applicable will primarily be implementing this option to reduce ammonia emissions. The implementation of this option is already underway to help reduce ammonia emissions in some countries such as the Netherlands.

This option is suitable for use with intensive pig rearing systems (i.e. with a large herd size), in which manures are stored for greater that one month. Following Bates (2001) and Bates et al. (2004) 73% of pig manure within Western Europe is stored for more than one month and 29% is stored in this manner in Eastern Europe. It is assumed that this option would only be implemented in countries with a cooler climate; where winter temperatures are low enough to cool manures that are simply moved to storage outside of the housing building. Therefore the option is applied to all countries in the EU-27 except Greece, Italy, Portugal and Spain.

4.1.2 Removal of the gas source

Frequent and complete removal of the manure from pits can also aid reducing emissions since manure that remains in storage acts as an inoculant so that anaerobic decomposition of fresh manure added to the system begins quickly (Osada et al., 1998). Model results show daily flushing of slurry from cattle houses would reduce total annual CH_4 and N_2O emissions by 35% (CO_2-eq.) (Sommer et al., 2004). Rinsing out the manure cellar or stable floor can ensure that all manure is removed from the cellar. As long as this is done using cleansed water separated from the collected slurry, the volume and dry matter content of the manure is not increased so that storage facilities for the slurry do not need to be increased.

4.1.2.1 Assumptions for this option

Abatement of 35% is estimated from removal of gas sources against baseline emissions of CH_4 and N_2O from manure management.

The net costs associated with this option are considered to be zero since its implementation is linked to efforts to reduce ammonia emissions (section 4.1.1.1).

The removal of manure from pits is applicable to intensive pig farms in which manure is stored in pits for greater than one month. As introduced in section 4.1.1 above, 73% of pig manure is stored in this way in Western Europe and 29% in Eastern Europe.

4.2 Controlled anaerobic digestion

Anaerobic digestion (AD) is the bacterial fermentation of organic material under controlled conditions in a closed vessel. The process produces biogas which is typically made up of 65% methane and 35% carbon dioxide with traces of sulphur compounds, volatile organic compounds and ammonia. This biogas has a typical calorific value of 17 to 25 MJ/m^3, depending on CH_4 content, and can be combusted directly in modified gas boilers, used to run an internal combustion engine or simply flared (Bates, 20011). Monteny et al. (2006) point out that biogas production through anaerobic digestion, combined with power/heat generation, seems to be the most logical measure to reduce greenhouse gas emissions from livestock manures. Applying anaerobic digestion in combination with some kind of combustion process, ensures that most of the carbon within the manure is ultimately converted to carbon dioxide before being released to the atmosphere. There are two types of AD processes[11]:

- Mesophilic digestion: The digester is operated at around 35°C and the feedstock remains in the digester typically for 15 to 30 days. This digestion process tends to be more robust than thermophilic digestion (below). However, gas production rate with mesophilic plants is lower, which consequently needs longer residence time within the anaerobic digester.
- Thermophilic digestion: The digester is operated around 55°C and the residence time of feedstock is around 5 to 15 days. In comparison with mesophilic systems, thermophilic digestion offers higher methane yield and rate of production, and more effective destruction of pathogens and micro-organisms. On the downside, the technology is more complex and requires a higher degree of control and management effort.

Typically 50 to 80% of the organic matter present is converted to biogas with the remainder consisting of a relatively odour free, peat-like residue and a liquid residue (in some systems), which can be used as a soil conditioner and fertiliser. Emissions from anaerobic digestion can be conceptually assigned to one of the following processes: storage, fermentation or field application of the substrates. A comparison between a cattle slurry anaerobic system and untreated slurry showed that GHG emissions from slurry stores were more significant than emissions after field application (Clemens, 2006). Therefore, slurry storage units should be sealed gas tight in order to capture any emissions from anaerobic digestion occurring prior to introduction into the fermentation unit.

4.2.1 European perspective AD

There is considerable variability in the penetration of AD systems across European Member States. Germany has significantly more plants (Figure 7) than all the other states combined and produced over 1000 million m^3 biogas in 2004.

[11] European Commission project, AD-Nett. See: http://www.ad-nett.net/

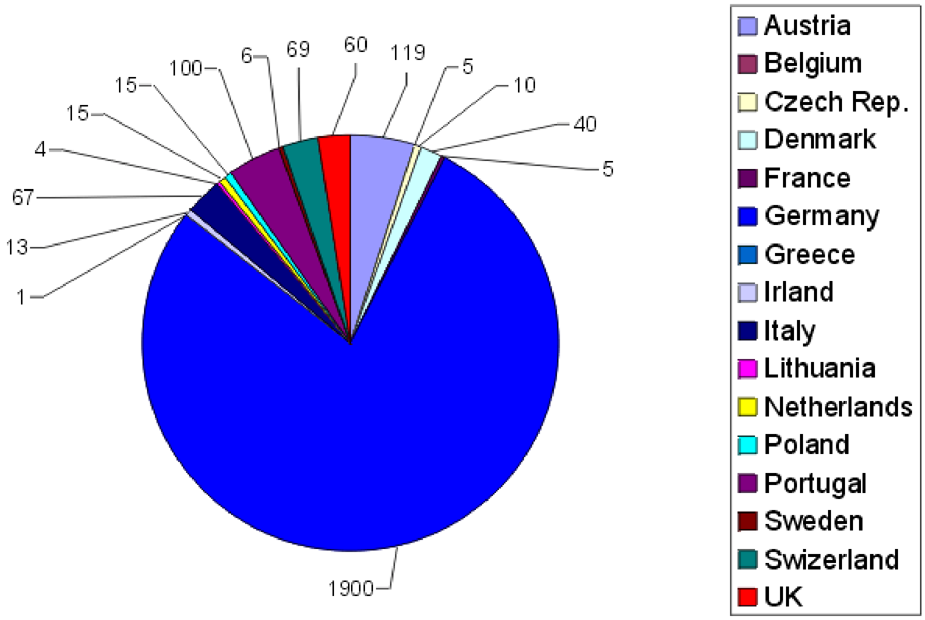

Figure 7: Farm biogas plants in EU Member States (AD-Nett)[12].

Perhaps the principal driver for these differences is economic and in light of Figure 7 it is no surprise to find that the feed-in tariff in Germany is higher than in any other European State. In addition, there are differences in capital and maintenance costs between States as a result of the different agricultural systems in place and the AD operational/production experience levels.

4.2.1.1 Cost and abatement assumptions for this option

Due to the temperature dependence of microbial activity, emissions from storage units in cool climates have previously been expected to be lower than those in warm climates (Bates, 20011; Bates et al., 2004). Emission reductions of 50% were considered achievable for countries with cold climates, compared to untreated liquid slurry systems. For countries with warmer climates, this reduction has been assumed to be higher at 75%[13]. Moorby et al. (2007) estimate that methane emissions from slurry storage in the UK could be reduced by as much as 90% by AD compared with conventional slurry storage. Therefore, given the identified need for slurry storage units to be sealed gas-tight, it is assumed here that for countries with a cool climate an optimal GHG reduction of 90% is achievable.

There are typically two scales of AD plants; farm-scale and centralised plant, which can also take in other biodegradable wastes or other agricultural products (as is the case in Germany). Both these scales of plant will be considered here and the economic issues concerning both

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[12] http://www.ad-nett.net/index.html
[13] Spain, Greece, Portugal, Cyprus, Malta and Turkey, for all other countries a value of 90% was used.

are discussed below. The technology is considered here for dairy and non-dairy cattle, pigs, sheep and poultry that are housed indoors. It is also considered for all EEA MS.

Table 7: Literature values for anaerobic digestion system costs.

Reference	Technology	Country	Livestock		Cost	Overhead and Maintenance	Notes
			Type	Head	(€$_{2005}$)	(€$_{2005}$)	
1. Farmscale digesters							
US EPA (1997)	Covered anaerobic digester	US	Dairy	500	135,571	Not given	The figures for capital cost are total installation costs. No values for overhead and maintenance costs were provided
	Covered anaerobic digester	US	Pigs	1000	73,011	Not given	
	Plug and flow digester	US	Dairy	500	184,807	Not given	
	Complete mix digester	US	Pigs	1000	81,072	Not given	
IGER (2001)	Low-tech digester	UK	Dairy		140,127	58,338	No values for the number of livestock or the capacity of the digester were provided
	Hi-tech digester	UK	Dairy		216,196	58,338	
AEA (2004)	Covered anaerobic digester	UK	Dairy	240	363,367	Not given	Costs quoted are total installed costs. Values for overhead and Maintenance costs were not supplied
	Covered anaerobic digester	UK	Dairy	250	372,376	Not given	
	Covered anaerobic digester	UK	Dairy	112	318,321	Not given	
	Covered anaerobic digester	UK	Dairy	135	343,847	Not given	
	Covered anaerobic digester	UK	Dairy	170	304,808	Not given	
	Covered anaerobic digester	UK	Dairy	250	373,877	Not given	
2. Centralised digesters				Daily capacity (m3)			
Nielsen et al. (2002)	Codigestion biogas plants	Denmark	Not given	300	4,558,110	776,567	Assumes 90% availability annually at an average 80% of daily capacity
		Denmark	Not given	550	6,705,861	1,176,105	
		Denmark	Not given	800	8,163,331	1,590,649	
Wulf et al. (2006)	Codigestion system	Denmark	Not given	40	2,275,700	92,698	Capacity assumes a lifetime of 120 days. O&M in the range 1% for buildings to 17% for machinery. Costs were provided as plant costs only. The total installation cost quoted assumes an extra 100% of this value.

Implementation of on-farm and centralized anaerobic digestion

Implementation of on-farm AD is assumed to be feasible for those holdings with herd sizes greater than 100 head, which operate a liquid manure management system. However, this implementation is further adjusted to align with projected maximum implementation of on-farm AD in Denmark, which is assumed to be 40% of viable farms for cows and 33% for pigs[14]. Since the viability of on-farm schemes is related to the head of livestock, the average herd size of each MS is calculated relative to that in Denmark and used to scale the 40% implementation of holdings on liquid system with herd size over 100.

Implementation of CAD in each MS is determined by considering the livestock density relative to Denmark, where it is estimated 40% of the livestock population could be covered by CAD. By scaling the remaining head of population by the coverage of Denmark adjusted by the relative population density of each MS yields estimates for the numbers of pigs and dairy cows covered by CAD. Given the daily excretion rates of dairy cows and pigs, the number of supportable CAD plants can be found.

Livestock numbers with herdsize greater than 100 and number of holdings with herd size greater than 100 taken from the Eurostats database. Proportion of livestock on liquid system was taken from RAINS model output.

The Table 8 below shows the resulting maximum proportion of total national pig and dairy manure that could be processed by anaerobic digestion in 2020 and 2030.

[14] The proportions chosen for viable farms which implement on-farm AD reflects the imperative for useful disposal of manures from dairy cows versus pigs.

Table 8 Projected proportions of pig and dairy manure processed by anaerobic digestion.

	Proportion of total Pig Manure processed by		Proportion of total Dairy Cow manure processed by	
	OFAD	CAD	OFAD	CAD
Austria	2%	4%	0%	7%
Belgium	10%	42%	3%	38%
Germany	6%	8%	3%	11%
Denmark	22%	31%	14%	18%
Spain	31%	8%	0%	1%
Finland	8%	3%	0%	9%
France	10%	4%	0%	5%
United Kingdom	15%	2%	9%	14%
Greece	6%	3%	0%	2%
Ireland	39%	3%	6%	40%
Italy	26%	3%	4%	10%
Luxembourg	1%	5%	0%	2%
Netherlands	23%	51%	34%	62%
Portugal	13%	6%	1%	4%
Sweden	9%	1%	0%	1%
Cyprus	26%	18%	9%	13%
Czech Republic	32%	5%	9%	10%
Estonia	39%	2%	0%	0%
Hungary	33%	4%	7%	9%
Lithuania	13%	1%	1%	10%
Latvia	27%	1%	1%	5%
Malta	12%	86%	8%	25%
Poland	3%	9%	1%	18%
Slovakia	19%	3%	7%	7%
Slovenia	39%	3%	0%	4%
Bulgaria	18%	2%	1%	11%
Romania	10%	1%	0%	15%

Costs of anaerobic digestion

The costs of anaerobic digestion plant included in the cost curve calculations are based on the values presented in Table 7. Costs are calculated separately for on-farm digesters and CAD plants. For on-farm digesters, the relationship between costs and head of livestock is derived from the AEA (2004) values, which are UK figures and assumed to be based on dairy farms. This relationship is then used to scale the US EPA (1997) values for pigs. Costs are then calculated for on-farm AD for dairy cattle and on-farm AD for pigs. For the head of livestock not already covered by on-farm schemes, CAD is considered. The assumption is made that CAD plants will have a capacity of around 500 m^3/day and cost information is therefore taken for this size plant based on a power series relationship fitted to the values of Nielsen et al. (2001). Results for 8 Member States are shown in Table 9.

Table 9: Investment costs for pigs and dairy cow manure processed through OFAD and CAD plants, examples for 8 EU Member States.

	Investment cost (€/head pig)		Investment cost (€/ head dairy cow)	
	OFAD	CAD	OFAD	CAD
Denmark	489	125	2238	773
Spain	481	133	1930	824
Greece	586	-	2263	-
Ireland	461	126	2417	781
Italy	485	124	1905	769
Netherlands	530	127	2433	785
Poland	509	127	1412	785
Bulgaria	458	134	2024	828

Overhead and maintenance costs (see Table 9) are calculated separately for OFAD and CAD. For OFAD plants, O&M costs are assumed to be 5% of investment costs. For CAD plants, the O&M costs are calculated for the assumed average sized plant (500 m^3) according to a power series through the O&M costs presented in the main report. As described for investment costs, O&M for CAD plants is split between pig and cattle according to the volume of manure generated respectively by each.

We subtract income generated from digestate and biogas sales from the O&M costs. We assume that 0.98 units of digestate are produced per unit of manure processed by OFAD plants 0 and .95 for CAD plants. Half of this digestate is assumed to be sold at a value of 10 €/tonne digestate. For biogas, we assume that around 15 m^3 is produced per tonne of manure processed and all of that biogas is valued at €0.1/m^3.

Table 10: O&M costs for pigs and dairy cow manure processed through OFAD and CAD plants, examples for 8 EU Member States.

	O&M costs (€/head pig)		O&M costs (€/head dairy cow)	
	OFAD	CAD	OFAD	CAD
Denmark	6	-11	0	-69
Spain	6	-11	-15	-67
Greece	11	-	1	-
Ireland	5	-11	9	-69
Italy	6	-11	-17	-69
Netherlands	8	-11	10	-69
Poland	7	-11	-41	-69
Bulgaria	4	-11	-11	-66
Romania	2	-11	-28	-68

5 Conclusions

The maximum technical potential identified for the agricultural sector across Europe as identified in the current study is 160 Mt CO_2-eq in both 2020 and 2030 (Figure 8). This is a 35% reduction against European agricultural sector emissions in 2005.

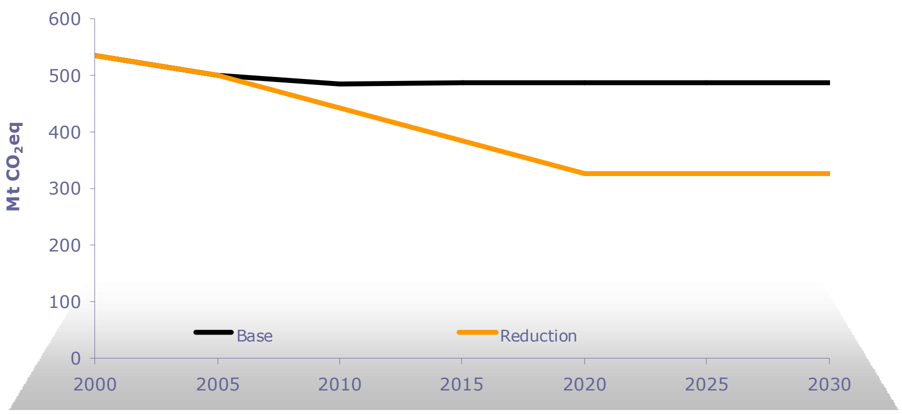

Figure 8 Historical agricultural emissions (1990-2005), business as usual baseline emissions and maximum identified abatement potential.

The abatement potential that is available at zero of negative cost in 2020 and 2030 is some 47 Mt CO_2-eq/year (see Figure 9 and Table 11). This amounts to a reduction of 10% against 2005 levels and is therefore consistent with the EU target for emission reduction across the non-emission trading sectors of the European economy. This suggests that agriculture could play a substantial role in delivering against the non-traded sector targets.

The remaining measures all have a cost effectiveness of greater than zero. However, there are several measures that are cost effective at up to €20, most important the addition of nitrification inhibitors. The measure with the highest specific costs is better livestock nutrient use efficiency, which has an estimated cost of more than 3,000 €/tCO2-eq (see Figure 9).

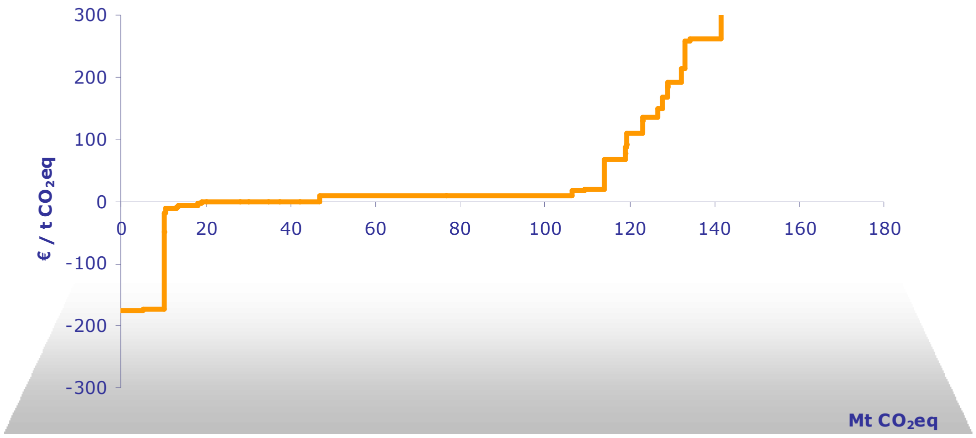

Figure 9: Abatement potential and specific costs of abatement options in the agriculture sector in the EU27 in 2020 (note, approximately 15 Mt of reductions with costs >300 €/tCO_2 are not shown).

The abatement potentials calculated here represent technical potential, it is unlikely that this potential would be realised without strong policy or regulatory assistance. The technical potential is therefore often adjusted to provide an estimate of realisable potential. This study has not attempted to construct curves of realistic potential since for the agricultural sector this is very difficult, since practices and policies with respect to reducing the climate impacts of agriculture have been in place for a relatively time compared to the energy field for example. We suggest that the barriers to delivering the abatement potentials estimated here would include:

- Lack of training and/or knowledge. Implementation of cost effective opportunities available in the agricultural sector requires an understanding of the measures available and their methods of application.
- Even for those measures identified with zero or negative cost, opportunity costs may present a barrier to implementation. For example, for Centralised AD plants, the capital expenditure required to build the plant could be invested elsewhere and potentially at a better rate of return.
- Hidden costs, such as farm workers time for decision making and administration; disruption of production whilst measures are implemented.

Stronger policy or regulatory frameworks that establish financial and behavioural levers could reduce the influence of these factors.

Table 11 Agriculture abatement measures: potentials and specific costs.

	Measures	**Specific costs (€/tCO_2)**	**Mton CO_{2-eq}/ year)**	Cum. Mt CO2-eq year
Soils	Reduce N application through precisison farming	-175	5	5
Soils	Reduced N application through improved spreader maintenance	-173	5	10
Manures	CAD dairy - West (warm)	-48	0	10
Manures	CAD dairy - East (temperate)	-19	0	10
Manures	CAD dairy - West (temperate)	-10	3	13
Manures	CAD pigs - East (temperate)	-8	0	14
Manures	CAD pigs - West (temperate)	-7	5	18
Manures	CAD pigs - West (warm)	-3	1	19
Soils	Reduce N application through fertiliser free zone	-1	1	20
Enteric	Long term management and use of genetic resources (Non-dairy cattle Eastern Europe)	0	5	25
Enteric	Long term management and use of genetic resources (Dairy cattle Eastern Europe)	0	2	27
Enteric	Long term management and use of genetic resources (Dairy cattle Western Europe)	0	3	30
Manures	Reducing the rate of microbial action	0	5	35
Manures	Removal of the gas source	0	8	43
Enteric	Long term management and use of genetic resources (Non-dairy cattle Western Europe)	0	5	48
Soils	Addition of Nitrification inhibitors - Mineral	10	30	78
Soils	Addition of Nitrification inhibitors - Manures	10	29	107
Soils	Reduced grazing on wet areas	18	3	110
Soils	Reduce N application through enhanced distribution geometry	20	5	115
Soils	Reduce N application through allowance for manure/residual N	67	5	120
Manures	On farm AD for dairy - West (warm)	77	0	120
Manures	On farm AD for dairy - East (temperate)	88	0	120
Manures	CAD dairy - East (warm)	93	0	120
Manures	On farm AD for pigs - West (warm)	111	4	124
Manures	CAD pigs - East (warm)	130	0	124
Enteric	Adding oils and oilseeds (Dairy cattle Western Europe)	137	4	128
Manures	On farm AD for dairy - West (temperate)	150	1	129
Enteric	Adding oils and oilseeds (Dairy cattle Eastern Europe)	168	1	130
Manures	On farm AD for pigs - East (warm)	183	0	130
Manures	On farm AD for dairy - East (warm)	186	0	130
Manures	On farm AD for pigs - West (temperate)	193	3	133
Manures	On farm AD for pigs - East (temperate)	214	1	134
Enteric	Adding oils and oilseeds (Non-dairy cattle Eastern Europe)	258	1	135
Enteric	Adding oils and oilseeds (Non-dairy cattle Western Europe)	262	7	143
Enteric	Replacement of roughage with concentrates (Dairy cattle Western Europe)	1,222	3	145
Enteric	Replacement of roughage with concentrates (Dairy cattle Eastern Europe)	1,497	1	146
Enteric	Replacement of roughage with concentrates (Non-dairy cattle Eastern Europe)	2,297	1	147
Enteric	Replacement of roughage with concentrates (Non-dairy cattle Western Europe)	2,338	5	152
Soils	Better livestock nutrient use efficiency - grazing	2,624	3	155
Soils	Better livestock nutrient use efficiency - fertiliser	3,432	6	161

Additional uncertainties exist around the emission factors used to derive the absolute emissions for the sector. We have already stated that in the scope of this project we have been unable to use country specific values for the N-excretion rate of livestock, instead, IPCC (2006) default parameters are used to estimate the emissions associated with, in this example, animal manures. We recognise that this gives rise to errors in emissions estimates; however, we also note that any emissions errors are likely to be an order of magnitude less than the uncertainties around costs and implementation for agricultural measures. For example, for some of measures, e.g. anaerobic digestion, cost information is derived from one member state. It is likely to be the case that costs will differ significantly across Europe reflecting differences in labour costs, the supply chain infrastructure and general experience. This deserves highlighting as an area for consolidation in the future.

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